Carbon Emissions and Evaporation Dynamics at Peatlands Under Active Extraction in Alberta and Quebec, Canada

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the 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.

Statement of Contributions

This thesis has been structured in accordance with the manuscript option, and as a result, there may be repeated information. Chapter 2 has been published, and Chapters 3 and 4 will be submitted for publication. Should these papers get accepted for publication, they may be different than what is in this thesis due to the incorporation of feedback during the peer-review process.

This research is part of a larger NSERC Collaborative Research and Development Grant, funded in part by the Canadian Sphagnum Peat Moss Association. Dr. Nigel Roulet as the principal investigator, and Dr. Maria Strack, Dr. Ian Strachan, Dr. David Olefeldt, and Dr. Tim Moore as co-investigators, acquired the funding and conceptualized the initial project.

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Abstract

The extraction of peatlands for horticultural peat use is a small but important source of peatland drainage in Canada, converting these long-term net sinks of carbon (C) into net sources. With demand for peat expected to grow over the coming decades, there is a need to understand how C emissions and water exchange vary with environmental and site management conditions. Extraction operations alter the site hydrology, peat physical and thermal properties, and peat substrate quality, which in turn alter the C cycling and water budget; however, few studies have quantified this. As part of site preparation for extraction, the surface vegetation is removed and a series of drainage ditches are installed to lower the water table, allowing machinery to access the sites. Exterior ditches surround the perimeter of the sites, while interior ditches run the length of the sites, dividing them into 30 m wide fields of peat bounded by ditches. Sites will be extracted for 15 to 40 plus years using the vacuum harvesting method. As part of this, companies repeatedly disturb the surface by harrowing (tilling) the top ~ 5 cm layer to cut it off hydrologically from the peat below. A portion of this newly dry layer is then harvested, processed and sold for use as a horticultural growing medium.

This study conducted plot and ecosystem scale measurements of carbon dioxide (CO_2), methane (CH_4) and evaporation (E) at multiple actively extracted peatlands of varying extraction durations at sites in both Alberta (AB) and Quebec (QC). Specifically, the research objectives were to: i) investigate spatial distribution and seasonal and interannual patterns of C emissions (Chapters 2 and 3); ii) understand the impact of site management, including drainage, harrowing and drainage ditch maintenance, on C fluxes (Chapters 2 and 3) and E (Chapter 4) and iii) assess the environmental drivers of C emissions and E (Chapters 2, 3 and 4). Plot scale C emissions at AB, and ecosystem scale C and E measurements at AB and QC, were conducted for multiple years between 2019 and 2022, with a focus on the March to October period. A peat incubation experiment and a lysimeter experiment were conducted during summer 2022 at AB to understand the impact of substrate quality on C emissions, and the impact of harrowing on E rates, respectively. Water table depth (WTD), volumetric water content (VWC), soil temperature, and a range of atmospheric variables were also measured throughout the study period.

We found that while C emissions did not vary spatially across fields, drainage ditches were hotspots of C loss, emitting around double, and ten times the CO_2 and CH_4 emissions, respectively, of the fields across a range of VWC. Carbon dioxide emission varied by up to 50% seasonally and

interannually, with differences driven by the interaction between soil temperature and moisture conditions. Our results demonstrated the importance of characterizing hydrological conditions at drained sites, as the temperature dependence of CO_2 emissions increased with increasing moisture content, and CO_2 emissions were significantly higher under heavily drained (WTD > 60 cm) conditions. A non-linear relationship between WTD and E demonstrated the strong upward capillary water movement in these heavily compacted sites, and how adequate surface moisture conditions for E can be maintained across a range of WTDs. Site specific WTD thresholds aided in classifying the relative importance of surface and atmospheric variables on E rates. The impact of harrowing on E rates varied with time, which highlighted the importance of considering the length of time between harrowing and harvesting for management operations. This work will have implications for national greenhouse gas reporting, providing data to support updating C emission factors for Canada. The research can be used by industry partners to inform management practices that balance harvesting yields with C emissions. Additionally, due to the absence of vegetation, these study sites provide a unique opportunity to understand environmental controls of heterotrophic respiration and peatland E, without having to partition out the impact of vegetation.

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Dedication

I dedicate this thesis to my grandmothers: Granny Pat (Patrica Hunter) and Baba (Virginia Humphreys), who paved the way for me to dream big and tackle any challenge. Despite needing my help with texting and taking photos, they taught me what it means to be a strong and independent modern woman.

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List of Abbreviations

AB	Alberta Site	NEE	Net ecosystem exchange
AE	Actual evaporation	PE	Potential evaporation
AR	Autotrophic respiration	QC	Quebec Site
C	Carbon	Q_E	Latent heat flux
CH_4	Methane	Q_G	Ground heat flux
C:N	Carbon to nitrogen ratio	Q_H	Sensible heat flux
CO_2	Carbon dioxide	R_g	Incoming shortwave radiation
d	Day	R_n	Net radiation
E	Evaporation	T	Transpiration
ER	Ecosystem respiration	VPD	Vapour pressure deficit
ET	Evapotranspiration	VWC	Volumetric water content
GPP	Gross Primary Productivity	WT	Water table
HR	Heterotrophic Respiration	WTD	Water table depth
NECB	Net ecosystem carbon balance		

Chapter 1
Literature Review

1.1 Introduction

Northern peatlands are globally important due to their role as long term carbon (C) sinks, accumulating partially decomposed plant matter (Primeau & Garneau, 2021) over thousands of years (Yu et al., 2011) due to greater production versus decomposition rates under waterlogged conditions (Clymo, 1984). Carbon and water cycling are closely linked in peatlands, with the waterlogged conditions controlling the productivity of vegetation and the oxygen availability for microbial decomposition of organic matter (Limpens et al., 2008), which in turn affect the peat hydrophysical properties (Liu & Lennartz, 2019). Peatlands cover ~ 3% (400 to 500 million ha) of the global land surface area, storing an estimated 400 to 1000 Gt of C (UNEP, 2022; Xu et al., 2018). Despite this important role as a C sink, peatland disturbance is increasing worldwide. At least 10% of peatlands have been drained (UNEP, 2022), often converting these sites from net sinks to net sources of C (Frolking et al., 2011; Nugent et al., 2019). It is estimated that carbon dioxide (CO_2) emissions from drained peatlands alone are equal to 3% of our yearly anthropogenic CO_2 emissions (Le Quéré et al., 2015). Northern peatlands are particularly important, as they account for ~85% of the total peatland C storage (Hugelius et al., 2020). Peatlands cover around 12% of the land surface area in Canada (Tarnocai et al., 2011). To project the fate of this large store of C in Canada, it is vital that we understand the mechanisms of production and consumption of C under varying environmental and management conditions (Nugent et al., 2021; Wilkinson et al., 2023).

In Canada, it is estimated that 1.7% (2,082,143 ha) of peatlands have been disturbed through anthropogenic activities, with the largest disturbances due to agriculture (~1,315,000 ha) and mining (370,000 ha) activities (Rochefort et al., 2022). Peat extraction is a small, but important source of peatland disturbance in Canada, with an estimated 0.03% (24,964 ha) of Canadian peatlands having been drained for extraction, either for previous or current use (CSPMA, 2017). It is estimated that 3 Mt of peat is extracted annually in Canada, with this extraction concentrated in Quebec, Manitoba, Alberta and New Brunswick (Natural Resources Canada, 2023). In comparison, around 19.6 Mt is extracted annually across all of Europe (numbers as of 2017; Hirschler and Osterburg, 2022). Importantly, this extracted peat is primary used for horticultural purposes in Canada (Cleary et al., 2005), whereas in Europe around 62% of extracted peat is used as a fuel source (Hirschler and Osterburg, 2022). Despite a rise in peat alternatives in recent years, extracted peat is still the current

preferred growing medium for many in the horticultural community due to its high organic content and water retention properties (Alvarez et al., 2018; Blok et al., 2021). Canada is one of the largest exporters of peat in the world, exporting over $ 500 million worth of peat each year (Government of Canada, 2024). Despite covering a small area, the shift from being a net sink to a net source of C (Clark et al., 2023, Waddington et al., 2002) makes this an important disturbance to study. Current estimates suggest yearly emissions of 2.1 Mt CO_2 equivalent from peat extraction activities (Environment and Climate Change Canada, 2024).

The extraction process alters a peatlands' hydrology, substrate quality and C dynamics. To prepare a site for extraction, drainage ditches are created to lower the water table (WT), and then the surface vegetation is removed. In addition to perimeter ditches, interior ditches running the length of the site are cut and spaced every ~30 m in Canada, dividing the site into fields of peat bounded by drainage ditches (Figure 1.1). The water level in the interior drainage ditches fluctuates based on recent precipitation levels and site management (Cleary et al., 2005). A common practice in Eastern Canada is to contour (dome) the fields to enhance lateral flow into the drainage ditches (Clark et al., 2023). A site is then vacuum harvested, the standard extraction method in Canada, for 15–40 plus years through a repeating two-step process. First, machinery is driven over the surface to harrow (till) the peat to dry the surface layer (3-5 cm) by disrupting the hydrologic connection between the surface layer and the peat below (Figure 1.2). Then, this newly dried layer is removed using the vacuum-harvesting method, where it is further dried in stockpiles, processed, and then sold for use as a horticultural growing medium (Waddington et al., 2009). Depending on precipitation frequency and the surface moisture content, the harrowed layer will remain on the site for half a day to a couple of days, and may be re-harrowed multiple times before the surface peat is extracted (Warren Walker, Sun Gro Horticulture Natural Resource Director, Pers. Comm.). While vacuum-harvesting is the current standard extraction method in Canada, many of the older research sites were based at cutover peatlands that were extracted using the block cutting method (Robert et al., 1999).

To date, research at extracted peatlands has focused on restoring these sites back to net C sinks, with Canada leading the world in the moss layer transfer restoration technique (González & Rochefort, 2014). A few studies have estimated the total C life cycle of peat extraction (Cleary et al., 2005; Waddington et al., 2009), though they lack comprehensive measurements of emissions during the 15 to 40 years of extraction. A substantial source of C loss will come from the physical removal of the peat and its decomposition (Sharma et al., 2024). However, the shift from being a net sink to a net

source of C (Clark et al., 2023), combined with the long extraction duration, means that site emissions during extraction cannot be ignored. Current estimates of site emissions, using process-based modelling, chamber measurements, and eddy covariance measurement range from 151 to over 400 g C m^{-2} yr^{-1} (He et al., 2023; He & Roulet, 2023; Rankin et al., 2018; Waddington et al., 2002). Canada's current Tier 2 emission factor value falls at the high end of this range at 310 g C m^{-2} yr^{-1} (ECCC, 2021). Only a few studies have measured N_2O emissions from extracted peatlands, with their work suggesting that emissions are less than 0.1 g N_2O m^{-2} yr^{-1} (e.g., Hyvönen et al., 2009; Minkkinen et al., 2020), and thus likely constitute a small portion of the total greenhouse gas emissions from these sites.

With the estimated increased demand for horticultural peat over the next few decades (Blok et al., 2021), there is a need for a process-based understanding of how C emissions and water exchange vary with environmental and management conditions. In particular, there is a need to better understand the tight coupling between surface moisture conditions and the associated redox potential, and C cycling in peatlands (Byun et al., 2021; Knorr and Blodau, 2009). Direct measurements of moisture content at extracted sites are difficult due to the need to not interfere with management activities. Additionally, harrowing operations are expected to alter surface moisture conditions on hourly to daily time scales (O'Kane, 1992), and this process has not been studied in situ. Evaporation (E), a key component of the water budget at extracted peatlands (He et al., 2022), has rarely been quantified at extracted peatlands (Petrone et al., 2004; Price, 1996), but can provide important insights on vertical capillary water transport and surface moisture content changes at these sites.

This research will provide the industry with better tools to balance C emissions, both gaseous and aquatic, with harvesting yields. It will also support Canada's accounting of anthropogenic greenhouse gas emissions, and the determination of the C benefit from restoration of these sites. The following sections provide an overview of relevant literature on our understanding of the impacts of peat extraction and important research needs. It will first discuss how active peat extraction alters the peatland's hydrophysical properties and substrate quality. It will then discuss the current state of knowledge, and highlight key research gaps, on how these changes will impact C cycling and E rates in actively extracted peatlands. Based on these identified research gaps, this review will end with a series of research objectives for this thesis.



Figure 1.1 Arial view (left) and side view (right) of a non-countered peat extraction site with ditch spacing of ~ 30 m. The segments of peat in-between ditches are referred to as fields. Arial image: Imagery © 2024 Airbus, CNES/Airbus, Maxar Technologies, Map data © 2024 Google



Figure 1.2 Image of the peat fields after harrowing (left) and after harvesting of the harrowed layer (right).

1.2 Drainage of Peatlands for Extraction

1.2.1 Impact of Drainage on Peat Hydrophysical Properties

Drainage for peat extraction alters several peat hydrophysical properties. Drainage causes peat subsidence and a decrease in the overall peat volume, with shrinkage and oxidation occurring above the WT, and compression occurring below (Holden et al., 2004; Kennedy & Price, 2005; Liu & Lennartz, 2019; Price et al., 2003). Shrinkage is the contraction of the peat pores following WT drawdown due to the negative matric potentials of the drying soil, while oxidation is the physical loss of organic matter due to decomposition (Kennedy & Price, 2005; Price et al., 2003). Below the WT, the increased effective stress following WT drawdown will compress the peat pores, reducing porosity (Van Seters & Price, 2002; Whittington & Price, 2006). In the short term, shrinkage and compression are the dominant controls on subsidence, while in the long term the oxidation of organic matter will be most important (Liu et al., 2020).

Drainage-induced subsidence alters peat pore structure in a few ways. The preferential collapse of larger pores decreases macropore abundance (Kennedy & Price, 2005) and total porosity (Liu et al., 2020) and increases bulk density (Mustamo et al., 2016; Price, 1997). The impact of these changes is not the same with depth, with the majority of volume change occurring in the unsaturated zone (Price et al., 2003; Schlotzhauer & Price, 1999). These shifts in peat physical properties directly affect the peat hydraulic properties. Drainage reduces the specific yield of the peat (Price & Ketcheson, 2009) due to the decrease in pore radius, and the associated decrease in matric potential (more negative) of the pore water at a given soil moisture content (Gauthier et al., 2018; Price, 1997; Price et al., 2003). The decrease in pore size and increased path tortuosity is expected to decrease the saturated hydraulic conductivity (Gharedaghloo et al., 2018; Price et al., 2003) while at the same time increasing the unsaturated hydraulic conductivity at a given matric potential (Carey et al., 2007; Price & Ketcheson, 2009). Drainage also increases WT fluctuations (Holden et al., 2011; Price, 1997). One mechanism is through the decreased specific yield (Price, 1997), which can be define as the ratio between the volume of water that will be drained by gravity from a saturated soil, to the total soil volume (Fetter, 1994). Additionally, peatlands can lose their ability to swell and shrink following subsidence induced compression such that WT fluctuations will increase relative to the peat surface (Howie & Hebda, 2018; Schlotzhauer & Price, 1999).

The land use change associated with peat extraction will further enhance these changes. In preparation for extraction, the surface layer of the peatland is removed. This layer is characterized by larger pore radii, fresh labile litter, and less decomposed peat compared to deeper layers (Clymo, 1984). Its removal will thus expose older, more decomposed peat with smaller pore radii and higher bulk density (Liu et al., 2020; Waddington & Price, 2000). Additionally, peatlands are extracted for 15 to over 40 years, and each bypass with the vacuum harvester will remove a thin layer of the peat (Cleary et al., 2005). This means that these peat hydrophysical properties will also change with increasing time since extraction as peat from deeper within the peat profile is exposed at the surface (Waddington et al., 2002). Finally, the weight of the harvesters will cause further compaction of the pores, amplifying the above-mentioned changes in peat hydrophysical properties, as has been observed in forestry logging practices (Williamson & Neilsen, 2000). These actions will be counteracted slightly by harrowing, which will create a few cm thick layer at the surface that is highly aerated with large pores and a low bulk density that is hydraulically cut off from the peat below it (O'Kane, 1992; Lai, 2022).

The installation of drainage ditches will further alter the peat hydrological dynamics. As mentioned, sectors are divided into 30 m wide fields that are bounded by drainage ditches (Waddington et al., 2002). These ditches create a hydraulic gradient, supporting the lateral flow of water from the peat into the ditches (Price & Ketcheson, 2009). We expect that drainage ditches will create a hydraulic gradient across the fields, with the deepest WTs by the ditches, and the shallowest WTs near the center of the fields, though this is not always the case (Price, 1996; Waddington et al., 2002). The efficiency of a drainage ditch will depend on many factors, including the age and depth of the ditch, and the peat structure and hydraulic properties of the peatland (Landry and Rochefort, 2012).

1.2.2 Impact of Drainage on Peat Quality and Microbial Community

Peat extraction will directly impact the peat biogeochemistry and microbial community structure. The removal of vegetation prior to extraction will eliminate all fresh litter input and root exudates to the peat (Laiho, 2006; Strack & Waddington, 2007) that are important sources of labile C in peatlands (Limpens et al., 2008). The continued extraction of peat will remove the surface layers, exposing older, more recalcitrant peat at the surface with reduced lability compared to surface layers of restored peatlands (Andersen et al., 2010). Extracted peat has lower nutrient availability than natural and restored peatlands (Glatzel et al., 2004), including increased C:N ratios, up to five-fold decreases

in extractable organic nitrogen (Andersen et al., 2006; Basiliko et al., 2007), and twelve-fold decreases in extractable phosphorous concentrations (Basiliko et al., 2007). The substrate is also more recalcitrant, with increased von Post humification indices, and DOC samples containing a significantly higher proportion of humic acids at extracted sites compared to natural sites (Basiliko et al., 2007). Studies have also observed significantly lower microbial biomass at extracted sites compared to natural sites (Croft et al., 2001) with as large as a fivefold decrease (Andersen et al., 2006), as well changes in microbial community structure (Artz et al., 2007). In addition, these peat biogeochemical properties will vary spatially at sites where companies dome the fields. A substrate quality gradient is created across the fields, with older more decomposed peat at the edges of fields compared to the center of fields (Clark et al., 2023).

1.3 Carbon Cycling in Peatlands

The net ecosystem C balance (NECB) describes the sum of the inputs and outputs of C from a system. Its components are the net ecosystem exchange of CO_2 (NEE), methane (CH_4), C monoxide, volatile organic C, dissolved inorganic C, dissolved organic C (DOC) and particulate C (Chapin et al., 2006). The NEE is the largest component of the NECB of a peatland and is equal to the sum of gross primary productivity (GPP) and ecosystem respiration (ER), with negative values indicating uptake, and positive values indicating net release. The GPP represents the total autotrophic fixation of C by the peatland vegetation. The ER is the respiration of CO_2 and is the sum of heterotrophic respiration (HR) from the microbial decomposition of organic matter, and autotrophic respiration (AR) from plants during growth and maintenance (Bansal et al., 2023). Peat thermal dynamics, microtopography, hydrologic conditions, vegetation cover, and peat quality will all affect a peatland's NEE (e.g., Strack et al., 2023).

The redox potential within the peat controls the oxidation and reduction capability of microbes for organic matter decomposition, thus influencing the rate of CO_2 and CH_4 production (Pezeshki & DeLaune, 2012). Carbon dioxide arising from heterotrophic respiration is produced under both oxic and anoxic conditions, though rates are more thermodynamically favourable under oxic conditions (Limpens et al., 2008). Methane is produced by methanogens under strict anoxic conditions once the other terminal electron acceptors have been reduced, and provides less than 1% of the energy as aerobic respiration (Lai, 2009). Methane is transported through the peat profile though either diffusion through the peat matrix, ebullition, or plant-mediated transport. As it travels, it can either be

released from the surface, or oxidized to CO_2 by methanotrophs under oxic conditions (Abdalla et al., 2016), although anaerobic CH_4 oxidation has also been quantified in peatlands (Gupta et al., 2013). As with CO_2, peat thermal dynamics, microtopography, hydrologic conditions, vegetation type and cover, and peat quality will all affect CH_4 production and transport (Feng et al., 2020; Heffernan et al., 2022; Turetsky et al., 2014).

Only a few studies have conducted field measurements of CO_2 and CH_4 emissions from actively extracted peatlands in Canada (Clark et al., 2023; Glatzel et al., 2003; Green et al, 2005), with the majorty having been conducted internationally in Sweden, Estonia, the United Kingdom, and Finland (e.g., Salm et al., 2012; Shurpali et al., 2008; Sundh et al., 2000; Vanags-Duka et al., 2022). While multiple studies have been conducted at post-extracted unrestored peatlands, they have all been based in Quebec, Canada, and many contain non-functioning drainage ditches and partial natural re-vegetation (Glatzel et al., 2003; McNeil & Waddington, 2003; Rankin et al., 2018; Strack & Zuback, 2013; Waddington et al., 2010; Waddington et al., 2002). This could result in altered C cycling due to different hydraulic conditions, and the potential for photosynthetic C uptake. These studies suggest that extracted peatlands are long term sources of CO_2, and a small source of CH_4 (Table 1.1).

To date, the majority of studies have focused on CO_2 emissions, and only from the peat fields. While the drained conditions will limit CH_4 emissions from the fields, some limited research suggests that the ditches are large sources of CH_4 (Clark et al., 2023; Vanags-Duka et al., 2022). With the conventional 30 m spacing of ditches in Canadian extracted peatlands, the ditches make up a small portion of the total site area. However, they can contribute disproportionately to the sitewide emissions, with Clark et al. (2023) measuring double, and eight times, the CO_2 and CH_4 emissions, respectively from the ditches compared to the fields. Research at drainage ditches in forestry and agricultural settings has also highlighted the importance of capturing ditch emissions (Peacock et al., 2021a; Peacock et al., 2021b). It is therefore essential that future studies include both fields and ditches in their measurement plans to accurately estimate sitewide emissions. Additionally, current research has focused on May to October emissions. The importance, and the relevant mechanisms, of freeze thaw dynamics on C emissions (Raz-Yaseef et al., 2017; Wang et al., 2014) has not been quantified in extracted peatlands. The following sections will examine the effect of hydrologic conditions, peat temperature, and substrate quality, on CO_2 and CH_4 emissions from extracted peatlands, and highlight key research gaps.

Table 1.1 Average CO_2 (g C m^{-2} d^{-1}) and CH_4 (mg C m^{-2} d^{-1}) emissions from actively extracted peatlands, and from post extracted unrestored peatlands in Quebec, Canada, using the closed chamber method. Values were converted into the given units when necessary. When measurements were taken in multiple years or at multiple sites, and an average was not given, multiple values are listed.

Source	Location	CO_2 emissions		CH_4 emissions	
		Ditch	Field	Ditch	Field
Actively Extracted					
Clark et al., 2023	Quebec, Canada	2.05	0.90	72.0	9.20
Greenwood et al., 2005	Quebec, Canada		0.18		-0.19
Glatzel et al., 2003	Quebec, Canada		1.3[a]		
Salm et al., 2012	Estonia		0.82		1.4
Vanags-Duka et al., 2022	Latvia	0.35		302	
Shurpali et al., 2008	Finland		0.28[b], 0.59[b]		
Sundh et al., 2000	Sweden	1.0	0.55	270	2.0
Post-Extracted Unrestored					
Waddington and Price, 2000	Quebec, Canada		3.0[a], 3.3[a]		13.2, 3.0
Waddington et al., 2002	Quebec, Canada		3.7, 2.7		
Glatzel et al., 2003	Quebec, Canada		9.9[a]		
McNeil and Waddington, 2003	Quebec, Canada		Up to 1.0[d]		
Waddington et al., 2010	Quebec, Canada	2.2	1.4		
Waddington et al., 2010	Quebec, Canada	2.2	1.4		
Rankin et al., 2018	Quebec, Canada		0.74[c], 0.89[c]		

[a] Emissions were reported as average study period emissions. We assumed a study period of 120 days to convert into g C m^{-2} d^{-1}

[b] Emissions were reported as average study period emissions. We assumed a study period of 214 days to convert into g C m^{-2} d^{-1}

[c] Values are based on reported maximum average CO_2 emissions

[d] Values had to be read off plots, and are thus an estimation

1.3.1 Hydrologic Controls

Hydrologic conditions are primary drivers of peatland CO_2 and CH_4 emissions (Abdalla et al., 2016; Evans et al., 2021) though this has been poorly studied in extracted peatlands. Drainage will increase the pore oxygen concentration at depth, creating conditions that favour aerobic decomposition, thereby increasing CO_2 production and decreasing CH_4 production in the fields (Pezeshki & DeLaune, 2012). The increased oxic portion of the peat profile, combined with the loss of plant-mediated CH_4 transport, will likely increase the proportion of CH_4 oxidized to CO_2 by methanotrophs (Lai, 2009). Measuring oxygen concentration and redox potential in peat profiles is difficult, and thus many studies use water table depth (WTD) as a proxy.

The similar, and sometimes confounding, mechanistic controls on AR and HR can make it difficult to use natural peatlands to understand hydrologic controls on HR CO_2 emissions in non-vegetated systems. Partitioning ecosystem respiration into AR and HR is complex (Järveoja et al., 2020; Rankin et al., 2022) and thus many studies with vegetated peatlands only report ER. Researchers have observed both positive effects, and small to no effect, of WTD on ER, with the strength and direction of these relationships depending on a variety of factors including the antecedent soil moisture conditions, degree of WT drawdown, microtopography and vegetation composition, as well as the partitioning of ER (Ballantyne et al., 2014; Huang et al., 2021; Lafleur et al., 2005; Laiho, 2006; Strack & Waddington, 2007; Strack et al., 2006).

In drained peatlands, studies have generally found that the sustained rewetting, and associated decrease in oxic conditions in the peat profile, will reduce CO_2 emissions and increase CH_4 emissions (Evans et al., 2021; Günther et al., 2020; Nielsen et al., 2023; Wilson et al., 2016). These results align with work at active or post-extraction unrestored extracted peatlands, which have little to no vegetation present. Of those that quantified hydrologic conditions, most observed reduced CO_2 emissions during wet periods (He et al., 2023; He & Roulet, 2023; Holl et al., 2020; Rankin et al., 2018) with this reduction as large as 24% (Waddington et al., 2002). However, the majority of these studies had WTs within the top 50 cm of the peat profile, while we know that actively extracted sites can have WTD greater than 80 cm (Glatzel et al., 2004). Greenwood (2005) generally observed that the largest emissions coincided with the greatest WTD, though the WT only ranged between 40 and 60 cm below the surface. One study did observe around double the CO_2 emissions in a wet year compared to a dry year, but did not quantify the WTD or surface moisture conditions (Shurpali et al., 2008).

Comprehensive CO_2 and CH_4 emissions, paired with WTD and VWC measurements across a range of environmental conditions, has not been a focus of research at actively extracted peatland. These measurements will be necessary to understand hydrologic controls on C emissions for the following reasons: the water table can become disconnected from the surface, the relationship between CO_2 and VWC is likely unimodal, harrowing will alter surface moisture conditions on hourly to daily time scales, there are likely lag times between CH_4 production and CH_4 fluxes from the surface, and hydrologic conditions vary spatially across the site.

Firstly, WTD, while easy to measure, may not always be a good indicator of the surface redox conditions controlling decomposition. The high unsaturated hydraulic conductivity and water retention of extracted peat can allow for the maintenance of high surface moisture content despite deep WTs (McCarter & Price, 2014). Additionally, the WT can become disconnected from the surface soil moisture, with a few studies suggesting this can occur at WTD of greater than 60 cm (Price, 1997; Strack et al., 2009). Soil moisture characteristic curves at extracted sites also suggest that there can be considerable hysteresis over the summer period (Lai, 2002; Schlotzhauer & Price, 1999), further complicating the WT−soil moisture relationship. We therefore need more measurements to understand if and when WTD can be a useful predictor of C emissions.

Secondly, the effect of VWC on CO_2 production is likely unimodal, with peak decomposition rates occurring at an optimal surface moisture content that balances oxygen availability with substrate diffusion (Byun et al., 2021; Castellano et al., 2011; Ojanen & Minkkinen, 2019). The hypothesis that maintaining very deep WTs could lead to reduced CO_2 production due to below optimal surface moisture conditions has not been tested in extracted peatlands, though it has been observed in other heavily drained systems (Strack et al., 2009). A study in a peatland drained for forestry found a negative relationship between WTD and CO_2 emissions when WTDs were greater than 70 cm (Mäkiranta et al., 2009).

Thirdly, harrowing will result in the soil moisture in the top few cm changing hourly to daily, with one study finding that harrowing can reduce the surface layer VWC from ~ 50 % to less than 10% (Lai, 2022). While a thin layer, it is unclear both how much this layer contributes to overall C emissions, and how the low surface moisture content in this layer will affect decomposition rates.

Fourthly, given the deep WTs at these sites, CH_4 production likely occurs deep within the peat profile, meaning that there could be substantial lag times between the CH_4 production and CH_4

emissions from the peat surface. We need studies in non-vegetated systems, with the absence of plant mediated transport and the associated increase in CH_4 oxidation potential, to improve our understanding of these lag times.

Fifthly, extracted peatlands are characterized by increased amplitude and frequency of WT fluctuations compared to natural peatlands (Price, 1997). Water table fluctuations have been shown to increase CO_2 and CH_4 emissions in natural peatlands (Blodau & Moore, 2003; Kim et al., 2021), though this will likely depend on the amplitude, antecedent redox conditions, microbial community composition, and length of time of altered hydrologic conditions. Additional studies in drained systems will allow us to better predict the impact of these fluctuations on heterotrophic C emissions.

Finally, these hydrologic conditions will vary spatially and temporally, and we thus need measurements across a wide range of environmental and management conditions. The presence of internal drainage ditches is expected to create a hydraulic gradient, with the shallowest WTs in the center of the fields, though few studies have quantified this. Waddington et al. (2002) observed the deepest WTD within the first few meters from the ditches in post-extraction peatlands. Water table depth did not decrease linearly with distance from the drainage ditches, however, and they observed the shallowest WTs around 7 and 15 m from the ditches. Surface contouring, or uneven topography created during extraction (Hunter, personal observations), may affect the patterns observed. It is presently unclear how drainage ditch-induced hydrologic gradients will affect spatial patterns of C emissions. A multi-year chamber flux study found an increase in CO_2 emissions, though not CH_4 emissions, with distance from the drainage ditches, though they did not quantify the hydrologic gradient (Clark et al., 2023). The contoured peat fields at their study site makes it difficult to distinguish between the effect of peat quality, and peat surface moisture content, on the resulting C emissions.

These comprehensive studies of paired C emissions and hydrologic conditions must also be made in the drainage ditches, and not just on the peat fields. The emissions from drainage ditches at extracted sites have only been quantified in a few studies, and without a focus on the impact of hydrologic and management conditions (Clark et al., 2023; Sundh et al., 2000; Vanags-Duka et al., 2022). The management and maintenance of the ditches is company- and site-specific and varies with yearly precipitation patterns. The ditches can be with or without standing water, and the ditches can also become filled in with surface peat during management operations (Hunter, personal observations,

Figure 1.3). The anoxic conditions when the ditches are saturated with water should lead to higher CH_4 emissions, and lower CO_2 emissions (Bansal et al., 2023). However, it is possible that the standing water in the drainage ditches could support CH_4 oxidation (Strack et al., 2004). Given that the ditches can contribute disproportionately to sitewide C emissions (Clark et al., 2023; Köhn et al., 2021; Peacock et al., 2021a), it is essential that future studies not only quantify these emissions, but also attempt to understand their hydrologic drivers.



Figure 1.3 Images showing a drainage ditch with standing water (left) and a partially filled in drainage ditch (right).

1.3.2 Peat Temperature Controls

Soil temperature is a well-studied driver of CO_2 production, with both field-based and laboratory experiments observing a positive exponential relationship between soil temperature and HR due to increased microbial activity under warmer conditions, with Q10 values ranging from 1−5 (Ballantyne et al., 2014; Byun et al., 2021; Chen et al., 2018). The strength of this relationship will be affected by the moisture availability, with the majority of studies reporting stronger CO_2–temperature

relationships in drier peatland soils (Byun et al., 2021; Liu et al., 2024). However, few studies have included heavily drained sites in their analysis. Research in forest and agricultural settings suggests that when moisture contents are below the optimal level for decomposition, increasing temperature can be a stronger control in wetter conditions (Balogh et al., 2011; Scott-Denton et al., 2006). Soil temperature is also positively correlated with CH_4 emissions, though the strength of this relationship depends on vegetation cover, hydrologic conditions, and time of year (Chang et al., 2021; Turetsky et al., 2014). In drained sites, peak CH_4 production will occur deep within the profile, meaning that surface temperatures may not be relevant for understanding CH_4 production. In very dry conditions, increased temepratures can lead to reduced CH_4 emissions as a consequence of increased CH_4 oxidation rates following warming (Munir & Strack, 2014). Lag times between production and transport to the surface can also complicate the CH_4-temperature relationship (Blodau & Moore, 2003; Estop-Aragonés & Blodau, 2012).

Research on the effect of soil temperature on C emissions at extracted peatlands has produced mixed results. Field-based and modeling studies have found strong positive (He et al., 2023; Rankin et al., 2018), moderate (Hunter et al., 2024; Salm et al., 2012; Waddington et al., 2002) and weak to no effect (Clark et al., 2023; Sundh et al., 2000) of soil temperature on CO_2 emissions. In an incubation study, Kendall (2021) observed nine times higher CO_2 production in extracted peat incubated at 22 °C compared to 3 °C. These conflicting results may have to do with the variety of moisture conditions measured at each site, and the lack of a standard soil temperature measurement depth. Understanding the effect of temperature on C emissions will be important given that extracted peatlands may undergo larger temperature fluctuations than natural peatlands due to the altered thermal conductivity (Waddington et al., 2001).

1.3.3 Peat Quality Controls

The low substrate quality and reduced microbial biomass in extracted peatlands (Basiliko et al., 2007) has been shown to decrease the production potentials of CO_2 and CH_4 compared to in natural peatlands (Andersen et al., 2006; Glatzel et al., 2004; Waddington et al., 2001). More research is needed to understand how emissions will vary with time since extraction. The continual vacuum-harvesting of peat from sites will expose older, more recalcitrant peat to the surface each year that is more resistant to further decomposition (Waddington & Price, 2000). Previous studies have observed no significant difference in aerobic production rates of CO_2 from 2- and 7-year post-cutover peatlands

(Waddington et al., 2001), as well as a lack of depth dependence of potential CO_2 and CH_4 production (Waddington & Day, 2007; Waddington et al., 2001). Others have found that CO_2 production potentials did decrease with increasing degree of peat decomposition at extracted sites (Glatzel et al., 2004). Clark et al. (2023) determined that the effect of substrate quality on CO_2 emissions might depend on the degree of recalcitrance of the peat. They conducted research at contoured vacuum-harvested peatlands of different extraction durations. They found that at newly extracted sites (less than 5 years), the substrate quality gradient across the fields resulted in significantly different CO_2 emissions with distance from the drainage ditches. The substrate quality gradient was no longer important for determining emissions at sites that had been extracted for longer than 5 years and thus had exposed peat from deeper into the peat profile. Similarly, CO_2 emissions were significantly highest in the first five years following the start of extraction, and after that, there was no significant reduction in emissions with time. They found no effect of the duration of extraction or substrate quality on CH_4 emissions.

The effect of substrate quality and time since extraction on C emissions will have important implications for national C accounting. There is a desire to update C emissions factors for extracted peatlands across Canada. Clark et al. (2023) proposed using one set of emission factors for sites within the first five years of extraction, and a second set for sites with longer extraction durations. Clark et al.'s study was conducted at one bog complex. We need C emission data from multiple bog complexes, with multiple extraction durations, to validate this theory.

1.4 Impact of Extraction on Evaporation

Peatland evapotranspiration (ET) is often modelled using the Penman-Monteith approach (Monteith, 1965), where ET is controlled by the available energy, the atmospheric demand for water vapour, represented by the VPD, and the aerodynamic and surface resistances (or conductance) to ET (Helbig et al., 2020; Peichl et al., 2013; Runkle et al., 2014). Meteorologically, VPD is the difference between the partial pressure of water vapour in a parcel of air if it were saturated, and the current vapour pressure, and is temperature dependent (Ahrens and Henson, 2019). At extracted peatlands, the removal of vegetation simplifies ET to just evaporation (E). This section will discuss how peat extraction, both the initial site setup, and the subsequent extraction processes, alter the relative importance and the mechanistic pathways of these controls.

The energy balance, given by

$$Q^* = Q_E + Q_H + Q_G \quad \text{(Equation 1.1)}$$

Which describes the sum of incoming and outgoing energy fluxes from a surface, where Q* is the net radiation at the surface, Q_E is the latent heat flux, Q_H is the sensible heat flux, and Q_G is the ground heat flux (Lafleur, 2008).

The extraction process alters the energy balance at these sites, though few studies have quantified these changes. The absence of trees and shrubs should increase the incoming solar radiation received at the surface (Thompson et al., 2015), thus increasing Q* and the available energy for E. Harrowing and harvesting operations will alter the surface moisture content (O'Kane, 1992), and thus albedo, on hourly to daily time scales. The impact of extraction processes on Q_G is hard to predict, and is likely dependent on the specific hydrologic conditions, which as previously mentioned, can range seasonally, and between sites. Drainage and the associated decrease in moisture content reduces the heat capacity of the peat, but also reduces the thermal conductivity of the peat (Kellner, 2001; Moore et al., 2013). Previous energy balance measurements at extracted peatlands (Petrone et al., 2004; Price, 1996; Price et al., 1998; Waddington & Price, 2000) have all been based in Quebec, Canada. Their work suggests that Q_G constitutes less than 15% of the total energy balance (Petrone et al., 2004; Price, 1996; Price et al., 1998). Despite being a drained system, the available energy is largely partitioned to Q_E, with Bowen ratios (ratio of Q_H to Q_E) as low as 0.13 during a wet year, and averaging ~ 0.7 in a dry year (Petrone et al., 2004).

Extraction also alters the control of VPD on E rates. Evapotranspiration in natural peatlands generally increases with increasing VPD (e.g., Helbig et al., 2020). However, this relationship varies with vegetation cover, as the regulation of their stomata by vascular plants forms a negative feedback mechanism that conserves water under high VPDs (Humphreys et al., 2006; Speranskaya et al., 2024). In the absence of vegetation, a saturating effect of VPD on E rates can still occur, during periods when the moisture content at the surface cannot sustain E rates to meet the atmospheric demand (Waddington et al., 2015). To date, no study has investigated the relationship between VPD and E in extracted peatlands, and in particular how this relationship varies with WTD and VWC.

Extraction also alters the aerodynamic and surface resistances to E. The aerodynamic resistance is controlled by the wind speed and surface roughness (Thom, 1975). The removal of surface vegetation increases the turbulence movement at the surface, thereby decreasing the aerodynamic resistance to E (Kellner, 2001). The surface resistance is the sum of resistances (or conductances) associated with the

E of intercepted water by plants, the E from the soil and moss surface, and the T by plant stomata (Stoy et al., 2019; Tran et al., 2015). Plant stomatal resistance is unique to each vascular plant species (Lafleur, 2008), and decreases with increasing VPD (e.g., Popović, 2024). Under the big leaf model, the E plane is assumed to be a homogenous vegetated surface with one E element (Monteith, 1965), though in practice, two-layer models are often used to estimate an integrated surface resistance (or conductance) to ET (Ershadi et al., 2015; Shuttleworth and Wallace, 1985).

In the absence of vegetation, surface resistance at actively extracted peatlands can be simplified to the resistance associated with the E from bare soil. This resistance is the reduction in E rates in non-saturated soils due to suction of water by the soil pores and the rate of water vapour flow through the unsaturated soil (Philip, 1957). It is related non-linearly to moisture content and pore structure (Lehmann et al., 2018), and is commonly parameterized using WTD and surface tension (Kettridge & Waddington, 2014; Sonnentag et al., 2008). Extraction and its associated hydrologic changes are expected to alter surface resistance in several ways. Site drainage and the increased WTD decreases the overall moisture content at the surface, thereby increasing surface resistance. However, drainage induced subsidence, and the associated hydrophysical changes may partially counteract this. Up until a certain WTD threshold, the increased unsaturated hydraulic conductivity at a given matric potential allows for maintenance of a high surface soil moisture content despite the decline in WT (McCarter & Price, 2014; Price & Ketcheson, 2009), limiting the increase in resistance (Moore et al., 2013). However, an increase in WTD beyond this threshold, or surface drying at a rate greater than the upward capillary recharge, can disrupt this, and result in an increase in surface resistance to E (Alekseychik et al., 2018; Kettridge & Waddington, 2014). Studies in vegetated peatlands have observed mixed results on the importance of WTD on ET (Lafleur et al., 2005; Moore et al., 2022; Sonnentag et al., 2010) which is likely explained by differences in WTD, vegetation succession, and distinct vegetation covers with different rooting depths and stomatal resistances. Extrapolating these findings to extracted peatlands is difficult given the removal of vegetation, and the altered peat structure (Liu et al., 2020) and hydrologic conditions (Price, 2004; Price & Ketcheson, 2009), which will all affect how surface resistance varies with WTD.

To our knowledge, only four Quebec-based studies have measured E from extracted peatlands, with a general focus on understanding the impact of restoration efforts on E (Petrone et al., 2004; Price, 1996; Price et al., 1998; Waddington & Price, 2000). They demonstrate that despite the absence of transpiration from vascular plants, E rates from extracted peatlands can be comparable to ET rates

from adjacent natural sites (Petrone et al., 2004; Price, 1996). The efficient upward capillary transport in extracted peatlands due to changes peat hydrophysical properties can support the maintenance of high E rates (Price & Ketcheson, 2009; Waddington & Price, 2000). However, generalizing these findings to all extracted peatlands will be difficult. It is unclear across what range of WTDs their E rates are representative, which will be important given the wide range of observed WTDs in extracted peatlands (Glatzel et al., 2004; Lai, 2022; Price 1997). Research is other climatic zones outside of Quebec will allow us to better understand how the WTD–E relationship varies with VPD. Investigating the controls on energy partitioning, and in particular the controls on Q_E , and thus E, will be essential for understanding the water budget at extracted peatlands. Given the tight coupling between hydrology and C cycling (Byun et al., 2021; Evans et al., 2021), improving our understanding of site water management will aid peat producers in balancing peat yields with greenhouse gas emissions.

Additionally, previous studies did not account for the impact of site management, namely the harrowing and harvesting process, on E. Prior to harrowing, the peat surface layer is composed of dense, wet peat. Evaporation rates will approach stage one E, where E occurs at the surface and is controlled by atmospheric conditions (Lehmann et al., 2008). A system will move into stage two E when a dry surface layer is formed, moving the E plane below the soil surface (Lehmann et al., 2018; O'Kane, 1992). Harrowing is an external process that will push the system more quickly from stage one to stage two E. Harrowing decreases the surface peat bulk density, thereby breaking the capillary connection with the peat below it (Waddington & Price, 2000). This, in combination with the greater surface area exposed to wind, increases E initially compared to pre-harrowing conditions as the surface layer dries out (Jalota and Prihar, 1990). However, this new dry layer then increases the surface resistance to E, moving the system into stage two E with lower E rates (O'Kane, 1992). While this process has been modelled (O'Kane, 1992), it has never been quantified at an extracted peatland. The effect of tillage on E rates in agriculture settings has shown similar results (Jalota and Prihar, 1990), though direct comparison is difficult given that many no-till procedures are performed in combination with crop residue addition (Verhulst et al., 2011). It is presently unclear how the frequency of harrowing, and the time between harrowing and harvesting, will affect E rates at these sites. This research will be important for the peat industry to optimize their management practices.

Measurements of E at extracted sites will further our scientific understanding of bare peat E. The partitioning of evapotranspiration into E and T is a current challenge, and there is a need for field-

based measurements of these variables to improve modelling efforts and better constrain assumptions (Stoy et al., 2019). Non-vegetated sites, such as extracted peatlands, provide an opportunity to understand E rates, and their environmental controls, without the need to make assumptions about how to partition out E from ET.

1.5 Objectives

Given the knowledge gaps identified in the literature review, the goal of this thesis was to understand gaseous C cycling and evaporation rates in peatlands under active extraction in Canada, with a focus on hydrologic controls and the impact of active site management. The specific objectives were to:

1. Investigate the plot scale spatial distribution of CO_2 and CH_4 emissions, focusing on both field versus ditch emissions, and emissions across the fields (Chapter 2).

2. Assess seasonal and interannual patterns of CO_2 and CH_4 emissions (Chapter 3).

3. Determine the impact of WTD, VWC, soil temperature, as well as their interaction, on CO_2 and CH_4 emissions at the plot and ecosystem scale (Chapter 2, 3).

4. Quantify E rates and assess the relative importance of surface moisture conditions and atmospheric demand on available energy partitioning (Chapter 4).

5. Investigate the impact of harrowing on E rates at both the hourly and daily time scale (Chapter 4).

1.6 Organization of Thesis

This thesis is organized into five chapters. The first chapter provides an overview of the peat extraction process in Canada and discusses previous research and knowledge gaps on how changes to peat hydrophysical properties and substrate quality will alter C cycling and E dynamics at actively extracted peatlands. Chapter 2 focuses on plot scale C emissions, including the spatial distribution, the effect of ditch management on C emissions, and the environmental drivers of C emissions at the plot scale (Objectives 1 and 3). Chapter 3 addresses ecosystem scale C emissions, with a focus on

how hydrologic conditions and soil temperature drive seasonal and interannual variability (Objectives 2 and 3). Chapter 4 investigates the surface and atmospheric drivers of E, as well as the impact of site management (Objectives 4 and 5). Chapter 5 summarizes the main findings of this research, discusses management implications of the research, and suggests priorities for future research. Appendices A, B and C provide supplemental information for Chapters 2, 3, and 4, respectively.

Chapter 2

Environmental and management drivers of carbon dioxide and methane emissions from actively-extracted peatlands in Alberta, Canada

2.1 Abstract

The installation of drainage ditches and removal of vegetation in preparation for vacuum harvesting alters the C dynamics of peatlands. However, we lack the measurements to understand the spatial distribution and environmental and substrate quality controls of carbon dioxide (CO_2) and methane (CH_4) emissions, as well as how these factors change over the 20 – 30 year extraction period. For three summers, we measured CO_2 and CH_4 emissions using the closed chamber method at three actively extracted peatlands near Drayton Valley, Alberta, ranging from 2 to 28 years since the start of extraction. Measurements were made in the ditches, and on segments of peat (fields) between adjacent ditches. Field emissions did not change with distance from ditches, likely due to the observed homogeneity of volumetric water content (VWC) and temperature across the fields. Understanding C dynamics in the ditches will be important, as they emitted on average two and ten times, respectively, the amount of CO_2 and CH_4 per square meter of the fields. We found moderate to weak relationships between C emissions and soil temperature, VWC and ditch water level, though ditch emissions were significantly reduced when there was standing water present. Altering conventional site management, such as increasing ditch spacing, could substantially reduce CH_4 emissions from the managed area. C emissions did not decrease with time since start of extraction. We suggest that Canadian emission factor calculations for land-based emissions consider both peat quality variations among sites, and a site's extraction duration, which has been important in other studies.

2.2 Introduction

Peatlands are wetlands that accumulate partially decomposed plant matter (peat), due to greater production than decomposition under waterlogged conditions (Abdalla et al., 2016; Clymo, 1984; Primeau and Garneau, 2021). They have been accumulating this C-rich organic matter over thousands of years (Roulet et al., 2007). Peatland drainage shifts peatlands from net C sinks to net C sources through changes in peat hydrophysical properties, substrate quality, redox conditions and vegetation cover and productivity (Basiliko et al., 2007; Kennedy and Price, 2005; Liu et al., 2020; Strack et al.,

2006). Nearly 11% of global peatlands have been drained to date (Fluet-Chouinard et al., 2023, Frolking et al., 2011), and burning and oxidation of this peat accounts for an estimated 5 % of anthropogenic greenhouse gas emissions (Joosten et al., 2016). There is a need for more research to understand the mechanisms of C emissions from drained peatlands to help us meet global greenhouse gas emission reduction targets.

One reason for peatland drainage in Canada is the extraction of peat for horticultural use as a growing medium (Cleary et al., 2005). In preparation for extraction, the sites are divided into peat fields, usually 30 m wide, bounded by drainage ditches (around 1 m deep by 1 m wide), and then the surface vegetation is removed. In eastern Canada (i.e., the provinces of Québec and New Brunswick), fields are often contoured to create a domed shape to aid drainage from the centre of the field to the ditches. This practice is not common in western Canada (e.g., the province of Alberta), with these fields being referred to as non-contoured fields. Vacuum harvesting is the primary extraction method in Canada, replacing the older block cutting method (Robert et al., 1999). Harrowing machines break up the top ~5 cm peat layer to aerate it and cut it off hydrologically from the peat below, and then a small portion of that is vacuum harvested and then processed based on its end use (Cleary et al., 2005; Waddington and Price, 2000). An estimated 10 cm of peat is removed each year, and sites can be extracted for over 40 years, depending on the initial peat depth and harvesting rate (Robert et al., 1999). The decomposition of the extracted peat has been estimated to contribute 71% of the total emissions of the peat extraction life cycle (Cleary et al., 2005). The land-based emissions, while only accounting an estimated 15% of the total emissions (Cleary et al., 2005), are nevertheless an important source of emissions that should be studied.

Peat extraction has been shown to alter the peat hydrophysical properties and substrate quality in the actively extracted fields (e.g., Basiliko et al., 2007; Price, 2003). Drainage increases the thickness of the unsaturated layer, shifting redox conditions towards aerobic decomposition (Pezeshki and DeLaune, 2012). Drainage induced subsidence, combined with the compaction from machinery and the removal of the vegetation and upper peat layer, increases the peat bulk density and relative proportion of micropores (Price, 1997; Price, 2003). The associated decreased specific yield (Mustamo et al., 2016; Price and Ketcheson, 2009) increases water table fluctuations and water retention at the surface, even under periods of deep water tables (Holden et al., 2011; Price, 1997). The drainage ditches also create a hydraulic gradient across the 30 m wide peat fields, facilitating lateral drainage of water into the ditches (Price and Ketcheson, 2009). The initial drainage and

removal of the vegetation and the upper portion of the acrotelm, along with the continued peat removal during active extraction, exposes older, more recalcitrant peat at the surface each year, decreasing the substrate quality for decomposition (Clark et al., 2023). Studies have observed decreased inorganic nutrients in the peat including nitrogen and phosphorus in extracted peatlands (Basiliko et al., 2007; Reumer et al., 2018), along with lower microbial C biomass (Anderson et al., 2013; Basiliko et al., 2007; Croft et al., 2001).

These changes in hydrology and substrate quality are expected to alter C dynamics, though only a few studies have measured carbon dioxide (CO_2) and methane (CH_4) emissions, two important components of the net ecosystem C balance, at sites under active extraction. Studies in Canada (Clark et al., 2023; Waddington et al., 2009), and internationally (Alm et al., 2007; Salm et al., 2012; Shurpali et al., 2008; Sundh et al., 2000; Wilson et al., 2015) have shown that actively extracted peatlands are net sources of CO_2, due to the loss of photosynthetic uptake of CO_2, and the increased thickness of the oxic layer. These sites are small sources to sinks of CH_4 (Alm et al., 2007; Salm et al., 2012; Sundh et al., 2000; Waddington et al., 2009), though few studies have taken measurements in the drainage ditches, which can have seven times the emissions per unit area as the fields (Clark et al., 2023). Globally, drainage ditches in disturbed peatlands have been shown to be large sources of CH_4 (Peacock et al., 2021a). The annual removal of peat has been shown to significantly decrease CO_2 emissions, though not CH_4, emissions in the first few years of extraction, with interannual variation in the years following likely due instead to environmental conditions (Clark et al., 2023). Research has shown that the decreased substrate quality at extraction sites reduces CO_2 (both aerobic and anerobic) and CH_4 production compared to natural sites (Glatzel et al., 2004; Waddington et al., 2001). Finally, work at post cutover and post vacuum harvested peatlands confirms that these sites continue to be net sources of CO_2 prior to restoration (Rankin et al., 2018; Waddington and Price, 2000; Waddington et al., 2002).

There is a need for more multi-year studies that go beyond quantifying the CO_2 and CH_4 emissions, and seek to understand the spatial distribution and environmental controls of these emissions. The effects of peat temperature, water table depth and volumetric water content (VWC) on C fluxes have been studied extensively in natural peatlands (Ballantyne et al., 2014; Lai et al., 2014; Laiho, 2006; Strack and Waddington, 2007), but not in actively extracted peatlands. Clark et al. (2023) and Sundh et al. (2000) found weak to no relationships between these variables and C emissions at peatlands under active extraction. To our knowledge, no study in Canada has assessed the effect of drainage

ditches on field C emissions on non-contoured fields. The absence of vegetation at these sites also provides an opportunity to further our understanding of environmental controls of heterotrophic respiration, without having to detangle the effects of vegetation productivity and decomposition on ecosystem respiration. Therefore, the research objectives of this study were to: i) assess the spatial distribution of CO_2 and CH_4 emissions in actively extracted peatlands, focusing on both field versus ditch emissions, and emissions across the fields; ii) understand the effect of soil temperature, ditch water level and VWC on these C emissions, and iii) determine whether these environmental drivers change with time since the start of extraction.

2.3 Materials and Methods

2.3.1 Study Sites

We conducted measurements at peat extraction sites near the village of Drayton Valley, Alberta (53.557° N, 114.736° W). Three sites representing long (Old Site: extraction began in 1991), medium (Intermediate Site: extraction began in 2010) and short (Young Site: extraction began in 2017) extraction durations were chosen to represent the range of conditions of actively extracted peatlands undergoing vacuum harvesting. The Old and Intermediate Sites were part of the same peatland complex while the Young Site was located 29 km northwest. The peat sites are divided into ~30 m wide fields of peat bounded by drainage ditches. To capture the within-site spatial variability, we placed transects perpendicular to the interior drainage ditches, and measurements were made in the ditches, and on the field 2 m, 5 m, and 15 m from the ditches. The 2 m, 5 m, and 15 m sampling positions will be referred to as field positions. Transects were placed equal distance apart along a field, and they began 20 m from the start of the perimeter drainage ditches. The number and location of transects changed over the course of the study due to both site management by the peat producer, and our own resources. The drainage ditches at the Young Site in 2021 and 2022 were partially to fully filled in with peat due to operations on site (See Text A1 for further details).

2.3.2 CO_2 and CH_4 Chamber Fluxes

CO_2 and CH_4 fluxes were measured at each flux position using the dynamic closed chamber technique (Griffis et al., 2000). At the field positions we used a removable metal collar (0.58 × 0.58 m) and metal or plastic chamber (0.6 × 0.6 × 0.3 m). In the drainage ditches, a cylindrical floating plastic chamber (height: 26 cm, diameter: 28 cm) with a Styrofoam cylinder, or an acrylic collar

(diameter 18.8 cm) and chamber (height: 23 cm, diameter: 18.8 cm) were used. The concentration of CO_2 and CH_4 gases in the chamber headspace was measured for 2–3 minutes using a trace gas analyzer. For CO_2 emissions, both an EGM-4 infrared gas analyzer (PP Systems, Amesbury, MA; used in June and July 2019, and July 2021) and a LI-7810 (LI-COR, Lincoln, NA; used in July and August 2019, August 2021, and May to August 2022) were used. CH_4 emissions were measured using the LI-7810. The collars were installed prior to each measurement, and then removed afterwards, to not interfere with extraction activities. Carbon gas sampling was conducted a few minutes after installing a collar. Measurements were made once to twice a week from July to August in 2019, in August 2021, and from May to August 2022. Ambient air temperature in the chamber was measured using a thermocouple wire and portable thermocouple reader (OMEGA, Saint-Eustache, QC).

CH_4 fluxes were divided into steady and episodic fluxes. The steady flux includes diffusion and the steady release of bubbles, while episodic ebullition refers to the rapid release of bubbles (Green and Baird, 2012). As we did not observe episodic fluxes of CO_2, only steady fluxes were calculated. We assessed CH_4 release via episodic ebullition using methods adapted from Goodrich et al. (2011) and Nugent et al. (2021). Episodic fluxes were identified in 39.3 % (11 fluxes), 0% and 6.9 % (7 fluxes) of Intermediate Site ditch fluxes in 2019, 2021 and 2022 respectively. They occurred in 57% (4 fluxes), 0% and 0% of Young Site ditch fluxes in 2019, 2021 and 2022 respectively, and in 19% (6 fluxes) at the Old Site in 2019. See Text A2 for details on the calculation of CO_2 and CH_4 fluxes. Instantaneous values were expressed as a daily rate, which assumed that the rate over the diurnal cycle was constant.

2.3.3 Environmental Variables

Monthly and 30-year (termed ‘normal’) precipitation and air temperature data were obtained from the Tomahawk AGDM station (53.43°, -114.72°; located ~11 km southwest of the Old and Intermediate Site) and the Evansburg2 AGDM station (53.61°, -115.06°; located ~ 8.5 km southeast of the Young Site; ACIS, 2023). Following a C flux measurement, soil temperature was measured at depths of 5, 10, 15, 20 and 30 cm using a portable thermocouple probe (HH200A Handheld Digital Thermometers, OMEGA, Saint-Eustache, QC) at each collar. In the case of standing water in the ditches, average water column temperature was measured. We measured volumetric water content of the top 20 cm of peat using a time domain reflectometry (TDR) probe (HydroSense II Handheld Soil

Moisture Sensor, Campbell Scientific, Edmonton, AB). Due to extraction activity, water table depth could not be regularly measured at each sampling point as permanent wells could not be installed on the sites. Water table depth was measured hourly at the center of an Intermediate Site field using a level logger (Van Essen Micro-Diver Water Level Loggers, Van Essen, Instruments Canada, Waterloo, ON) from August 1, 2019, to August 10, 2022. This well was located within a harvesting exclusion zone for the duration of our measurements.

2.3.4 Carbon Gas Production from Fields Versus Ditches

To understand the influence of soil moisture and substrate quality on differences in CO_2 emissions from the fields versus the ditches, an incubation experiment was conducted during June and July 2022. Twelve peat grab samples of a known volume (~0.5 L) were collected from both the ditch and 2 m position at two transects at the Intermediate site (4 locations × 3 replicates). At the Young site, nine peat samples (3 locations × 3 replicates) were collected from a transect at the 2 m position, the ditch top, and ditch bottom (about 80 cm below the field surface). The ditch top refers to the current peat surface in the ditch at the time of sampling. The ditch bottom refers to the ditch surface that was exposed when the ditches were dug out prior to the start of peat extraction in 2022, and the subsequent filling in of the ditches. Replicate samples were combined to create seven homogenous bulk samples, one from each sampling location. The equivalent of 5 g of dried peat bulk sample were added to sixty-three 500 mL jars (7 locations × 3 VWC treatment levels × 3 replicates). For each sampling location, we incubated triplicates of peat at treatment levels of 40%, 60% and 80% VWC. Based on the bulk density and VWC of each bulk sample, peat was either air dried, or distilled water was added to reach these desired VWC levels. A portion of each bulk sample was dried in an oven at 70 ºC for over 48 hours and weighed to estimate its bulk density and VWC.

Fluxes of CO_2 and CH_4 were measured three times a week for the first two weeks of the experiment, and then weekly for the final four weeks using a LI-7810 trace gas analyzer (LI-COR, Lincoln, NA). Ambient air temperature was measured using a thermocouple wire and portable thermocouple reader (OMEGA, Saint-Eustache, QC). During measurements, the jars were sealed, and the gas concentrations were corrected for volume and temperature using equation 1. In between sampling events, the lids were left partially on the jars to reduce evaporative losses, while also avoiding the creation of anoxic conditions. The jars were weighed weekly, and distilled water was added to compensate for evaporative losses.

2.3.5 Peat Substrate Quality

Peat humification indices (using Fourier transform infrared (FITR) spectrometry), C to nitrogen ratio (C:N) and peat age (using ^{14}C dating) were determined to investigate substrate quality differences across our sites. Peat samples of known volume, ranging from 400 to 1000 cm^3 were collected for this analysis. In August 2021, we collected samples at the 2 m position from a transect at the Intermediate Site at depths of 0.8 m, 2.05 m and 3.25 m (See Figure 2.1 for schematic of sampling locations). This was repeated at the Young Site at depths of 0.8 m and 1.85 m. The deepest samples from both sites were located directly above the mineral layer. We also collected samples from the surface at the 2 m and 15 m position at the Intermediate Site and the 2 m position at the Young Site. In August 2022 at the Intermediate Site, we collected samples from the ditch surface (91 cm below field surface), and 45 cm below the ditch surface (136 cm below field surface). At the Young Site, the same samples were collected at the surface of one of the partially filled in ditches (15 cm below the field surface) and 10 cm below the original ditch surface (88 cm below the field surface). The original ditch surface refers to the ditch surface when the ditches were dug out in fall 2021, prior to them being partially filled in by machinery during extraction. This surface was identified visually by its greater bulk density. See supplemental section Text A3 for details on analysis methods.

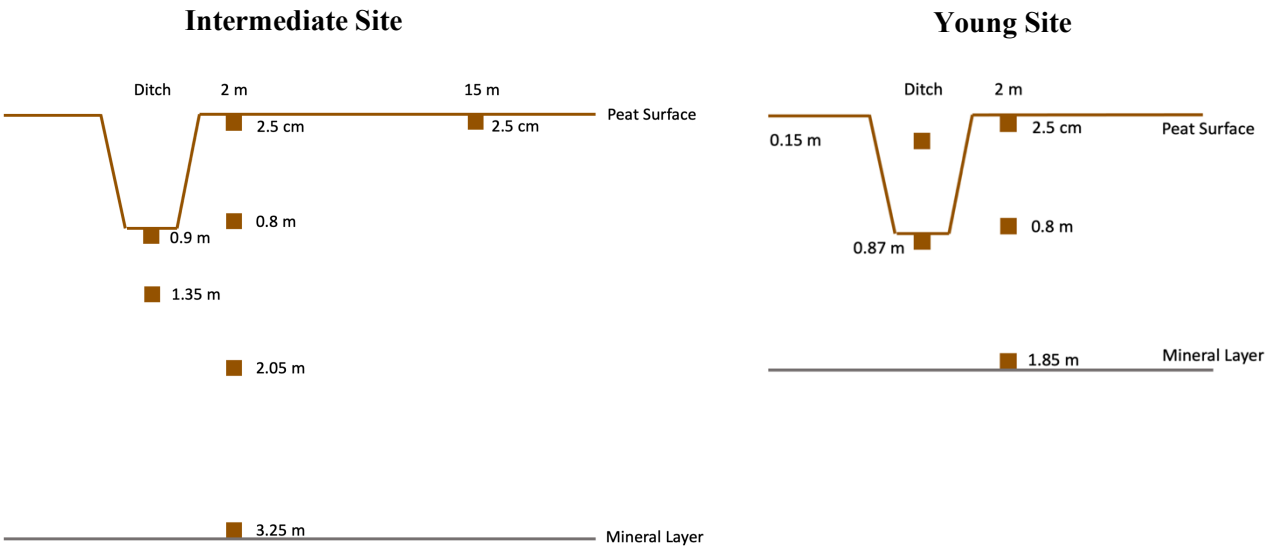


Figure 2.1 Schematic of sampling design for collecting of peat for ^{14}C, C:N and FTIR analysis at the Intermediate and Young Sites. The brown and grey lines represent the approximate peat surface and depth of the mineral layer. The horizontal axis is not to scale. The depths provided are the depth below the field surface.

2.3.6 Statistical Analysis

Statistical analysis was performed in RStudio (R Core Team, 2019; RStudio Team, 2020) and all figures were produced using the ggplot2 package (Wickham et al., 2016). The chamber and incubation fluxes were log transformed to meet the assumptions of normality for all statistical tests; prior to transformation, the chamber-based CO_2 and CH_4 fluxes were shifted by 0.2 mg C m^{-2} d^{-1} and 11.1 mg C m^{-2} d^{-1}, respectively to remove the negative values. Normality was assessed through visual interpretation of Q-Q plots (Zurr et al., 2009).

For the chamber flux data, we used a linear mixed effects model (LME) with Collar as a random factor (lme4 package; Bates et al., 2015) and a variance structure with Position to account for unequal variances of CO_2 and CH_4 emissions between Positions. A linear model was used for the incubation flux data as jar was not a significant random factor (Chi-squared test, p=0.93) for an LME. A series of one and two-way analysis of variances (ANOVAs) were performed on the LME and linear models to determine the effect of fixed effects on CO_2 and CH_4 emissions. When fixed effects were significant,

the Tukey post hoc test (emmeans package; Lenth et al., 2018) was used. Comparisons were not made between years due to differing sample sizes and date range of data collection.

Multiple LME model regressions were performed to assess the effect of volumetric water content and ditch water level individually on CO_2 and CH_4 chamber fluxes. For soil temperature, both LME and non-linear regressions were run to determine the effect of soil temperature from the Field (field positions combined) and Ditch positions on CO_2 and CH_4 emissions (See Text A4).

2.4 Results

2.4.1 Environmental Conditions

The 30-year average total growing season (May to August) precipitation and average air temperature are 305.1 mm and 14.6 °C, and 308.2 mm and 14 °C at Tomahawk and Evansburg, respectively (ACIS, 2023). Across the study years, 2019 was considerably wetter (*Tomahawk*: 433.6 mm; *Evansburg*: 381 mm), and slightly cooler (*Tomahawk*: 12.7 °C; *Evansburg*: 12.6 °C) than normal, 2021 was considerably drier (*Tomahawk*: 214.9 mm; *Evansburg*: 220.1 mm) and warmer (*Tomahawk*: 15.5 °C; *Evansburg*: 14.7 °C) than normal, and 2022 was close to the historical normal for precipitation (*Tomahawk*: 290.2 mm; *Evansburg*: 278.2 mm) and air temperature (*Tomahawk*: 14.6° C; *Evansburg*: 14° C; ACIS, 2023).

Across all the study sites in all three years, average top 20 cm peat VWC ranged from 25.0 to 71.1% in the ditches (when there was no standing water present) and 47.1 to 63.8% at the field positions. There was no significant field position effect on VWC in any of the years (Table 2.1, Table A1). Ditch VWC was generally significantly different than the field position VWC (Table 2.1, Table A1). During 2021 (drier year), the ditches were significantly drier, about 50% and 9%, than the Young and Intermediate Site field positions, respectively. The near normal precipitation levels in 2022 resulted in the Intermediate Site ditches being about 15% wetter than the field positions (Table 2.1, Table A1). While average WTD at the Intermediate Site at the well in a field center varied by almost 50 cm between study years, top 20 cm field VWC varied by less than 10%.

The height of the standing water in the ditches (water level) was measured in all three years (Table 2.1). There was no standing water in the ditches during our 2021 sampling period at the Intermediate Site and our 2022 sampling period at the Young Site. The ditches were filled with water 100% of the time at the Old, Intermediate and Young Site in 2019, and 51% of the time in 2022 at the Intermediate

Site. In 2019, water level was around 15 cm greater on average in the Intermediate Site ditches compared to the other sites, but this was not statistically significant ($F_{2,16}$=2.435, p=0.119; Table 2.1). Average water temperature of the water column in the ditches varied by only 0.5 ºC, and there was no significant difference between sites in 2019 ($F_{2,16}$=0.0390, p=0.962).

Average soil temperature at 10 cm depth varied by only a few degrees across all positions and sites each year (Table 2.1). Soil temperature was generally highest in 2021, likely due to sampling only occurring in July and August. There was no significant effect of position or site, or their interaction, on soil temperature, except for a significant site effect in 2019 (Table 2.1, Table A1). In 2019, soil temperature was significantly lowest at the Young Site, though the effect size was only a couple of degrees (Table 2.1, Table A1).

Table 2.1 Average and median (in brackets) VWC (%), water level (in the ditches; cm) or water table depth (WTD; in the fields; cm), soil temperature at 10 cm depth (°C), and average temperature of the water column in the ditches (°C) at the Young, Intermediate and Old in 2019, 2021 and 2022. There was no standing water in the Young Site ditches in 2022. Variables that were not measured are indicated with n.m. Significant differences between variables within a column are indicated with superscript letters. The superscripts for VWC and water temperature can be compared along the whole column, as the interaction between Position and Site was significant (Table A1, Section 3.1). There was no significant interaction effect for water level, WTD or soil temperature, although soil temperature varied among sites (Table A1). Average soil temperature at the Young Site was significantly lower than at the Intermediate and Old Sites (Table A1).

Measurement Position	VWC 2019	VWC 2021	VWC 2022	Water Level/WTD 2019	Water Level 2022	Water Temp 2019	Water Temp 2022	Soil Temp 2019	Soil Temp 2021	Soil Temp 2022
Young Site										
Ditch	n.m	25.7^{a}	53.7^{ab}	8.5	n.m	18.0	n.m	n.m	14.2	12.0
		(25.3)	(54.7)	(8.1)		(14.6)			(13.6)	(12.1)
2m	50.8^{ab}	50.7^{bc}	51.2^{ab}	n.m	n.m	n.m	n.m	11.7	13.0	11.2
	(50.2)	(50.4)	(51.3)					(11.9)	(12.9)	(11.9)
5m	54.4^{abc}	51.3^{bc}	51.3^{ab}	n.m	n.m	n.m	n.m	12.02	12.2	10.8
	(53.1)	(51.2)	(51.3)					(12.5)	(12.2)	(10.7)
15m	56.17^{a}	51.7^{bc}	49.2^{a}	n.m	n.m	n.m	n.m	12.5	12.3	10.3
	(55.42)	(50.7)	(48.8)					(13.0)	(12.5)	(10.6)
Intermediate Site										
Ditch	n.m	47.1^{b}	71.1^{c}	24.6	13.8	18.0	18.0	n.m	14.4	11.4
		(44.8)	(71.0)	(15.0)	(12.0)	(17.6)	(17.0)		(14.4)	(11.0)
2m	59.2^{cde}	55.7^{c}	53.7^{ab}	62.8	n.m	n.m	n.m	13.3	14.5	10.6
	(59.0)	(55.7)	(53.7)	(71.5)				(14.0)	(14.5)	(10.1)
5m	63.8^{e}	56.0^{c}	56.8^{b}	35.6	n.m	n.m	n.m	13.7	14.0	10.9
	(64.5)	(56.3)	(56.4)	(33.5)				(14.0)	(14.1)	(12.5)
15m	62.0^{de}	56.1^{c}	56.1^{b}	30.8	n.m	n.m	n.m	13.6	13.4	10.1
	(62.3)	(56.0)	(55.5)	(30.6)				(13.6)	(13.3)	(10.1)
Old Site										
Ditch	n.m	n.m	n.m	9.9	n.m	18.5	n.m	n.m	n.m	n.m
				(9.0)		(16.7)				
2m	48.9^{a}	n.m	n.m	n.m	n.m	n.m	n.m	13.3	n.m	n.m
	(47.0)							(14.0)		
5m	59.6^{cde}	n.m	n.m	n.m	n.m	n.m	n.m	13.6	n.m	n.m
	(60.5)							(14.0)		
15m	60.2^{cde}	n.m	n.m	n.m	n.m	n.m	n.m	13.8	n.m	n.m
	(60.3)							(14.4)		

2.4.2 CO_2 and CH_4 Emissions

After quality control, we retained 456, 847 and 998 CO_2 measurements for data analysis in 2019, 2021 and 2022, respectively. We did not make direct comparisons between years due to differences in sampling periods. Mean and median values for each year for each measurement position and site can be found in Table A2. The fields and ditches were all sources of CO_2. We did not observe a field position effect on CO_2 emissions, with no significant difference in emissions between field positions within a site in any of the years (Table 2.2), though there was a pattern of emissions decreasing with increasing distance from the drainage ditches (Figure 2.2, Table 2.2). There were differences between the ditches and fields, however, with ditch emissions in 2019 significantly four to five times higher than the field positions at the Young Site, and double, though not significantly greater, at the Old Site. In contrast, the Intermediate Site ditch CO_2 emissions were less than two thirds of field position emissions, though this difference was only significant at the 2 m and 15 m positions (Figure 2.2). At the Young Site in 2021 and 2022, and the Intermediate Site in 2021, there was a significant doubling of ditch emissions compared to the field positions. While ditch emissions were about double the field position emissions at the Intermediate Site in 2022, this difference was only significant at the 2 m field position.

The effect of duration of extraction on CO_2 emissions was not consistent between years (Table 2.2). In 2019, field position emissions were significantly higher at the Young and Intermediate Site compared to the Old Site, and the Young Site ditches had over five times the emissions of the other sites' ditches. In 2021, mean CO_2 emissions (averaged across the field and ditch positions; Table 2.2) were about double, and significantly higher, at the Intermediate Site compared to the Young Site. In 2022, the Intermediate Site 2 m position had significantly higher emissions than all the other field positions, with no differences between any of the others. The Young site ditches had significantly higher CO_2 emissions than the Intermediate Site ditches.

After quality control, we kept 287, 419 and 984 CH_4 measurements for data analysis from the sites in 2019, 2021 and 2022. There were 23 instances of ebullition from ditches in 2019, and 7 in 2022. We did not make direct comparisons between years due to differences in sampling periods each year. Mean and median values for each year for each measurement position and site can be found in Table A2. Similar to CO_2, we did not observe a significant difference in CH_4 emissions between field positions within a site in each year. Ditch emissions were significantly higher than the field position emissions at all three sites in 2019, and at the Intermediate Site in 2021 and 2022, with this difference

ranging from over 80 times to more than 1000 times the field position emissions. We observed no difference in CH_4 emissions between the peat filled ditches and the field positions in 2021 and 2022 at the Young Site. When comparing between sites, we did not see an effect of extraction duration on field CH_4 emissions, with no difference in field position emissions among any of the sites in any year. For the ditches, we did observe that the Young Site ditch emissions in 2019 were significantly (over eight times) higher, than the Intermediate and Old Site ditch emissions. The dug-out Intermediate Site ditches had significantly higher emissions than the peat filled Young Site ditches in 2021 and 2022 (Figure 2.2).

Table 2.2 Results from linear mixed effects models showing the effects of site and position on CO_2 and CH_4 fluxes from measurement positions in 2019, 2021 and 2022. Collar was used as a random factor, and a variance structure with measurement position was added to account for the heterogeneity of variance. The comma under the degrees of freedom (DF) heading was used to separate the degrees of freedom in the numerator and the denominator.

		CO_2			CH_4		
Year	Effect	DF	F-Value	p-value	DF	F-Value	p-value
2019							
	Site	2, 78	50.59	<0.0001	2, 65	0.36	0.6991
	Position	3, 78	1.84	0.1471	2, 65	40.74	<0.0001
	Site: Position	6, 78	5.40	0.0001	4, 65	7.83	<0.0001
2021							
	Site	1, 91	42.52	<0.0001	1, 58	0.45	0.5055
	Position	3, 91	17.50	<0.0001	2, 58	10.70	<0.0001
	Site: Position	3, 91	2.21	0.0925	2, 58	6.07	0.0012
2022							
	Site	1, 56	6.31	0.0149	1, 56	18.53	0.0001
	Position	3, 56	15.58	<0.0001	2, 56	34.90	<0.0001
	Site: Position	3, 56	4.28	0.0086	2, 56	37.05	<0.0001

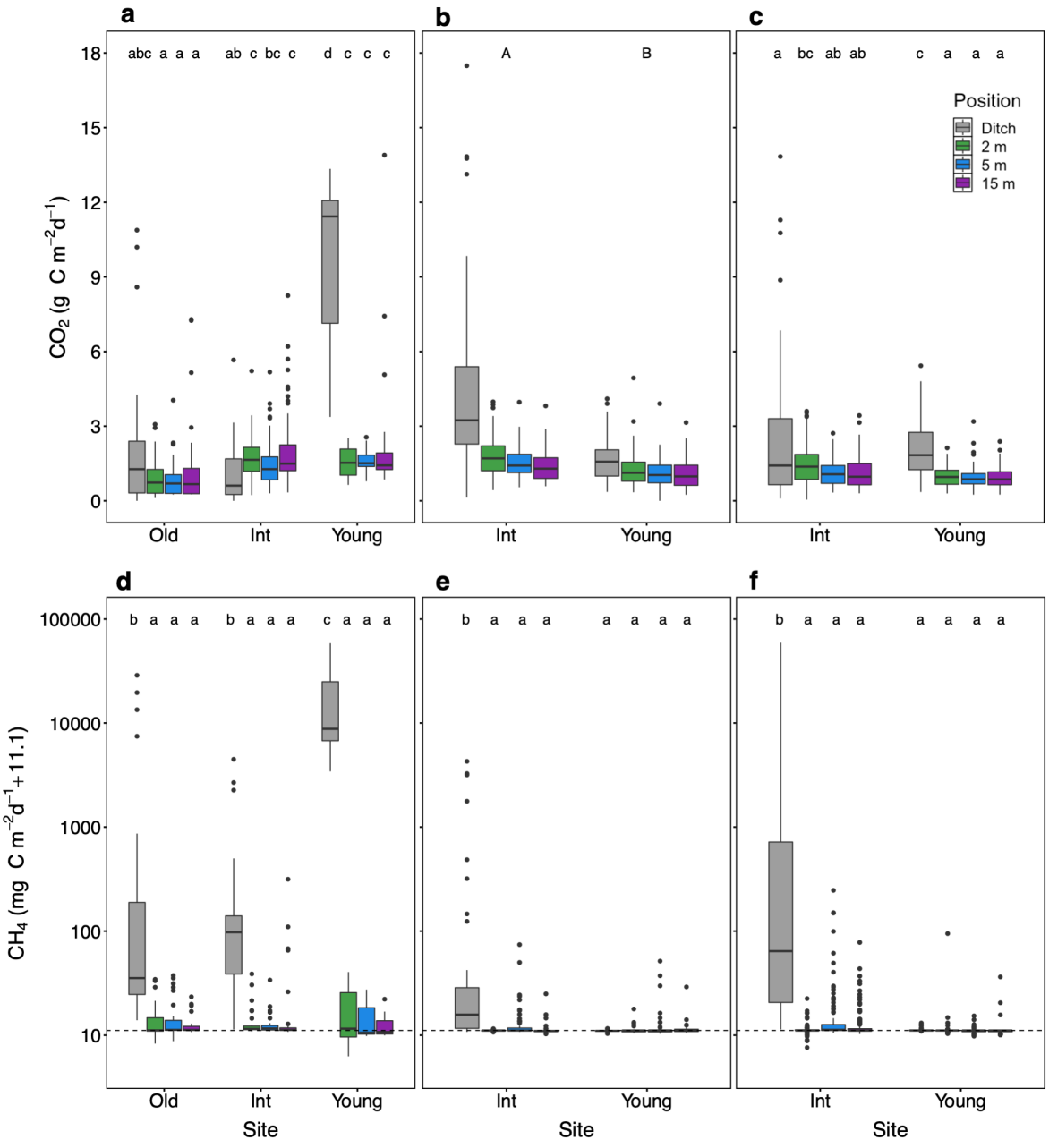


Figure 2.2 Boxplots showing CO_2 (top) and CH_4 (bottom) emissions at each site and measurement position in 2019 (a, d), 2021 (b, e) and 2022 (c, f). Boxplots that share letters are not significantly different from each other (Table 2.2). For CH_4, the vertical scale has been offset to allow for presentation of results on a log scale, i.e., zero flux is plotted as 11.1 (see horizontal line on plots).

2.4.3 Carbon Emissions and Ditch Water Level

The presence of standing water in the ditches led to reduced CO_2 and CH_4 emissions (Figure 2.3). In 2022, there was standing water for 50.1% of ditch measurements at the Intermediate Site. Measurement position (where positions were defined as: average of 2, 5 and 15 m field positions, ditches with standing water, and ditches without standing water) had a significant effect on both CO_2 ($F_{2,447}$= 55.9513, $p<0.0001$) and CH_4 emissions ($F_{2,437}$= 87.3791, $p<0.0001$). Ditch CO_2 emissions when there was no water (2.93 g C m^{-2} d^{-1}) were significantly higher than from the field positions (1.22 g C m^{-2} d^{-1}), and from the ditches with standing water (0.93 g C m^{-2} d^{-1}). Methane emissions were highest from the ditches with no standing water (2698 mg C m^{-2} d^{-1}) compared to ditches with water (762 mg C m^{-2} d^{-1}) and the field positions (3.13 mg C m^{-2} d^{-1}; Figure 2.3).

In general, the depth of standing water in the ditches was not an effective predictor for CO_2 or CH_4 emissions. At the Old Site in 2019, there were significant positive relationships between depth of standing water and CO_2 ($F_{1,28}$=42.53, $p<0.0001$) and CH_4 ($F_{1,29}$=40.51, $p<0.0001$) emissions, but they appear to be driven by a few very high emissions (Figure A1). The relationship was not significant at the Intermediate or Young Site in either 2019 or 2022, while no standing water was present in 2021 (Table A3).

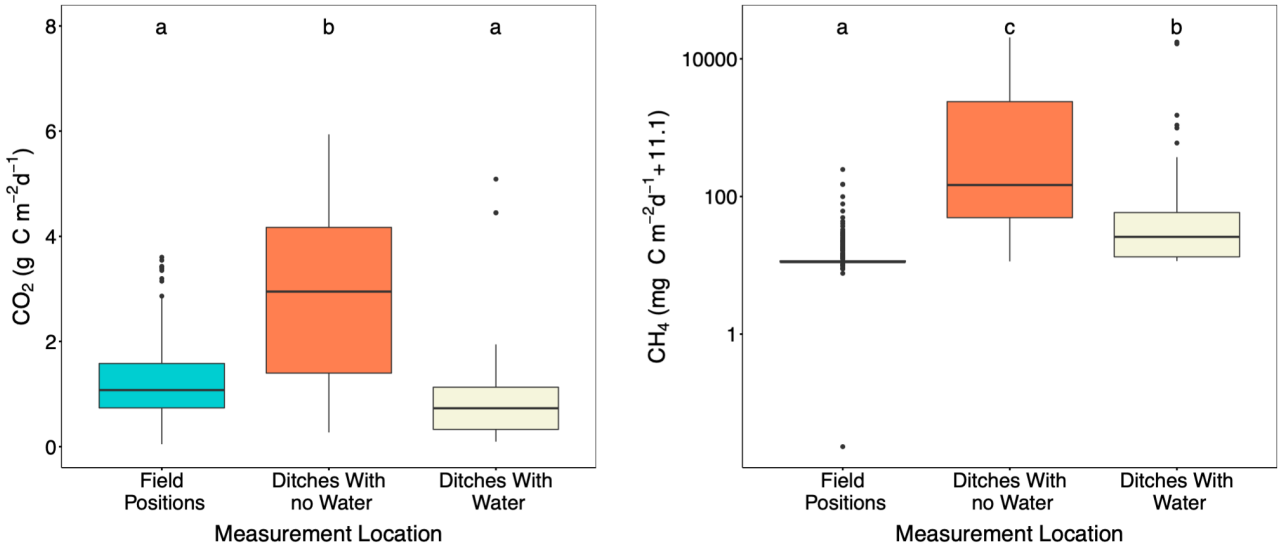


Figure 2.3 Boxplots of CO_2 (left) and CH_4 (right) fluxes from the Intermediate Site in 2022. The field positions boxplot is an average of emissions from the 2 m, 5 m and 15 m positions. For CH_4, the vertical scale has been offset to allow for presentation of results on a log scale, i.e., zero flux is plotted as 11.1.

2.4.4 Carbon Emissions and Volumetric Water Content

Across the three study years, higher VWC was significantly associated with increased CO_2 emissions, both individually ($F_{1, 1637}$=7.5742, p=0.006), and in its interaction with Position ($F_{3, 1637}$=2.662, p=0.0466) and Site ($F_{2, 1637}$=14.215, p<0.0001). There were only significant positive effects of VWC on CO_2 emissions at the 2 m and 15 m positions at the Intermediate Site, and a significant negative effect at the Intermediate Site ditch position (Table 2.4, Figure A2). The total variation explained by VWC was small at only 3–8%. Higher VWC was not significantly associated with CH_4 emissions individually ($F_{1, 1234}$=0.699, p=0.403) or in its interaction with Position ($F_{3, 1234}$=2.367, p=0.069), but was in its interaction with Site ($F_{2, 1234}$=6.752, p=0.0012). There was a significant positive relationship at the Intermediate Site, though it explained less than 1% of the variation in CH_4 emissions (Table A5, Figure A3).

2.4.5 Carbon Emissions and Soil Temperature

Across the three study years, higher soil temperatures at 10 cm depth were not significantly associated with increased CO_2 emissions individually ($F_{1, 2040}$=2.272, p=0.1319) or in its interaction with Position ($F_{3, 2040}$=1.357, p=0.2543), but were in their interaction with Site ($F_{2, 2040}$=3.685, p=0.0253) when running an ANOVA on a regression with ln(CO_2). The regressions at each Site were significantly different between the field positions and the ditches. Based on this, we separated the data into field and ditch data. We found positive exponential relationships between soil temperature and CO_2 field fluxes at the Young, Intermediate and Old Sites (Figure 2.4). This relationship was strongest at the Intermediate Site and weakest at the Old Site. Results of the fit of the observed data versus the predicted modeled data are presented in Table A6 and Figure A4. The corresponding Q10 values were 1.99, 1.91 and 2.86 at the Young, Intermediate and Old Sites respectively. There were significant exponential relationships between soil temperature and ditch CO_2 emissions at all three sites but the modeled predicted values explained less than 5% of the variation in observed values (Figure A5).

When using a similar exponential model for CH_4 emissions, the regressions of observed versus predicted values were not significant at any of the Sites (Table A6). Soil temperature at 10 cm depth was also not significantly linearly associated with CH_4 emissions individually ($F_{1, 1481}$=0.0881, p=0.767), or in its interaction with Position ($F_{3, 1481}$=0.638) or Site ($F_{2, 1481}$=0.3075, p=0.735). The ditch and field data, and data from each site, were plotted separately for better visualization (Figure 2.4, Figure A5).

Water column temperature in the ditches was not significantly linearly associated with CO_2 ($F_{1, 90}$=0.0329, p=0.857) or CH_4 ($F_{1, 87}$=0.1405, p=0.709) emissions individually, but was in its interaction with Site (*CO_2*: $F_{2, 90}$=4.679, p=0.0117; *CH_4*: $F_{2, 87}$=48.641, p=<0.0001). There were significant positive relationships only at the Young Site, though they explained less than 5% of the variance.

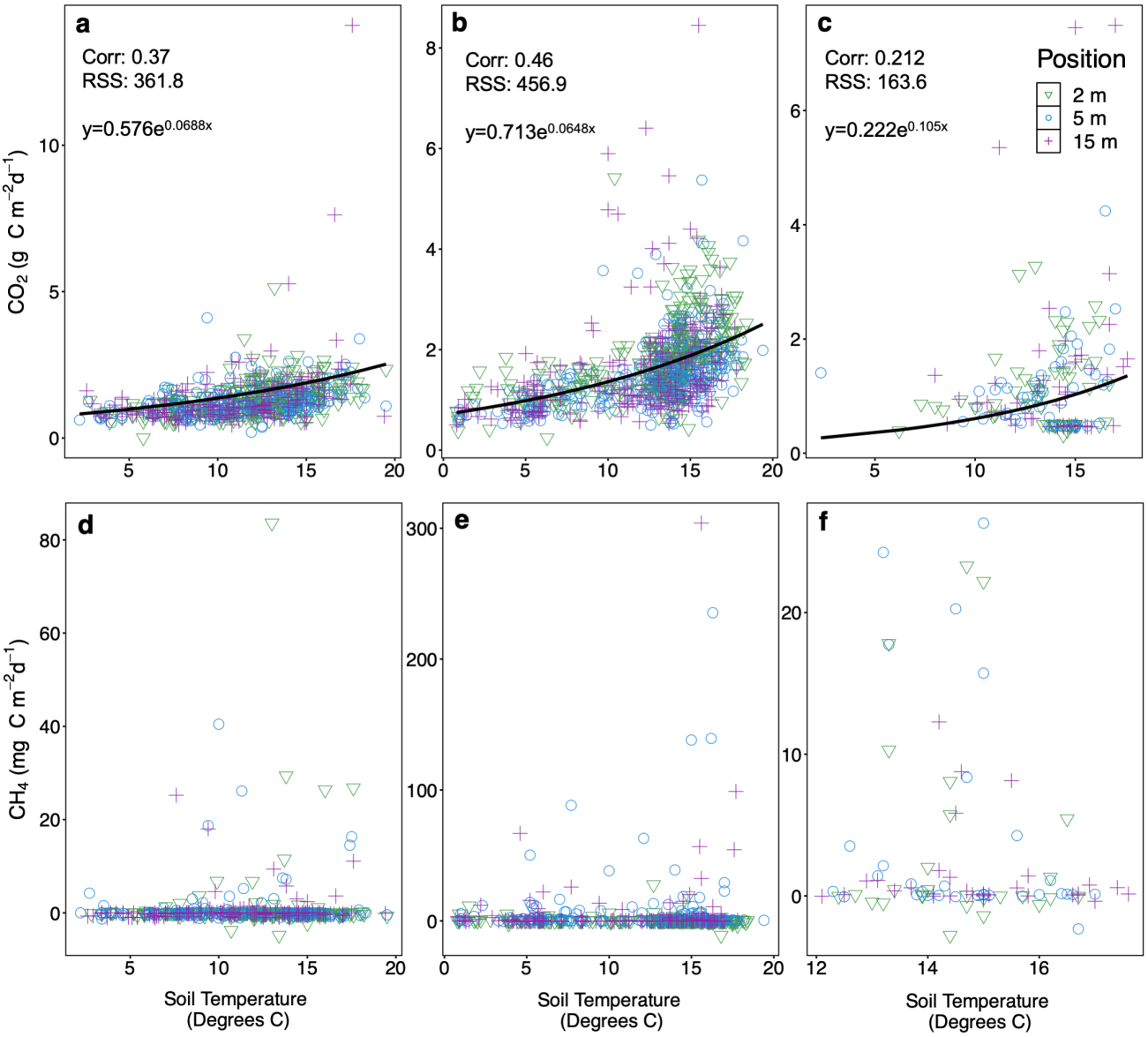


Figure 2.4 Scatterplots showing the effect of soil temperature at 10 cm depth on CO_2 and CH_4 emissions at the 2 m (green triangle), 5 m (blue circle) and 15 m (purple cross) positions at the Young (a, d), Intermediate (b, e) and Old (c, f) Sites over the three study years. The correlation (corr) and residual sum of squares (RSS) are shown for the significant exponential relationships. The CH_4 – temperature relationships were not significant and thus were not shown on the plots.

2.4.6 Carbon Gas Production from Fields Versus Ditches

Following quality control, we kept 98% of CO_2 incubation fluxes (545 fluxes). Methane production was negligible, which was expected given that the peat was not saturated. There was no significant difference in CO_2 production between the two transect sampling locations at the Intermediate Site ($F_{1,31}$=00771, p=0.7831), so the data was combined into Intermediate ditches and Intermediate fields for the rest of the analysis (Figure 2.5). Sampling location ($F_{4,541}$=237.80, p<0.0001) and VWC treatment ($F_{2,541}$=50.61, p<0.0001) influenced CO_2 production from the jars, with no significant interaction between the effects ($F_{8,541}$=1.31, p=0.2346). Production was highest from the Intermediate Site ditch location (0.337 mg C g^{-1} d^{-1}) and lowest from the Young Site ditch bottom location (0.0875 mg C g^{-1} d^{-1}). There was no difference between the Young Site ditch top (0.149 mg C g^{-1} d^{-1}) and field (0.154 mg C g^{-1} d^{-1}) locations, and the Intermediate Site field location (0.157 mg C g^{-1} d^{-1}). Production significantly increased with increasing VWC (Figure 2.5). These patterns of higher production in the ditches, and a positive relationship with VWC generally mirrored patterns observed with the field fluxes.

The Intermediate Site ditches (1.031) had slightly higher humification indices than the field centers (0.866), and they were both lower than those from the Young Site field center (1.090), ditch top (1.201) and ditch bottom (1.176) locations. There was no evidence for an effect of peat humification on CO_2 production from the incubation experiment (Figure A6).

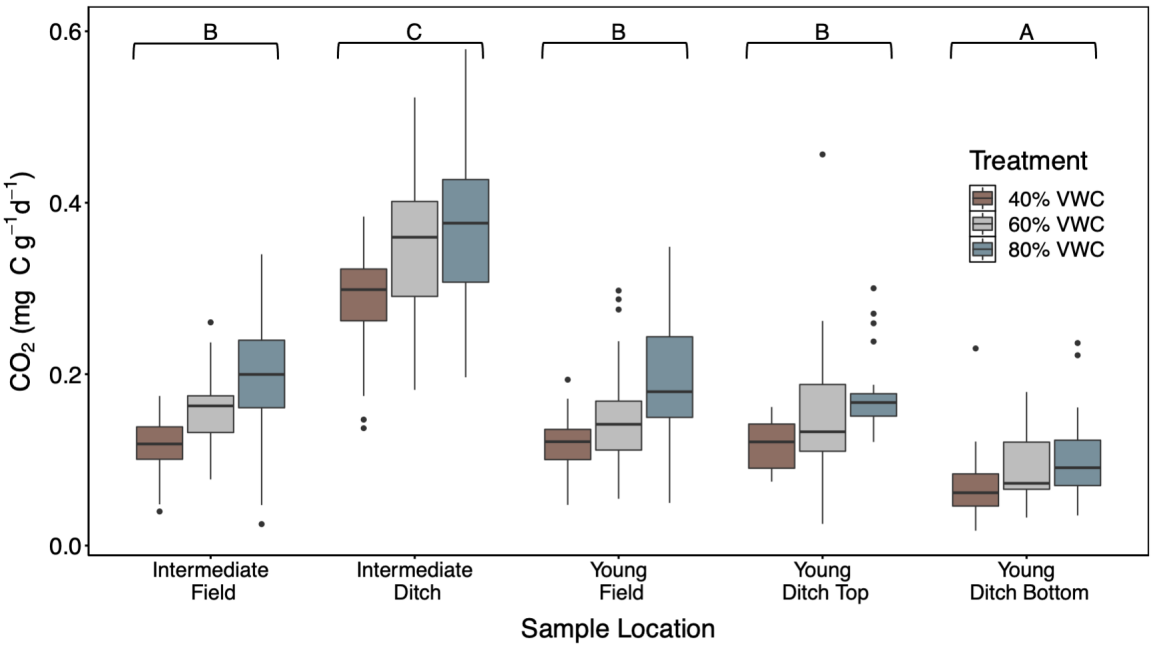


Figure 2.5 CO_2 production at each VWC treatment level from each sampling location. Boxplots that share letters are not significantly different from each other.

2.4.7 Peat Substrate Quality

The Young Site, while having been extracted for a shorter length of time, had surface peat about double the age of the Intermediate Site (Table 2.3, Figure 2.1). Peat from the Young Site had C:N ratio over four times higher than the Intermediate Site, though the surface peat was slightly less humified (Table 2.3). Peat ^{14}C age and humification indices generally increased, and C:N ratio decreased, with depth at both sites. The Intermediate Site 2 m and 15 m position ^{14}C ages of the surface peat were similar, and about half the age of the ditch surface. Humification indices and C:N ratios were also similar, with slightly more humified peat in the ditches. The ditch surface peat was about half the age of the peat at a similar depth below the 2 m position (Table 2.3). At the Young Site, the field surface peat was about half the age of the surface of the partially filled in ditches, was slightly less humified, and had double the C:N ratio. The peat at the original ditch surface was a similar age and had similar humification index and C:N ratio of the peat at a similar depth below the 2 m position.

Table 2.3 ^{14}C age (yr BP), C:N ratio and humification index of peat samples at the Intermediate and Young Site measurement positions. The depth below field surface (m) is the depth we sampled at below the field surface. The depth below ditch (m) is the depth below the ditch surface. Refer to Figure 2.1 for schematic of sampling locations.

Position	Depth Below Field Surface	Depth Below Ditch Surface	^{14}C Age	C:N Ratio	Humification Index
Intermediate Site					
2 m					
	0.025		335	30	0.871
	0.8		2101	27	1.201
	2.05		3865	14.6	1.191
	3.25		3923	10	
15 m					
	0.025		333	31.8	0.912
Ditch					
	0.9	0.05	858	26	1.072
	1.35	0.5	1349	25.7	1.299
Young Site					
2 m					
	0.025		625	143	0.777
	0.8		2205	55	0.973
	1.85		3042	78	1.475
Ditch					
	0.15	0.05	1178	63	0.954
	0.87	0.77	2395	66	1.008

2.5 Discussion

2.5.1 CO_2 and CH_4 Field Emissions

The actively extracted fields were all sources of CO_2, as the removal of vegetation prior to extraction eliminates the photosynthetic CO_2 uptake by vegetation. When considering the whole extraction process, the decomposition of the extracted peat will account for the majority of the C emissions (Cleary et al., 2005). However, emissions related to the land use conversion of the actively extracted peatlands accounted for an estimated 15% of total emissions, making it still an important topic of study (Cleary et al., 2005). Growing season field emissions were on the high end of previous measurements from extracted peatlands in Canada (Greenwood, 2005; Clark et al., 2023) and internationally (Alm et al., 2007; Salm et al., 2012; Shurpali et al., 2008; Wilson et al., 2015), though on the lower end of others (Sundh et al., 2000). The fields ranged from small sources to small sinks of CH_4, in line with findings from other actively extracted peatlands (Clark et al., 2023; Salm et al., 2012; Waddington et al., 2009). Work in both newly drained peatlands, and peatlands undergoing long-term drainage, has shown a general increase in ecosystem respiration, and a reduction in CH_4 emissions. The mechanisms controlling CO_2 and CH_4 flux in drained peatlands generally depend on vegetation, microtopography, hydrologic dynamics and the relative partitioning of respiration into autotrophic and heterotrophic components (Ballantyne et al., 2014; Bubier et al., 2003; Chivers et al., 2009; Strack et al., 2006). The deeper water table following drainage increases oxygen availability in more of the peat profile (Pezeshki and DeLaune, 2012), limiting anaerobic CH_4 production and increasing both the proportion of CH_4 that is oxidized, and the aerobic production of CO_2 (Lai, 2009).

Of particular interest is comparing our work to Clark et al. (2023), who followed a similar sampling design in Rivière-du-Loup, Quebec, and is the only other multi-year study of actively extracted peatlands in Canada. Their work at one bog complex found that the annual removal of peat led to decreasing CO_2 emissions in the first few years following the start of extraction due to a decrease in substrate quality of the newly exposed deeper peat. After three years of extraction, the continual removal of peat did not further reduce CO_2 emissions. They found no effect of duration of extraction on CH_4 emissions. Clark's work therefore suggests that within one bog complex, a single emission factor can be used for CH_4 and for CO_2, one emission factor would apply for the initial years, and a second lower value for the remaining years of the extraction. While our CH_4 findings support this framework, our research suggests that care needs to be taken when using extraction duration to

extrapolate CO_2 emission factors from one peatland complex to another with different substrate qualities. Our newly extracted site (Young Site) had significantly lower CO_2 emissions than our site that had been extracted for ~15 years (Intermediate Site). We suggest that C:N ratio could be a relatively inexpensive and easy metric to group extracted peatlands into emission factor categories. Using C:N ranges reported by Meng et al. (2015), our Intermediate Site peat had more fen-like C:N ratios compared to our more bog-like Young Site (31 ± 3 vs. 42 ± 7; Kendall, 2021), and correspondingly significantly higher CO_2 emissions. Our more bog-like Young and Old Sites (Kendall, 2021) fit into the two-emission factor model proposed by Clark el al. (2023). More research in extracted peatlands with lower C:N ratios will be needed to determine emission factor values for those sites.

2.5.2 Environmental Controls of CO_2 and CH_4 Field Emissions

A positive relationship between soil temperature and CO_2 and CH_4 emissions, due to stimulation of microbes in warmer conditions, is well-established in natural and drained peatlands (Lai et al., 2014; Ballantyne et al., 2014). We found a positive exponential relationship with CO_2 emissions and our Q_{10} values fell within the range of 1 to 5 reported for natural peatlands (Ballantyne et al., 2014; Blodau et al., 2004; Bubier et al., 2003; Chen et al., 2018), and for potential CO_2 production rates from extracted peatlands (Waddington et al., 2001). We did not find a relationship between soil temperature and CH_4 emissions, though the deep water tables and associated increased CH_4 oxidation (Turetsky et al. 2014) could complicate this relationship (Bansal et al., 2018). Our larger sample size compared to previous studies could explain why we detected a significant relationship while Sundh et al. (2000) and Clark et al. (2023) found limited to no effect of soil temperature on CO_2 emissions at actively extracted sites. Understanding the soil temperature-C relationship will be important given that extracted peatlands can have double the temperature variability of natural sites (Waddington et al. 2001). This variability could lead to increased heterotrophic CO_2 emissions than from natural sites with a steadier temperature regime (Clymo, 1984). Our findings suggests that despite these sites being comprised of catotelmic, non-vegetated peat, they exhibit similar CO_2-temperature relationships to natural peatlands. It also appears that this relationship does not depend on the extraction duration. Research at these sites could help further our knowledge of heterotrophic respiration from natural peatlands, where it is difficult to partition autotrophic and heterotrophic respiration (Rankin et al., 2022).

We found that top 20 cm VWC, a proxy for pore oxygen concentration, was not a strong predictor of chamber CH_4 or CO_2 fluxes from the fields or ditches, with the few significant positive relationships explaining less than 10% of the C emission variability. Our incubation experiment showed that CO_2 flux increased with increasing VWC treatment level (40%, 60% and 80%). Waddington et al. (2001) found similar results in block cut peat, with optimum CO_2 production at 70% VWC. The peat fields contain drained, aerated peat, with a rarely saturated surface. In forest ecosystems with aerated soils, studies have observed a similar positive correlation between CO_2 and VWC that we observed (e.g., Cooperdock et al., 2020). The small range of VWCs (~25%) observed in the field could explain the limited explanatory power of VWC for CO_2 and CH_4 emissions, especially compared to the incubation experiment (40% range). The possible presence of hysteresis between C emissions and VWC, which our study could not adequately test, could also explain the observed weak relationship. Our results, during both very wet and very dry summers, along with the lack of relationships found in other extracted peatlands (Sundh et al., 2000; Clark et al., 2023) suggests that under the conditions observed at these sites, there is no C emission advantage to limiting the degree of drainage. The few instances of a positive relationship with VWC suggest that doing so could actually increase both CO_2 and CH_4 emissions. This finding contradicts with a recent synthesis study showing that decreasing the water table depth in managed peatlands would significantly reduce CO_2 emissions (Evans et al., 2021). However, most of their sites had some extent of vegetation and had water tables within the top 50 cm of the peat profile. Additional research at the Intermediate Site (personal observations, not published) suggests that the water table in 2021 (dry year) was deeper than 65 cm for most of the study period. More research, with a specific focus on soil respiration, is needed to explain C dynamics in deeply drained peatlands.

2.5.3 CO_2 and CH_4 Ditch Emissions

The drainage ditches were all sources of CO_2, in line with previous studies from extracted peatlands (Clark et al., 2023; Sundh et al., 2000; Vanags-Duka et al., 2022), and other anthropogenically created peatland ditches (Peacock et al., 2021a). The presence of ditch vegetation likely explains Sundh et al.'s (2000) observed net uptake of CO_2 in some study plots. The ditches were very large sources of CH_4, except at the Young Site in 2021 and 2022 (discussed below). Our mean values for steady CH_4 emissions (Young: 473 mg C m^{-2} d^{-1}; Intermediate Site: 1353; Old Site: 1121 mg C m^{-2} d^{-1}) are considerably higher than those reported from other extracted sites, with average CH_4 emissions of 25.68 mg C m^{-2} d^{-1} (Quebec, Canada: Clark et al., 2023) and 72 mg C m^{-2} d^{-1} (Latvia: Vanags-

Duka et al., 2022). As mentioned above, site specific conditions might have caused these differences, though without more supporting data on the hydrologic conditions in those studies, it is difficult to comment on the exact causes. We found that the ditch hydrology was an important control on C emissions. At the Intermediate Site, the presence of standing water in the ditches, though not the quantity of water in the ditches, led to significantly reduced CO_2 emissions, likely due to preferential production of CO_2 under aerobic conditions (Pezeshki and DeLaune, 2012). For CH_4, the effect depended on the length of time the ditches had standing water. In 2022 at the Intermediate Site, when the ditch water level was variable, the presence of standing water reduced ditch CH_4 emissions by over 300%. The drainage ditches at this site have high dissolved oxygen concentrations compared to nearby streams (personal observation, data not shown), which would support increased oxidation of CH_4 as it diffuses through the water column (Lai, 2009; Strack et al., 2004). However, the absence of standing water in the ditches for the whole 2021 period at this site resulted in less than half the average emissions compared to 2022. These findings can help guide industry on drainage ditch water level management. Precipitation duration and frequency are a primary control on ditch water levels, making it unlikely for the ditches to remain without water for a whole extraction season. Our results suggest that maintaining some water in the drainage ditches would actually help reduce CO_2 and CH_4 emissions during rainy years when the ditch water level fluctuates.

2.5.4 Spatial Distribution of CO_2 and CH_4 Emissions

We did not observe an effect of field position (distance from the drainage ditch) on CO_2 and CH_4 emissions on our non-contoured fields. In eastern Canada, fields are often contoured to create a domed shape to aid drainage from the centre of the field to the ditches. Clark et al. (2023) found no position effect at contoured fields, except in the first few years following the start of the extraction. In this case, the age gradient on the contoured fields, with younger peat near the centre of the field, drove CO_2 but not CH_4 emission differences. Our work, conducted during both wet and dry summers, highlights how the 30 m ditch spacing did not create substantial enough environmental gradients across the fields to affect C emissions. Despite evidence for drainage ditch-induced water table gradients, VWC in the surface 20 cm varied by less than 10% across the fields. Drainage-induced subsidence and the associated decreased specific yield likely contributed to the maintenance of high surface VWC even during periods of deep water tables (Price, 1997; Price and Ketcheson, 2009). Top 10 cm soil temperature only varied by a few degrees, and we observed similar ^{14}C ages, C:N ratio and humification indices of the surface peat at the 2 m and 15 m positions at the Intermediate Site. We did

not conduct substrate quality measurements at the Old and Young Site, but assume similar results given that all the fields were levelled by the peat producers. Thus, these similar substrate and hydrological conditions across the field likely explain the lack of variation in C emissions.

We found that ditch CO_2 and CH_4 emissions were about double and ten times those from the fields, respectively, though this varied with the site and year. This is consistent with research from extracted sites in Quebec, where about double, and seven-times higher CO_2 and CH_4 emissions, respectively were reported from the ditches compared to the fields (Clark et al., 2023). Work by Sundh et al. (2000) in Sweden found greater CO_2 emissions from fields compared to ditches, though as mentioned above, this was likely due to the plant photosynthetic uptake in their ditches. These findings demonstrate the necessity of future studies in actively extracted peatlands to include drainage ditches in their sampling plans to accurately determine sector wide C emissions. See section 4.5 for a further discussion of this.

The water level in the drainage ditches helped explain patterns in CH_4 and CO_2 emissions between the ditches and fields. Ditch CH_4 emissions at the Intermediate and Old Site were always significantly higher than from the fields, regardless of the presence or absence of standing water. The ditches had standing water in them for all of 2019, and about half of the 2022 measurement periods. Ditch VWC (when there was no standing water) was higher than the field VWC in 2022 and slightly lower in 2021. Even when the ditch surfaces were drier than the fields, we visually observed that the water tables were closer to the surface than in the fields, which likely explains the higher ditch CH_4 emissions. Studies in natural and drained peatlands have found negative relationship between CH_4 emissions and water table depth, due to the increased depth of the anoxic layer (Abdalla et al., 2016). For CO_2, we found that the presence or absence of standing water in the drainage ditches explained patterns in emissions. At the Intermediate Site, we observed reduced, but not significantly different, CO_2 emissions from the ditches during periods when they had standing water compared to the fields in 2022. When the ditches were not saturated, we observed significantly higher CO_2 emissions from the ditches than from the fields. This pattern was consistent with results from 2019 and 2021 at the Intermediate and Old Sites.

The elevated ditch CO_2 emissions under non-saturated conditions compared to the fields was a surprising result given that the ditch surface peat is about 1 m deeper than the field surface, and thus is comprised of older, more humified peat. While the observed positive relationship between VWC

and CO_2 could be driving higher emissions in the ditches, our incubation experiment showed that the ditches had higher CO_2 emissions than the fields at all VWC treatment levels (40%, 60% and 80%). This suggests in situ peat differences affecting CO_2 production. Substrate quality, an important control on decomposition in peatlands (Wang et al., 2015), does not explain these differences either. The ditch surface peat had similar C:N ratios to the fields, and while it was slightly more humified, there was no significant relationship between humification indices and CO_2 emissions in our incubation experiment (Supporting Information Figure A6).

We hypothesize a few potential biogeochemical mechanisms for the higher ditch CO_2 emissions than the fields under non-saturated conditions. First, the pore water transport of dissolved labile C from the fields to the ditches could be providing a fresh substrate for decomposition. Our ^{14}C data provides evidence for the input of young C from the fields to the ditches, with the ditch surface peat being only half as old as we would expect given its depth (856 versus 2101 years). Some of this younger C could also be coming from the mixing of field and ditch peat during harvesting and ditch maintenance. We also observed growth of small shrubs and mosses along the edges of the ditches, which could be providing fresh organic matter inputs. Second, the ditches had a more varied hydrologic regime than the fields, with surface peat experiencing both oxic and anoxic conditions. Several studies have observed increased decomposition during periods of water table fluctuations (Blodau and Moore, 2003; Kim et al., 2021). Finally, we found that under periods of non-saturation, the ditches had significantly greater CH_4 emissions than the fields (discussed further below). This could allow for higher rates of CH_4 oxidation to CO_2 by methanotrophs.

Increasing the width of the fields, and thereby decreasing the number of interior ditches, could be a viable management strategy to reduce C emissions from these sites. Our results from the Young Site in 2021 and 2022 demonstrate that when the ditches are partially to fully filled in with peat, they have higher CO_2 emissions, but similar CH_4 emissions to the fields. Our incubation experiment found no difference in CO_2 emissions between these locations. More work is needed to fully understand the mechanisms driving ditch C cycling, as our substrate quality and hydrologic measurements do not fully explain the difference in chamber-based CO_2 emissions. Nonetheless, the significant reduction in CH_4 emissions in peat-filled ditches could provide a management strategy to reduce CH_4 emissions during active extraction, provided drainage effectiveness needed to dry surface layers of the fields for vacuum-harvesting can be maintained. Given the large global warming potential of CH_4, any small increase in CO_2 emissions would be offset by the much larger reduction in CH_4 emissions.

2.5.5 Comparison to Natural Peatlands

We found that the actively extracted sites were emitting more CO_2 and comparable CH_4 to adjacent natural sites. Natural boreal peatlands are long term net sinks of CO_2, with uptakes ranging from 0.2 to 11.2 g C m^{-2} d^{-1} (Webster et al., 2018). Mean growing season CH_4 emissions range from 29.9 to 72.7 mg C m^{-2} d^{-1} (Turetsky et al., 2014; Webster et al., 2018). To determine a site-wide emission value, we used minimum and maximum field to ditch ratios of 30:1 and 100:1, which assumed a field width of 30 m, and ditch widths ranging from 0.3 to 1 m to account for their trapezoidal shape. The Intermediate Site had average site integrated CH_4 and CO_2 emissions of 18.4 to 52.9 mg C m^{-2} d^{-1} and 1.47 to 1.50 g C m^{-2} d^{-1}, respectively for the May – August period. Studies report CH_4 emissions and understory and tree net ecosystem exchange from the adjacent natural site of 6.3–10 mg C m^{-2} d^{-1} (Bieniada and Strack, 2021; Murray et al., 2017) and 0.27 g C m^{-2} d^{-1}, respectively (Murray et al., 2017). While interannual variability could be driving some of the differences, our CO_2 and CH_4 values are two to five times higher than those of the nearby natural site. This demonstrates that actively extracted sites are not negligible sources of CH_4, and that site estimates that do not consider the role of drainage ditches will underestimate emissions. The ditches contributed 83–94 % of the total CH_4 flux, while occupying only 2–6 % of the total area. This spatial partitioning of CH_4 is similar to what was observed at an actively extracted peatland in Quebec (Clark et al., 2023), as well as at a rewetted temperate fen (Köhn et al., 2021). This demonstrates that these sites are substantial sources of CH_4, and excluding drainage ditches underestimates total emissions (Clark et al., 2023; Peacock et al., 2017; Peacock et al., 2021a). Our findings disagree with the conceptual model presented by Waddington and Price (2009), which suggested that peatlands under extraction would have reduced CH_4 emissions compared to natural sites. Their model did not consider the contributions of drainage ditches.

2.6 Conclusion

Our work adds to the limited body of research on CO_2 and CH_4 emissions from peatlands under active extraction. We confirmed that there was limited spatial variation of emissions across the fields, with emissions not varying with distance from the drainage ditches on the non-contoured fields. This finding was consistent across sites with different extraction durations and interannual weather variability. The drainage ditches were hotspots of CO_2 and CH_4, emitting a disproportionate amount

given their surface area relative to the fields. We found moderate to weak effects of soil and water temperature, VWC, and ditch water level on CO_2 and CH_4 emissions. The strength and direction of these controls did not change with the duration of extraction. The results from this study suggest the need for more research on the connection between hydrology and C emissions at actively extracted sites.

Peat extraction companies should consider their ditch maintenance practices in the context of our results. The companies drain the fields to allow for harvesting machinery to drive across the sites, and to reduce costs associated with transporting the drier harvested peat. We found that the presence of standing water in the ditches led to a significant reduction in CO_2 and CH_4 emissions. Additionally, when the ditches were partially to fully filled in with peat, ditch CH_4 emissions were significantly reduced, becoming similar to the field emissions. Permanently increasing the ditch spacing at sites such as the Young Site, which was able to be harvested in 2021 and 2022 despite partial to complete infilling of ditches with loose peat, could help reduce greenhouse gas emissions associated with extraction activities. Our results can support modeling efforts looking at how increasing the ditch spacing will affect CO_2 and CH_4 emissions.

These results will contribute to emissions factor calculations for the peat industry in Canada. Clark et al. (2023) suggested having two emission factors, one for sites that are within five years of the start of extraction, and one for sites with extraction durations beyond five years. This was based on the significant effect of the duration of extraction on CO_2 emissions during those first five years, when monitoring sites of multiple extraction durations at one bog complex. Our results demonstrate that this simple relationship may not universally hold and variation in peat quality among peatlands should also be considered when determining emissions factors. We suggest that C:N ratio of peat may be a useful indicator for grouping peatlands into similar emissions categories.

Chapter 3

Ecosystem-scale carbon gas emissions from actively extracted peatlands: Moisture limitations are important for interannual variability

3.1 Abstract

Peat extraction substantially alters a peatland's pore structure and hydrology, which in turn affects the surface-atmosphere exchange of carbon (C). Peatland sites are drained, their vegetation is removed, and then the peat is vacuum harvested for use as a horticultural growing medium. Despite this disturbance covering only a small percentage of Canadian peatlands, the shift from being a net sink to a net source of C during the 15–40 plus years of active extraction makes it an important system to study. Ours is the first study in Canada to conduct ecosystem scale measurements from actively extracted peatlands to understand environmental drivers of seasonal and interannual patterns of carbon dioxide (CO_2) and methane (CH_4) emissions. Daytime ecosystem scale measurements of CO_2 and CH_4, using the eddy covariance technique, along with average hourly water table depth (WTD) and soil temperature, were conducted from March to October in 2020, 2021 and 2022 at a Western Site (near Drayton Valley, Alberta), and from May to October in 2020 and 2022 at an Eastern Site (near Rivière-du-Loup, Quebec). Soil temperature, while a poor predictor of CO_2 emissions on its own, had a significant interaction with WTD. We observed positive relationships during moderate and wet periods (WTD < 50 cm) and weakly positive to negative relationships during dry periods (WTD > 50 cm) with lower explanatory power. Process-based models using soil temperature alone may overestimate emissions during dry periods. Understanding freeze-thaw dynamics could be important for creating C budgets at these sites, given that April CO_2 emissions were comparable to June emissions at the Western Site in one year. At each site, the highest CO_2 emissions were associated with the wettest periods, in contrast to the positive CO_2–WTD relationship observed in many studies. This suggests that in deeply drained peatlands, we need to consider periods when insufficient surface moisture conditions can affect microbial decomposition. The sites were small sources of CH_4 compared to natural boreal bogs, though we were not able to capture freeze-thaw periods. After making assumptions for missing nighttime and wintertime data, we estimated an annual C source of 112 to 178 g C m^{-2} yr^{-1}, which is considerably lower than Canada's current Tier 2

emission factor. This research will aid in updating emission factors for peat extraction in Canada, and will help guide industry practices on site management.

3.2 Introduction

Peatlands are an important ecosystem in Canada, covering around 12% of the land area (Hugelius et al., 2020); Tarnocai et al., 2011). The accumulation of peat under waterlogged conditions results in their long-term storage of C (Yu et al., 2010). Peatland disturbance will alter their C balance, and in some cases, result in the system shifting from a net sink to a net source of C (Nugent et al., 2019). Water table drawdown in peatlands, either through drainage for resource extraction and infrastructure, or through climate change induced drought, has been shown to have long term impacts on C cycling (Harris et al., 2020; Kitson & Bell, 2020; Strack et al., 2006). One cause of peatland drainage is the extraction of peatlands for the production of horticultural growing media. To prepare a site for extraction, drainage ditches are created to lower the water table, and then the surface vegetation is removed. In addition to perimeter ditches, interior ditches running the length of the site are cut and spaced every ~30 m in Canada (Figure 3.1). A site is then vacuum harvested, the standard extraction method in Canada, for 15–40 plus years. Briefly, harrowing machines break up the top 5 cm layer of the peat, disconnecting it hydrologically from the peat layer below, allowing it to dry out. Harvesting machines vacuum up a portion of this dried peat, where it is transferred to stockpiles to further dry before being removed for processing and sold for use as a growing medium (Cleary et al., 2005). The water level in the interior drainage ditches fluctuates based on recent precipitation levels and site management (Hunter et al., 2024).

The actions associated with peat extraction alter the hydrology, peat structure, thermal properties, substrate quality, nutrient status, and microbial community of the peatland. Drainage-induced subsidence and the associated shift towards smaller pore size will decrease the specific yield, leading to larger water table fluctuations and increased upwards capillary movement of water (Holden et al., 2011; Kennedy & Price, 2005). The presence of drainage ditches will create hydrologic gradients across the fields (Waddington et al., 2002). The decreased soil moisture will decrease the heat capacity and thermal conductivity of the peat (Mäkiranta et al., 2009). These actions will also alter the nutrient dynamics (Basiliko et al., 2007; Glatzel et al., 2004; Kendall, 2021), decrease the total microbial biomass (Basiliko et al., 2007; Croft et al., 2001), and cause shifts in the composition of the microbial community present (Bieniada et al., 2023; Reumer et al., 2018).

These influence of peat extraction on these properties will also vary temporally. In the short term, harrowing will create hourly to multi-day periods where a thin, dry layer is hydrologically cut off from the peat below, affecting gas diffusion, water movement and soil temperature (O'Kane, 1992; Waddington & Price, 2000). In the medium to long term, the continued removal of peat each year will expose older, more recalcitrant peat to the surface, with this having a significant effect on CO_2 emissions after the first few years following the start of extraction (Clark et al., 2023).

These altered conditions result in these sites shifting from net sinks to net sources of C (Clark et al., 2023; Waddington et al., 2002). However, the magnitude of this shift, the seasonal patterns of C emissions, and the effect of site management on these emissions, is poorly understood. Only a few studies have conducted measurements of CO_2 and CH_4 emissions from actively extracted peatlands in Canada (Clark et al., 2023; Glatzel et al., 2003; Hunter et al., 2024). Internationally, there has been more research, predominantly in Finland, Estonia, Sweden, and the United Kingdom (Salm et al., 2012; Shurpali et al., 2008; Sundh et al., 2000; Vanags-Duka et al., 2022). Most of these studies have focused on CO_2 emissions, and only from the peat fields. We know that the drainage ditches are an important component of the C balance, with recent studies finding that they emit at least double the CO_2 and at least seven times the CH_4 emissions of the fields (Clark et al., 2023; Hunter et al., 2024). While multiple studies have been conducted at post-extracted unrestored peatlands, they have all been based in Quebec, Canada, and many contain non-functioning drainage ditches and partial natural re-vegetation (Glatzel et al., 2003; McNeil & Waddington, 2003; Rankin et al., 2018; Strack & Zuback, 2013; Waddington et al., 2010; Waddington et al., 2002). Additionally, due to fieldwork logistics, most extraction studies have focused on May to August emissions, meaning that we are missing important spring thaw and winter processes observed in other peatland systems (Raz-Yaseef et al., 2017; Wang et al., 2014).

There is also uncertainty regarding the environmental controls on these C gas emissions. Studies have generally reported positive relationships between CO_2 emissions and WTD, due to higher decomposition rates under oxic conditions (He & Roulet, 2023; Rankin et al., 2018; Waddington et al., 2002), which aligns with findings in vegetated drained peatlands (Evans et al., 2021). However, in the majority of these studies, average WTD was less than 50 cm. Despite being a drained system, extracted peatlands are prone to frequent WT fluctuations (Lai, 2022; Price, 2003), and local climate and site management can result in large differences in average WTD between sites, and between years within a site. Studies have found both strong positive (Rankin et al., 2018) and weak to no

effect (Clark et al., 2023) of soil temperature on C emissions from extracted peatlands, but have not considered how the range in WTDs and surface soil moisture content might affect the strength and direction of this relationship. We know that temperature dependence of CO_2 production can vary with moisture content (Liu et al., 2024; Swails et al., 2022), yet there is limited research on this in heavily drained peatlands.

While peat extraction only covers a small portion of Canadian peatlands, the long extraction duration of a site, and the shift from it being a net sink to a net source of C, means that a substantial amount of C is lost to the atmosphere from these sites, in addition to the loss of C from the physical removal of the peat. Yearly CO_2 emissions from numerical modelling and eddy covariance measurements at active and post-extracted unrestored sites cover a wide range from 151 to over 400 g C m^{-2} yr^{-1} (He et al., 2023; He & Roulet, 2023; Rankin et al., 2018; Waddington et al., 2002). Canada's current Tier 2 emission factor value falls within this range at 310 g C m^{-2} yr^{-1} (ECCC, 2021). To our knowledge there are no annual estimates of CH_4 emissions from extracted peatlands in Canada. Studies that can reduce uncertainty will aid Canada in its accounting of anthropogenic greenhouse gas emissions. A better understanding of C dynamics during extraction will provide the industry with better tools to balance C emissions, both gaseous and aquatic, with harvesting yields. It will also aid in estimating the C benefit arising from restoration of these sites. No study has measured ecosystem scale CO_2 and CH_4 emissions from actively extracted peatlands in Canada, and to our knowledge, only one study has been conducted in Europe (Holl et al., 2020). Eddy covariance provides the ability to conduct measurements at the ecosystem scale, capturing both field and drainage ditch dynamics at a higher temporal frequency than chamber measurements. The objectives of this study were therefore to i) assess seasonal and interannual patterns of CO_2 emissions, and seasonal patterns of CH_4 emissions, at two actively extracted peatlands; and ii) investigate the effect of WTD and soil temperature, along with their interaction, on CO_2 and CH_4 emissions.

3.3 Methods

3.3.1 Study Sites

This study was conducted at two actively extracted Canadian boreal peatlands. The site in western Canada (AB) (~8 ha), located near Drayton Valley, AB (53.222° N, 114.977° W), has been extracted since 2009. The site in eastern Canada (QC site; ~65 ha), located near Rivière-du-Loup, QC (47.836° N, 69.536° W), has undergone extraction since 2007 (Figure 3.1). At each site, prior to extraction, the

peat producers removed the surface vegetation and built drainage ditches to lower the water table and allow harvesting machinery to drive over the site. Perimeter ditches extend around the site, and a series of ~1 m deep, 0.5 m wide parallel interior drainage ditches, spaced 30 m apart, run the length of the site. The 30 m wide segments of peat between interior drainage ditches are referred to as fields (Waddington et al., 2009). Extraction occurs from May and October each year, with an average extraction duration of 20 to 30 years. Chamber based CO_2 and CH_4 emissions were previously measured during the summers of 2019, 2021 and 2022 at AB (Hunter et al., 2024) and 2018, 2019 and 2020 at QC (Clark et al., 2023).

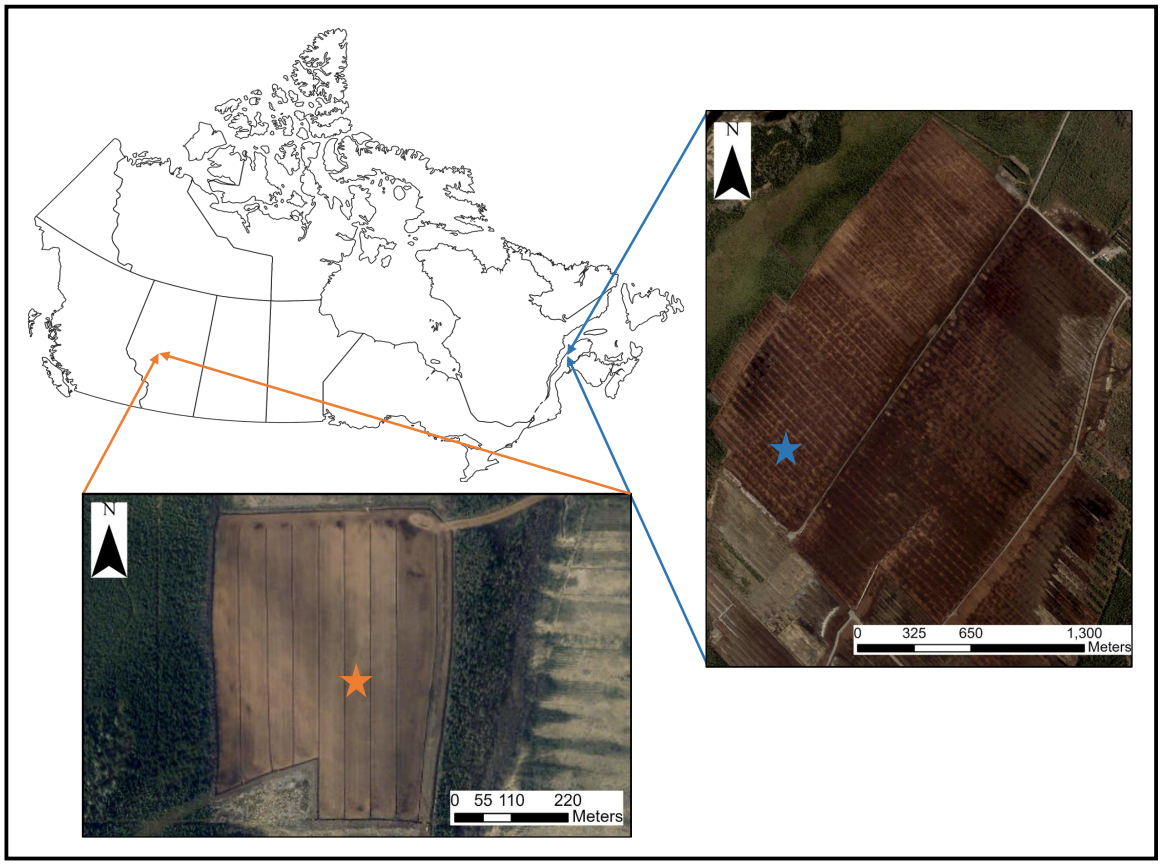


Figure 3.1 Arial images of AB (left) and QC (right) and their locations in Canada. The position of the eddy covariance towers at each site are indicated with stars. The map of Canada is by Craig Clark from Pixabay. The arial images were made using Esri (2024). Flux footprint models for each site in each year are shown in Appendix B, Figure B2.

3.3.2 Instrumentation

The near identical flux tower setup at each site consisted of open path analyzers for CO_2/H_2O and CH_4 (LI-7500A and LI-7700, LI-COR, Nebraska, USA), and a sonic anemometer (CSAT-3, Campbell Scientific, Edmonton, Canada). Sensors were mounted on a cross arm at 2.1 m (AB) and 1.8 m (QC) height. The high frequency (10 Hz) and 30-minute average data were stored on a USB drive connected to the control unit (LI-7550 AIU, LI-COR, Nebraska, USA). A series of environmental variables were logged half-hourly on a logger (CR3000, Campbell Scientific, Edmonton, Canada) including air temperature and relative humidity (HMP 45C, Vaisala, Finland) and incoming solar radiation (CNR-1, Kipp and Zonen, Netherlands). Due to large data gaps in air

temperature and incoming solar radiation at the AB site, we used weather station data for the whole study period. Air temperature was obtained from the Tomahawk AGDM weather station located ~11 km southeast of the AB site (53.43° N, 114.72° W; ACIS, 2023) and incoming solar radiation from the Evansburg 2 AGDM weather station (53.61 ° N, 115.06° W; ACIS, 2023), located ~ 20 km northwest of the AB site. A regression between the measured solar radiation data at AB and Evansburg radiation data was significant ($F_{1,38451}$=103305, p<0.0001), with an R^2 of 0.73 (Figure B1).

Additionally, soil temperature data was measured with Type T thermocouples (Omega Engineering, Stamford, Connecticut, USA) at a depth of 10 cm. At QC, the integrated top 14 cm VWC was measured using a 20 cm long water content reflectometer (CS 616, Campbell Scientific, Edmonton, Canada) inserted at a 45° angle. A combination of CS 655 and CS 616 probes were inserted horizontally into the peat at depths of 10 and 20 cm at the AB sites. The probes at both sites were logged half-hourly on a data logger (CR1000X, Campbell Scientific, Edmonton, Canada). Periods of insufficient power supply and equipment failure resulted in sporadic VWC measurements. Finally, WTD in 2020, 2021 and 2022 at AB and 2020 and 2022 at QC was measured every 10 seconds and averaged hourly using pressure readings from Micro-Divers (Van Essen, Waterloo, Canada). These instruments were placed at the centre of a field in a harvesting exclusion zone near each tower (Figure 3.1). The measured pressure values were converted to WTDs by subtracting the atmospheric air pressure data measured at the Stoney Plains climate station near AB (53.56° N, 114.11° W; ECCC, 2023b) and from a Micro-Diver hung above the peat surface at QC. A barometric correction was performed at AB to account for the elevation difference between our site and the climate station. Daily precipitation data was obtained from the Tomahawk AGDM station (ECCC, 2023c) and the Rivière-du-Loup Climate Station (47.81° N, 69.55° W; ECCC, 2023a), located ~3 km northwest of the QC site. A trail camera (Moultrie Products, Alabama, USA) was installed on the AB tower during August and September 2022 to capture the timing of extraction activities (see Text B1).

3.3.3 Flux Processing and Quality Control

The eddy covariance data was processed using EddyPro software (version 7.0.9, LI-COR, Nebraska) advanced mode with a WPL correction to account for the effect of humidity and air temperature on density (Webb et al., 2007). The CO_2 and CH_4 data were storage corrected, and then de-spiked (Papale et al., 2006). Fluxes beyond three standard deviations of the mean were marked as outliers and removed. To account for low turbulent conditions, only half-hourly fluxes with the highest

possible quality flag were used for this study (Mauder and Foken, 2004). Additionally, due to the potential for incorrect air density calculation, all 30-minute CO_2 and CH_4 fluxes where the corresponding sensible or latent heat flux quality flag (Mauder and Foken, 2004) was at the lowest value were removed. All CH_4 fluxes with a signal strength less than 30% were removed.

To remove periods of insufficient turbulence, we applied a u* threshold of 0.15, 0.14 and 0.18 at AB in 2020, 2021 and 2022, respectively, and 0.17 and 0.19 for the 2020 and 2022 at QC, respectively. This threshold was objectively determined using the REddyProc online processing tool (Wutzler et al., 2018). Due to the absence of photosynthetic uptake, we applied this correction to both daytime and nighttime fluxes. We applied the same u* threshold to the CH_4 data, assuming that insufficient turbulent conditions for CO_2 would also be insufficient for CH_4 fluxes. A 2D flux footprint at each site in each study year was estimated using the Flux Footprint Prediction (FFP) online tool (Kljun et al., 2015). Based on these calculations, fluxes from excluded from the analyses at AB when their 80% probability footprint distance was beyond the fetch of the site (Figure B2). Given the larger fetch at QC, no fluxes had to be removed (Figure B2).

Incoming solar radiation of 20 W m^{-2} was used as a cut-off between daytime and nighttime fluxes. For the CO_2 and CH_4 data, there were no months in our data set that had more than 25% of the original nighttime half hourly periods, with the majority having less than 15% (Table B1, Table B2, Table B3). This is due to the highly stable atmospheric conditions present overnight. We thus excluded all nighttime data from our analysis due to low data confidence. For the daytime CO_2 fluxes, we excluded all months that had fewer than 20% of the potential half hourly periods. Our resulting dataset for CO_2 includes data from March to October 2020, March to August 2021 and May to October 2022 at AB, and July to October 2020 and May to October 2022 at QC. Due to instrument damage at QC and COVID travel restrictions, our CH_4 dataset extends from May to August 2022 at AB only.

We observed significant exponential relationships between air temperature and CO_2 emissions at our study sites; however, they had a high degree of scatter and low correlation (Figure B3). Well established gap-filling methods, such as those employed by REddyProc use moving point windows to gap fill CO_2 data based on similar periods of global radiation, air temperature and vapour pressure deficit. These variables are more strongly linked, and have a shorter lag time, with photosynthesis than with heterotrophic respiration and the NEE from our extracted sites (with vegetation having been

removed) represents respiration only. Based on the low percentage of original flux data after strict quality control, we did not gap fill our data.

3.3.4 Calculation of Annual Carbon Budget

We estimated an annual CO_2 budget for extracted peatlands. Given the limited number of measurements at each site, we combined data from both AB and QC, and averaged data across all study years. Due to the absence of nighttime data, we assumed that average daytime (9 am to 5:30 pm) monthly March to October emissions were representative of the whole month. We modelled the missing wintertime (November to February) emissions in three ways. The first was to assume emissions of 0.5 g C m^{-2} d^{-1} (Webster et al., 2018); the second was to use a daily value equivalent to 15% of the March to October emissions rate (Saarnio et al., 2007); the third was to compute 50% of our average monthly May to October emissions, and then use that as a monthly emissions rate for the whole year (He & Roulet, 2023). For CH_4, we assumed that emissions for the missing months (October to April) were equivalent to 15% of the March to October emissions rate.

3.3.5 Statistical Analysis

The statistical analysis was performed in RStudio (R Core Team, 2019; RStudio Team, 2020). The scatter, line and boxplots were produced using the ggplot2 package (Wickham et al., 2016). We used QQ plots to visually assess the normality of the C fluxes (Zurr et al., 2009). The CH_4 fluxes were shifted by 6.12 mg C m^{-2} d^{-1} to remove negative values and then log transformed to meet the assumption of normality for analysis of variance (ANOVA) tests. For our analysis, we only used daytime data between 9 am and 5:30 pm local time. This was based on the low percent of available data that met our quality standards during the early morning and later evenings compared to the middle of the day. This daytime period is also representative of the times when we were able to make chamber-based C emissions measurements at these active sites (Clark et al., 2023; Hunter et al., 2024). A two-way ANOVA was performed on a linear model to determine the effect of month and location (factor with five levels combining site and year) on CO_2 emissions. For CH_4 emissions, a one-way ANOVA was performed to determine the effect of month on CH_4 emissions at AB in 2022. The Tukey post hoc test (emmeans package; Lenth et al., 2018) was used when a fixed effect was significant. To create a balanced design and remove the effect of unequal number of fluxes between half hourly periods, these models were run on monthly half hourly averages of CO_2 and CH_4. Linear

regressions were performed to determine the effect of soil temperature and WTD on C emissions using all available half-hourly data.

3.4 Results

3.4.1 Environmental Conditions

At AB, the 2020 study period (March to October) was wetter (482 mm) than the 15-year average of 377 mm, while the 2021 (341 mm) and 2022 (350 mm) periods were comparable to it (ACIS, 2023). Of note, in 2022, 65% of the precipitation occurred during June and July, compared to only 27% in 2021 (Table 3.1). Average air temperature across the study period was cooler in 2020 (7.6 °C), and warmer in 2021 (10.3 °C) and 2022 (9.9 °C) than the 15-year average (9.1 °C) (Figure 3.2) (ACIS, 2023). March and October had the largest monthly differences in temperature between years. At QC, the 2020 (609 mm) and 2022 (601 mm) study period precipitation totals were similar to the 15-year average (596 mm) (ECCC, 2023a). However, the spread was different, with 2020 being wettest in September and October compared to in March and May in 2022 (Table 3.1). Similarly, average air temperature was within 0.1 °C of the 15-year average of 9.6 °C (Figure 3.2).

Soil temperature followed the seasonal pattern of air temperature (Figure 3.2). Temperatures were a couple degrees higher in July onwards in 2022 compared to 2021 at AB, with smaller diurnal and daily temperature fluctuation. At QC, temperatures were similar between years, reaching a monthly average peak of ~ 20 °C in July of each year.

Both the average WTD, and the degree of water table fluctuations, were different among years and sites (Figure 3.2). At AB, WTD steadily increased from ~ 70 cm to 60 cm below the surface from April to August in 2021. In contrast, 2020 and 2022 were characterized by large water table fluctuations and WTDs of less than 40 cm for most of the study period. At QC, WTDs were shallowest in May (2020) and June (2022), though unlike AB, the water table was rarely within the top 30 cm of the peat. In 2022 at the QC site, the water table dropped to greater than 90 cm depth by early fall.

Volumetric water content generally followed the patterns in WTD (Figure B4). In 2022, horizontal VWC at 10 and 20 cm depth at AB fluctuated between 60 and 80% from May to July, and then gradually decreased to ~45% by October. The VWC was ~10% lower in July and August 2021 compared to 2022. Integrated top 14 cm VWC measurements at QC showed that VWC peaked at

~50-60 % in September and June in 2020 and 2022, respectively. Soil water retention curves showed relatively linear wetting and drying curves at QC, with a large degree of hysteresis (Figure B5). At AB, there was a break at around 55 cm WTD, with a steeper slope when the WTD was greater than 55 cm (Figure B5).

Table 3.1 Total monthly precipitation (mm) at AB and QC over the study years, and the 15-year average (from 2008 to 2022). This data was obtained from the Tomahawk climate station (for AB; ECCC, 2023c) and the Rivière-du-Loup climate station (for QC; ECCC, 2023a).

	AB				QC		
Month	2020	2021	2022	15-year Average	2020	2022	15-year Average
March	19.5	5.8	15.1	22.4	94.7	101.8	60.9
April	11.1	17.5	31.8	28.2	63.9	66.1	74.1
May	139.8	80.6	25.9	45.9	65.8	101.3	84.7
June	131.2	37.2	146.9	72.8	20.5	96	78.8
July	108	52.2	83.9	91.3	67.4	71.6	62.7
August	37.6	44.9	33.5	61.9	48.8	35	56.7
September	15.9	77.2	3.0	32.5	100	39.3	79.8
October	18.8	25.3	10.3	22.1	148	89.4	98.4
Sum	482	341	350	377	609	601	596

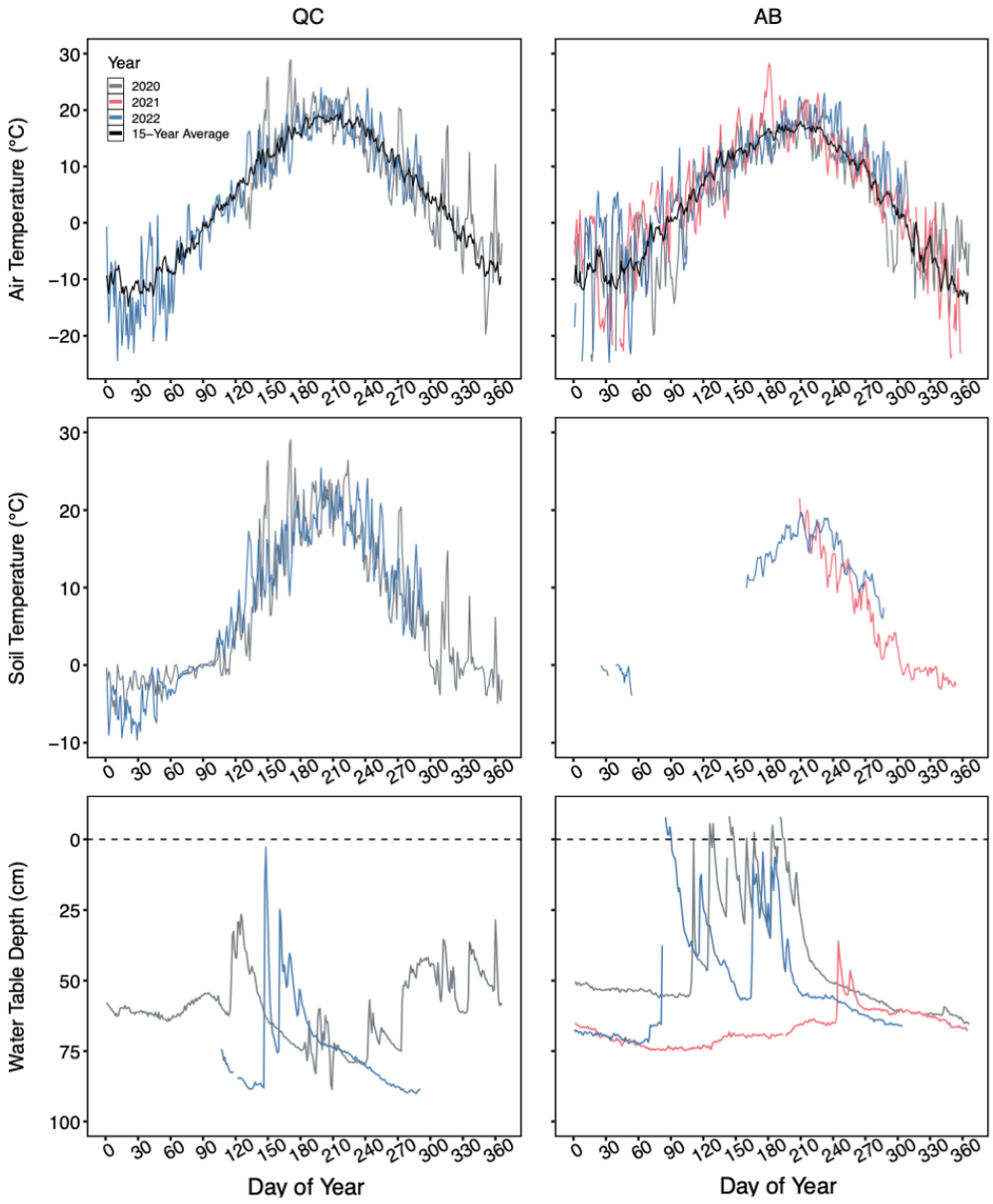


Figure 3.2 Daily mean air temperature (°C), 10 cm depth soil temperature (°C) and WTD (cm) at QC (left) and AB (right) during the study years. Water table depth was measured in the centre of a field (see Figure 3.1). A negative value implies that the water table was above the peat surface. The 15-year average (2008 to 2022) air temperature at AB and QC is shown in black. Air temperature at AB was obtained from the Tomahawk AGDM weather station (ACIS, 2023).

3.4.2 CO_2 and CH_4 Emissions

Mean monthly daytime (9 am to 5:30 pm local time) CO_2 emissions ranged from -0.19 g C m^{-2} d^{-1} to 1.0 g C m^{-2} d^{-1} at AB and 0.39 to 1.09 g C m^{-2} d^{-1} at QC (Figure 3.4, Table B4). There was a significant effect of month ($F_{7, 480}$=52.97 p<0.0001) and location (factor with five levels combining site and year; $F_{4,480}$=130.43, p<0.0001), and their interaction ($F_{18,480}$=9.65, p<0.0001) on monthly daytime CO_2 emissions. At AB, we observed a pattern of CO_2 emissions increasing during May and June, reaching a peak in July, and then decreasing in September and October (Figure 3.4). The effect was largest in 2021 and 2022, with a significant increase in average monthly CO_2 emissions of 0.6 and 0.5 g C m^{-2} d^{-1}, respectively from May to July (Table B4).

Late winter and early spring March and April emissions were not consistent between years. April 2020 emissions were comparable to those in June, while there was a small average uptake in 2021 in March (-0.01 g C m^{-2} d^{-1}) and April (-0.19 g C m^{-2} d^{-1}). Interannually, May to August emissions were highest in 2022, and lowest in 2021. While 2021 emissions were at least 25% higher than 2021 emissions during this period, this difference was only significant in August (Figure 3.4, Table B4).

We observed similar patterns at QC (Figure 3.4). Carbon dioxide emissions peaked in June and July, in 2020 and 2022, respectively, significantly decreasing by 0.43 and 0.56 g C m^{-2} d^{-1} by October in their respective years (Table B4). Of note, August 2020 emissions were significantly lower than in both July and September (Table B4). Interannually, July and August emissions were 0.28 and 0.45 g C m^{-2} d^{-1} higher in 2022 than 2020, while September and October emissions were similar between years. When comparing between sites, May and June CO_2 emissions in 2022 at QC were significantly higher than those in the three AB study years. The effect size was large, with QC being over four times the 2021 AB emissions, and at least 0.57 and 0.41 g C m^{-2} d^{-1} higher than the 2020 and 2022 AB CO_2 emissions, respectively. July, August, September and October CO_2 emissions, which were measured at each site over two years, were similar between sites and years, with the highest emissions generally at QC and AB in 2022 and the lowest average emissions at AB in 2021.

Methane emissions were only measured for one year at AB (Figure 3.5, Table B5). There was a significant effect of Month ($F_{3,64}$=5.68, p=0.0016) on CH_4 emissions. We observed a seasonal pattern, with average monthly daytime (9 am to 5 pm) emissions increasing significantly from 6.54 mg C m^{-2} d^{-1} in May to 9.13 mg C m^{-2} d^{-1} in August.

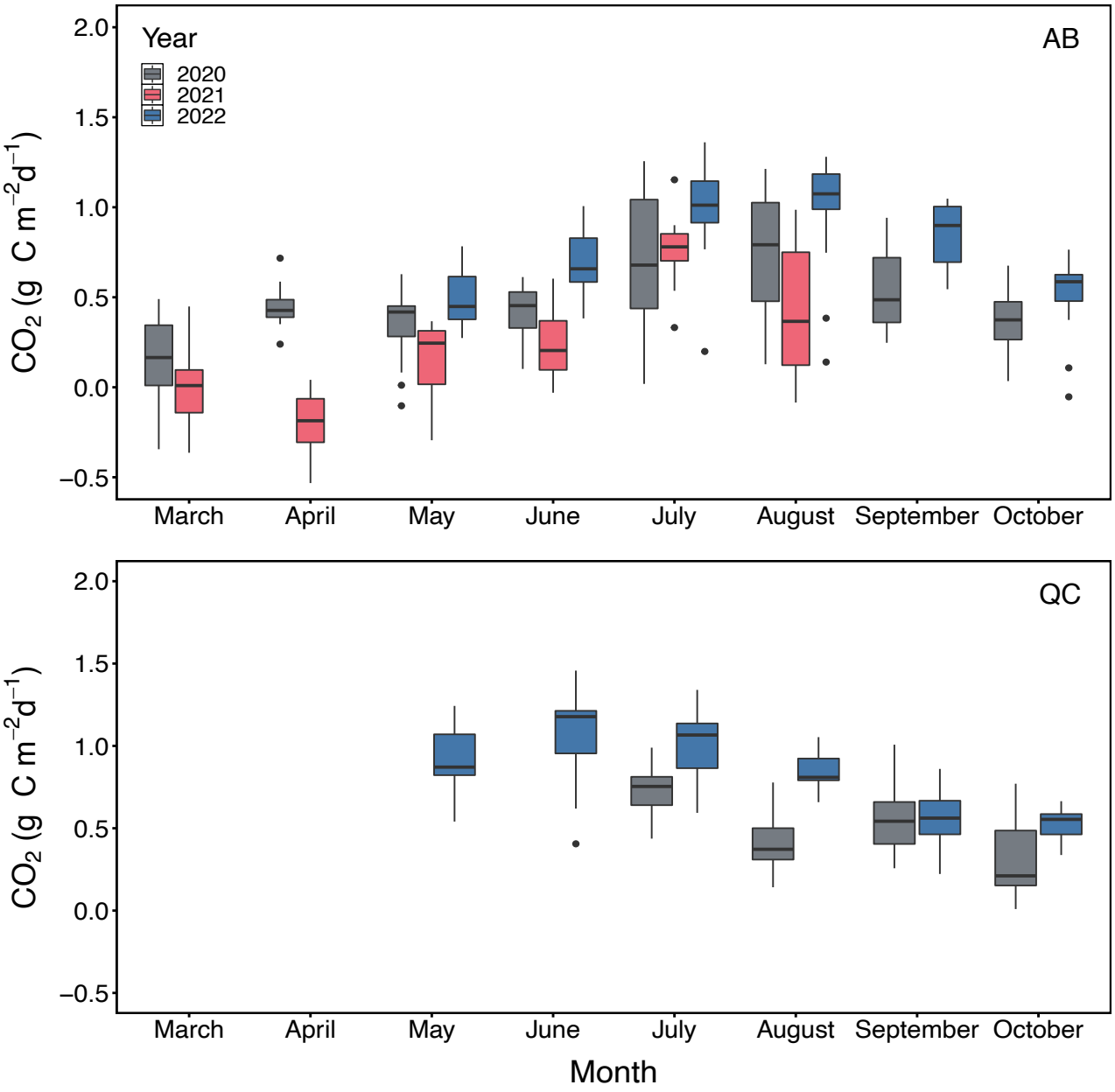


Figure 3.3 Boxplots of average monthly daytime (9 am to 5:30 pm local time) CO_2 emissions during 2020, 2021 and 2022 at AB (top) and QC (bottom). Measurements from March and April in 2022 at AB, from March, April, May and June at QC in 2020, and from March and April at QC in 2022 are missing due to equipment damage and COVID restrictions. See Table B4 for summary of results from post hoc test performed on linear model looking at the effect of location and month on CO_2 emissions.

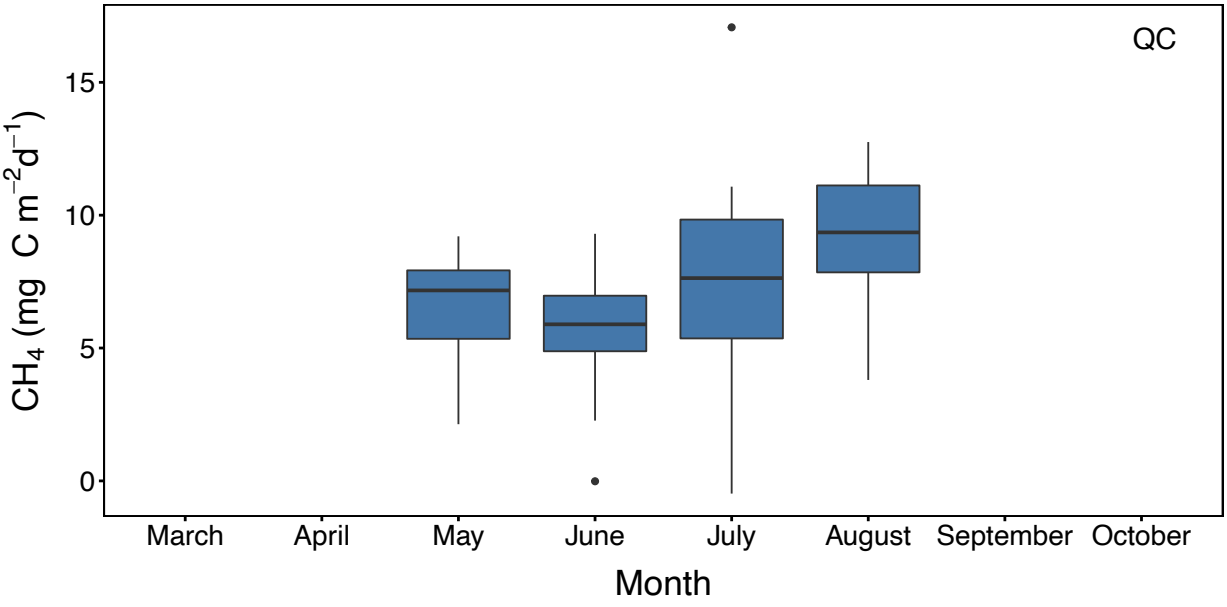


Figure 3.4 Boxplots of average monthly daytime (9 am to 5:30 pm local time) CH_4 emissions during 2022 at AB. Measurements from March and April and October are missing due to equipment damage and COVID restrictions. See Table B5 for summary of results from post hoc test performed on linear model looking at effect of month on CH_4 emissions.

3.4.3 Annual CO_2 and CH_4 Budget

We estimated annual CO_2 budget from extracted peatlands using three methods to fill in missing wintertime data (see Section 3.3.4). Assuming wintertime emissions of 0.5 g C m^{-2} d^{-1}, we estimated annual emissions of 178 g C m^{-2} yr^{-1}. We estimated emissions of 129 g C m^{-2} yr^{-1} when wintertime emissions were 15% of March to October emissions. Finally, assuming monthly emissions for the whole year were 50% of average March to October emissions, we estimated an annual CO_2 budget of 112 g C m^{-2} yr^{-1}. By assuming the October to April CH_4 emissions were 15% of May to September emissions, we estimated annual CH_4 emissions of 1.1 g C m^{-2} yr^{-1}.

3.4.4 Environmental Controls of CO_2 and CH_4 Emissions

We observed significant positive linear relationships between half-hourly CO_2 emissions and 10 and 20 cm depth soil temperature at AB, though they had high degree of scatter and low explanatory power, explaining only 0.8% and 2% of the variation in CO_2 emissions, respectively. The results were similar at QC, with 10 cm depth soil temperature explaining 0.7% of the variation, though soil

temperature was not significant at 20 cm depth. Soil temperature was more strongly correlated with daily CO_2 emissions, with 20 cm depth temperature explaining 15% of the variation in CO_2 emissions at each site. Given that most studies in drained peatlands have had WTDs less then 50 cm, we divided our data into periods where WTD was greater than and less than 50 cm (Figure 3.5). At Seba, there was a significant positive relationship when WTD was less then 50 cm ($F_{1, 1524}$=45, p<0.0001) and a significant negative relationship when WTD was greater than 50 cm ($F_{1, 3125}$=390, p<0.0001), though they explained only 3% and 11% of the variability, respectively. At RdL, there was a significant negative relationship when WTD was greater than 50 cm ($F_{1, 1304}$=27, p<0.0001), though it explained only 2% of the variability. There was no significant relationship when WTD was less than 50 cm.

There was however a significant effect of the interaction between soil temperature and WTD on daily mean CO_2 emissions at AB (20 cm depth: $F_{3, 100}$ =8.72, p<0.0001) and QC (10 cm depth: $F_{3, 211}$ =23.59, p<0.0001). Grouping the CO_2 data by WTD revealed that the relationship between CO_2 and soil temperature changed with WTD. We categorized WTD as wet periods (WTD < 25 cm), moderate periods (50 cm > WTD > 25 cm), dry periods (75 > WTD > 50 cm) and very dry periods (WTD > 75 cm). At AB, during dry periods there was no significant effect of soil temperature on CO_2 emissions (Figure 3.6, Table B6). This relationship shifted during wetter periods, with significant positive relationships when the water table was between 25 and 50 cm below the surface, with soil temperature explaining 59% of the variance in CO_2 emissions. There was no significant effect when the WTD was less than 25 cm, likely due to the very small sample size (Figure 3.6, Table B6). At QC, there was a significant positive effect of soil temperature on CO_2 emissions at all WTD levels, though the relationships were strongest, and had the steepest slope, during wetter periods. We observed no effect of harrowing or harvesting on ecosystem scale CO_2 emissions (Figure B6).

There was a significant positive effect of soil temperature at 20 cm depth on daily CH_4 emissions at AB ($F_{1, 117}$=15.98, p<0.0001), explaining 11% of the variation (Figure B7). There was a weaker relationship with half-hourly CH_4 emissions at AB ($F_{1, 1264}$=49.42, p<0.0001), as soil temperature only explained 4% of the variation in CH_4 emissions. Due to the small number of data points, and the narrow range of temperatures recorded when the WTD was less than 50 cm, we did not divide the data by WTD level.

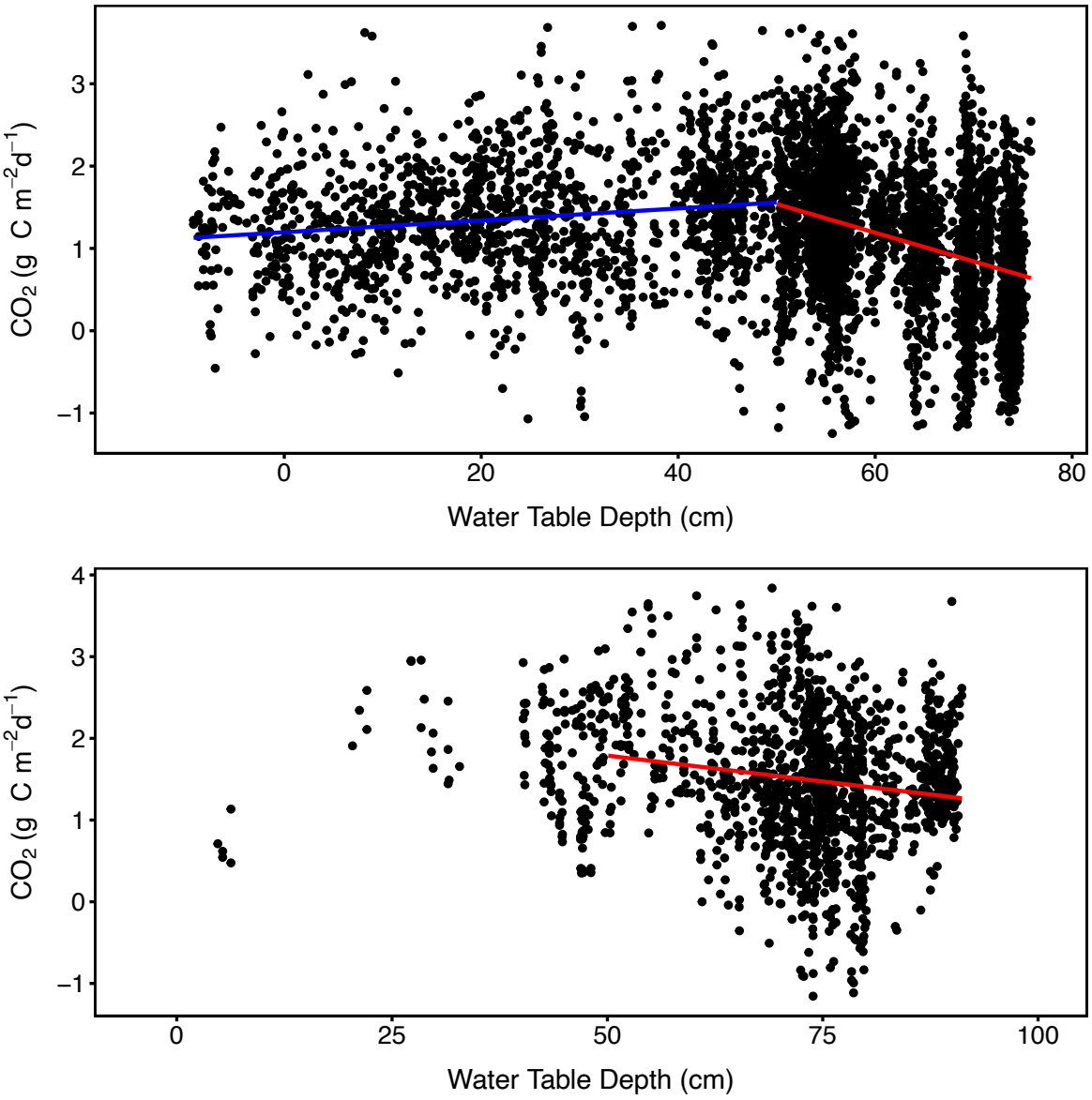


Figure 3.5 Scatterplots showing the effect of water table depth (cm) on CO_2 emissions at AB (top) and QC (bottom) across the study years. Significant linear regressions are shown for periods when the WTD was less than 50 cm (blue) and greater than 50 cm (red).

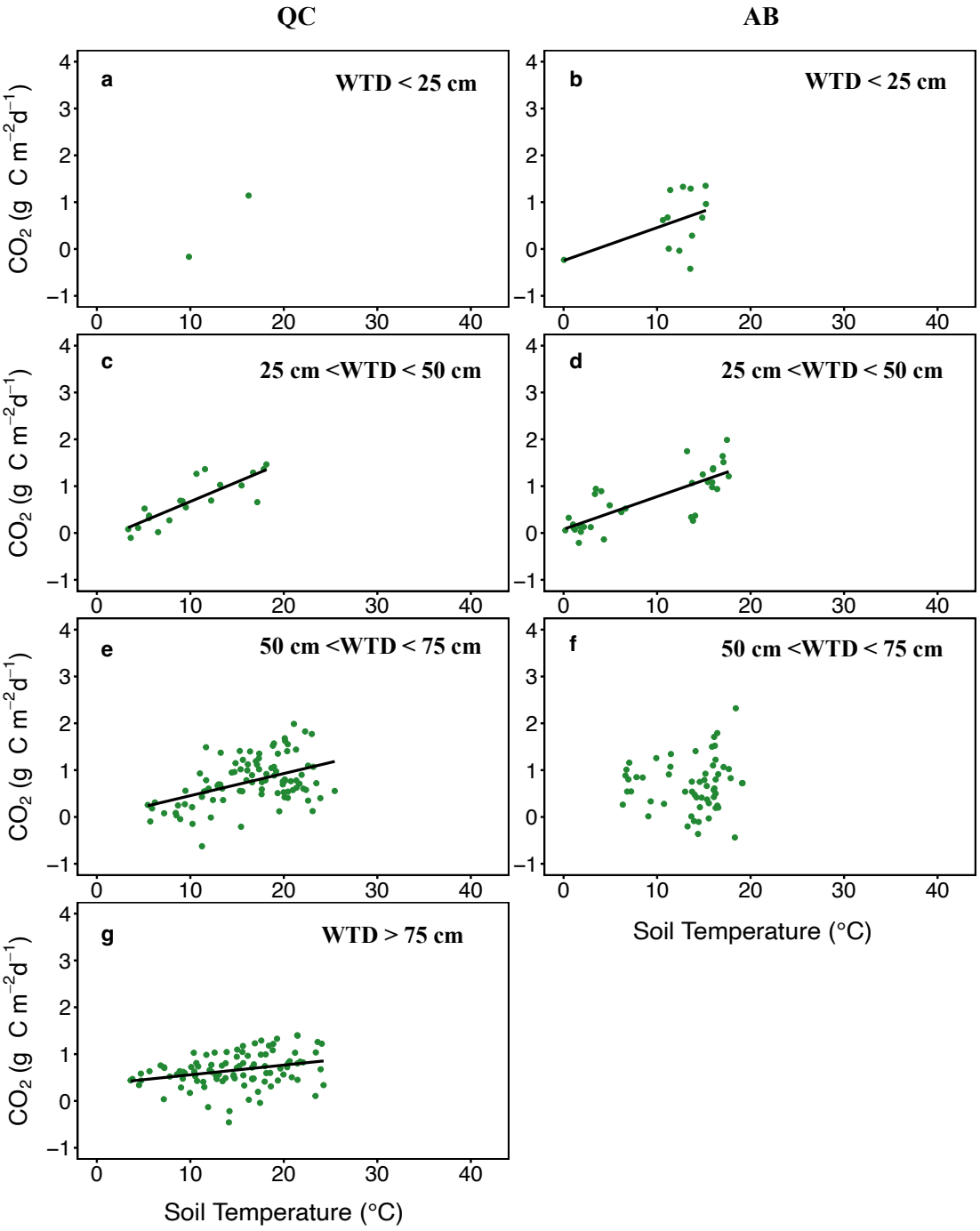


Figure 3.6 Scatterplots showing the effect of 10 cm depth (QC) and 20 cm depth (AB) soil temperature on daily daytime (9 am to 5:30 pm) CO_2 emissions at QC (left column) and AB (right column) across the study years. The CO_2 data has been divided by WTD, with wet period (a, b: WTD < 25 cm), moderate period (c, d: 50 cm > WTD > 25 cm), dry period (e, f: 75 cm > WTD > 50 cm) and very dry period (g: WTD > 75 cm). All linear regressions are significant (Table B6).

3.5 Discussion

Across both study sites, we observed important controls on C emissions that must be considered when estimating the annual greenhouse gas budgets of extracted peatlands. These include the interacting effects of temperature and WTD on CO_2 emissions, effect of moisture limitation on CO_2 emissions at deeply drained sites, and the importance of springtime emissions.

3.5.1 CO_2 Emissions

We measured comparable ecosystem-scale daytime emissions at QC and AB. In contrast, previous multi-year chamber studies from our study sites reported that AB (Hunter et al., 2024) had around double the site-integrated ditch and field CO_2 emissions as QC (Clark et al., 2023). Clark's et al. (2023) average summertime emissions of 0.76 and 2.56 g C m^{-2} d^{-1} from the fields and ditches, with a field to ditch ratio of 30:1, results in sector emissions of 0.82 g C m^{-2} d^{-1}. This is in line with our average emissions at QC. Hunter et al. (2024) calculated integrated summer June to August emissions of 1.47 g C m^{-2} d^{-1} at AB, which is just over double our average emissions of 0.66 g C m^{-2} d^{-1} during that same period.

3.5.1.1 Decreased CO_2 Emissions During Periods With Deep Water Tables

The persistent drainage at extracted sites means that surface VWC may be below the optimum for aerobic decomposition for large portions of the year. Our interannual and seasonal patterns suggest a decrease in CO_2 emissions during periods with deep water tables. At AB, May, June and August average emissions were at least double (at least 0.2 g C m^{-2} d^{-1} higher) in 2020 and 2022 compared to 2021, which coincided with 20 to 50 cm deeper water tables in 2021 compared to the other two years. Our soil temperature data suggests that the surface temperature varied by only a few degrees between these years. This is consistent with findings by Hunter and others (2024) who observed significantly lower chamber emissions in 2021 compared to 2022 at AB. Seasonally, in 2022 at QC, we observed some of the highest emissions when WTDs were their shallowest (June and May). This was despite the peat being 10 to 20 ºC cooler than later in the summer. Thus, both AB and QC results show higher CO_2 with wetter peat conditions (water tables closer to the surface). This directly contrasts with other studies and data compilations at extracted peatlands, which have observed reduced emissions during wet periods (He & Roulet, 2023; Holl et al., 2020; Rankin et al., 2018). This reported reduction in emissions can be considerable, with CO_2 emissions during a wet summer being 24% of the dry summer emissions (Waddington et al., 2002). The exception is a study in Finland that measured over

double the emissions in a wet year compared to a dry year (Shurpali et al., 2008). It is important to note that most of these studies had water tables within the top 50 cm of the peat profile. Previous studies in drained peatlands have also generally observed higher emissions during dry periods (Evans et al., 2021; Nielsen et al., 2023; Salm et al., 2012; Wilson et al., 2016), though one study in a heavily drained peatland for forestry found a positive relationship with CO_2 emissions when WTDs were greater than 70 cm (Mäkiranta et al., 2009).

We offer four possible explanations for the conflicting results on the effect of WTD on CO_2 emissions. The first is that the degree of drainage varies considerably across disturbed peatlands. Our sites can be considered heavily drained, with water tables deeper than 50 cm for 68% and 84% of the days during the AB (March to October) and QC (May to October) study periods, respectively. The water table–CO_2 relationship is likely unimodal, with peak emissions coinciding with a WTD that maximizes aerobic conditions while also maintaining sufficient surface moisture conditions for substrate diffusion (Byun et al., 2021; Ojanen & Minkkinen, 2019). When water tables varied within the top 50 cm of the peat, as is the case for many previous studies in drained peatlands (Evans et al., 2021), we observed a weak but significant positive relationship at AB. While we have limited data points at QC, it appears that there is a positive relationship between WTD and CO_2 emissions when the WTD is less than 30 cm. This work shows that we may need to divide drained peatlands into classes based on depth of drainage when trying to understand CO_2-water table relationships. Secondly, WTD measurements in heavily drained peatlands may provide misleading information about redox conditions in the near surface peat. Peatlands with high specific yield can maintain high surface moisture contents despite deep water tables (Lai, 2022; Price et al., 2003). Additionally, wetting fronts may never reach the water table, resulting in increases in surface moisture content, but no measurable change in WTD (Waddington et al., 2002). Therefore, continuous measurements of surface VWC should be conducted in future studies to better assess how changes in WTD relate to surface moisture conditions. Thirdly, this study could not assess whether there was a hysteretic relationship between soil moisture and WTD, and how the frequent water table fluctuations in May and June may have contributed to the observed CO_2 emissions (Rezanezhad et al., 2014).

Fourthly, there has been limited research on the shifts in the microbial community following long term drainage of extracted sites (Basiliko et al., 2007; Bieniada et al., 2023; Croft et al., 2001), and how these changes will affect CO_2-water table relationships. Therefore, while results at these sites suggest that maintaining deep water tables could be an important management strategy to reduce CO_2

emissions, more work will be needed to generalize this to other sites. Given the management (harrowing and harvesting) activities at these sites, the surface layer (top 5 cm) is expected to undergo hourly to multi-day changes in moisture content and bulk density (Lai, 2022; O'Kane, 1992). The effects of management on C gas production and transport have not been adequately studied; however, we did not observe noticeable effects of these activities on CO_2 emissions (Figure B6).

3.5.1.2 Water Table Depth Affects the Dependence of CO_2 Emissions on Soil Temperature

Although soil temperature on its own does not explain much of the variance in CO_2 emissions at our sites, when categorized by WTD, a stronger trend emerges likely due to the dependence of soil temperature impacts on surface soil moisture conditions. Based on thermodynamics of peat decomposition, studies generally observe higher aerobic CO_2 production with increasing peat temperature (Byun et al., 2021; Limpens et al., 2008). Our seasonal patterns highlight the importance of soil temperature as a control on CO_2 emissions at the monthly time scale. However, our regressions yielded weak positive relationships with a high degree of scatter and less than 16% explanatory power. Soil temperature had a moderate positive effect on chamber based measurements at AB (Hunter et al., 2024), and this mismatch could be due to measurement methods. In that study, temperature measurements were made directly adjacent to the C flux measurements, while our study made temperature measurements in a harvesting exclusion zone adjacent to the eddy covariance tower. It is possible that our 10 cm depth soil temperature was not representative of the temperature of peat directly below a harrowed layer in the fields. This could also explain why soil temperature at 20 cm depth was a stronger predictor than at 10 cm depth at AB, as it was less likely to be affected by harrowing and harvesting practices.

When CO_2 data was grouped by water table position, we found that there was a stronger effect of soil temperature when the water table was within the top 50 cm of the peat. The decreasing importance of soil temperature during dry periods is likely due to the increasing importance of surface soil moisture for decomposition (see Sect. 3.5.1.3). In addition to measurement method, this complex interaction between water table and temperature control on CO_2 emissions could help explain the mismatch in the importance of soil temperature within the peat extraction literature. Studies in extracted peatlands (both active and unrestored) using chamber and eddy covariance techniques have observed strong positive effects (He et al., 2023; Rankin et al., 2018; Shurpali et al., 2008), moderate (Hunter et al.,

2024; Salm et al., 2012; Waddington et al., 2002) and weak to no effect (Clark et al., 2023; Sundh et al., 2000) of soil temperature on CO_2 emissions. These studies generally did not measure hydrologic conditions, or report VWC measurements from different depths, making it hard to compare among them. However, the strong effect observed by Rankin and others (2018) was at a post-extraction, unrestored peatland with a water table that was within the top 50 cm of the peat until mid-August, and we might thus expect a stronger temperature dependence. In our data set, while the water table was within the top 50 cm for a large portion of one of our study years (2022) at AB, most of our data includes periods with deeper water tables.

This importance of hydrologic conditions on the effect of soil temperature on CO_2 emissions has not been well-explored, with the presence of vegetation at many sites making it difficult to disentangle the impact of water table-soil temperature interactions on autotrophic (plant) respiration versus heterotrophic respiration. While the majority of recent peatland studies on soil respiration have found higher temperature dependence in drier conditions (Li et al., 2021; Liu et al., 2024; Swails et al., 2022), a few studies in drained extracted peatlands observed similar results to ours (Shurpali et al., 2008; Waddington et al., 2002). Additionally, in soils drained for forestry and agriculture, multiple studies have observed increased dependence of heterotrophic CO_2 emissions on soil temperature under wet periods (Balogh et al., 2011; Scott-Denton et al., 2006). The VWC that constitutes "dry" conditions varies among studies, and therefore, comprehensive measurements of surface VWC at heavily drained sites would allow us to better compare our findings to the wider literature. Nevertheless, our findings will have implications for modelling work at drained peatlands. Models that rely on CO_2 – temperature relationships without considering the interacting effects of soil moisture may overestimate emissions during very dry periods.

3.5.1.3 Soil Temperature Relationships Underestimate March and April CO_2 Emissions

March and April are associated with the start of peat thawing and snow melt. April fluxes were comparable to summer values at AB in 2020 (Figure 3.3), demonstrating the importance of capturing spring thaw emissions in our annual C budgets. Episodic pulses of CO_2 in the late winter and early spring have been measured at some drained peatlands (Rankin et al., 2018), but not others (Holl et al., 2020). While we lack soil temperature data in those months, average air temperatures were just above freezing, compared to around 20 °C in July, showing that temperature relationships alone cannot

explain CO_2 emissions during this period. Possible mechanisms include the presence of ice lenses in the frozen soil and snow layer that trap produced CO_2 during the winter, leading to the release of this built-up CO_2 when the soil thaws (Raz-Yaseef et al., 2017). Freeze-thaw cycles have also been shown to alter respiration rates though changes in microbial community structure and enzyme activity (Matzner & Borken, 2008; Wang et al., 2014). Understanding both the magnitude, and the underlying processes involved with these springtime CO_2 emissions will be important as they can contribute a significant portion to the yearly net C balance and are often not captured in northern peatland measurements. Modelling studies that use CO_2 - temperature relationships may underestimate CO_2 fluxes during this period. However, we also observed slightly negative average CO_2 fluxes in March and April 2021 at AB, when there should have been no biological source of CO_2 uptake. Eddy covariance studies have observed false uptakes caused by instrument heating under cold conditions (Burba, 2006). These springtime emissions should therefore be interpreted with caution.

3.5.2 Extracted Sites are Small Sources of CH_4

Our study suggests that methane emissions at AB are considerably lower at active peat extraction sites than in natural peatlands. Our measured mean May to August CH_4 emissions of 7.22 mg C m^{-2} d^{-1} (median of 7.51 mg C m^{-2} d^{-1}) is considerably lower than the growing season 29.9 to 72.7 mg C m^{-2} d^{-1} reported for natural boreal bogs in Canada (Turetsky et al., 2014; Webster et al., 2018). It is also considerably lower than the 18.4 to 52.9 mg C m^{-2} d^{-1} site integrated emissions (combines field and ditch emissions) calculated for this same site using chambers (Hunter et al., 2024). Our mean is slightly higher than Hunter's measured field emissions of 2.1 mg C m^{-2} d^{-1} (median of 0), confirming that the ditches are important contributors to ecosystem-scale emissions. Given the errors associated with each method, a mismatch between chamber fluxes and ecosystem scale fluxes is not uncommon (Peltola et al., 2015; Riederer et al., 2014). Despite direct chamber measurements indicating that ditches can emit strongly (Clark et al., 2023; Hunter et al., 2024), the CH_4 emissions from extracted peatlands are likely lower than from natural peatlands, and only make up a small portion of the total site C budget (see Section 3.5.3).

The CH_4 emissions at AB in 2022 followed the expected seasonal pattern, with emissions increasing over the summer following seasonal temperature patterns, as has been observed in natural peatlands (Abdalla et al., 2016). To our knowledge there are no reported March and April CH_4 emissions from actively extracted, or post extraction unrestored, peatlands in Canada. In natural peatlands, ebullition

associated with spring thaw can be significant, with some studies reporting larger values than the annual diffusive CH_4 emissions from a site (D′Acunha et al., 2019).

Water table depth and soil temperature were not strong predictors of CH_4 emissions, likely due to a combination of the unique site hydrology, and the overall low CH_4 emissions. The absence of vegetation at our sites means that diffusion, and the associated oxidation of CH_4 to CO_2 in the oxic peat layer, will be the primary transport pathway of CH_4 (Lai, 2009). While peatland studies have usually observed a negative relationship between WTD and CH_4 emissions (Abdalla et al., 2016; Dean et al., 2018; Wilson et al., 2016), we did not find a significant effect. This is likely due to the deep water table at the AB site. Couwenberg et al. (2011) found that peatland CH_4 emissions were near zero for WTDs greater than 20 cm, making it hard to detect a WTD effect. The large water table fluctuations at the sites combined with the known long lag time between the establishment of anoxic conditions and the start of CH_4 production could also make it difficult to isolate the effect of WTD on CH_4 emissions (Blodau et al., 2004). Finally, the high specific yield of extracted peatlands (Price et al., 2003) can also support the presence of anaerobic pockets for CH_4 production in the surface peat, even during periods of deep water tables (Bieniada & Strack, 2021).

We observed a weak effect of soil temperature on CH_4, as has been seen in other extracted sites (Clark et al., 2023; Sundh et al., 2000). In natural peatlands, the importance of soil temperature is complex and varies with season, hydrologic conditions, and vegetation type (Chang et al., 2021; Turetsky et al., 2014). Given the observed WTDs, the majority of CH_4 was likely produced below the soil temperature sampling depth, or in the drainage ditches, where we did not measure soil temperature. Future measurements from dry and wet years will aid in understanding CH_4 dynamics from extracted peatlands and support management plans that balance harvesting yields with CO_2 and CH_4 emissions.

3.5.3 Implication for Annual Carbon Budget and Emission Factors

To date, estimates of CO_2 emissions from extracted peatlands in Canada are wide ranging. A modelling study based at QC reported emissions of 151–168 g C m^{-2} yr^{-1} from the fields (He et al., 2023). At post-extracted unrestored peatlands, Rankin and others (2018), using eddy covariance, reported an annual C budget of 173 to 259 g C m^{-2} yr^{-1}, while Waddington and others (2002) modelled (using chamber data to validate) emissions of 363 to 399 g C m^{-2} during a dry year, and 88 to 112 g C m^{-2} during a wet year. Some vegetation had naturally regenerated since the end of

extraction at Rankin and other's site, and so their value likely includes some photosynthetic uptake of CO_2. Our calculated annual C budget range of 112 to 178 g C m^{-2} yr^{-1} is considerably lower than the current Tier 2 emissions factor value of 310 g C m^{-2} yr^{-1} that Canada is using (ECCC, 2021). A recent study (He & Roulet, 2023), using model simulations run with 27 years of climate data, suggested a revised Tier 3 emission factor value of 139 g C m^{-2} yr^{-1} (1.4 t C ha^{-1} yr^{-1}) for a standard extraction site in Eastern Canada. Our data straddles this proposed value. Our findings suggest that the CO_2 emissions make up the bulk of an extracted site's total C budget. If CH_4 emissions for the winter periods (October through April) were 15% of our daytime average measured emissions (Saarnio et al., 2007), annual emissions of CH_4 would be only 1.1 g C m^{-2} yr^{-1}. Assuming a global warming potential over a 100-year time period for CH_4 that is 28 times that of CO_2 (IPCC, 2013), CH_4 emissions are equal to 41 g CO_2e m^{-2} y^{-1}. Methane emissions are thus at most 9% of the total CO_2 and CH_4 emissions from these sites, not including downstream gaseous loss. Export of C as DOC is likely a small component of the C budget, with previous research finding DOC export at AB was only <2 g C m^{-2} yr^{-1} during 2021 (dry year) and <10 g C m^{-2} yr^{-1} in 2022 (wet year) (Frei, 2023). Despite the small contribution of CH_4 to sitewide emissions, management actions to reduce CH_4 emissions may be the easiest to implement to achieve greenhouse gas emissions reductions from actively extracted sites. Given the at least 7 times higher emissions from ditches compared to fields (Chapter 2; Clark et al., 2023), increasing the drainage ditch spacing could substantially reduce CH_4 emissions.

3.6 Conclusion

Our study is the first to measure ecosystem scale CO_2 and CH_4 emissions from actively extracted peatlands in Canada. Seasonally, we observed emissions in April that were comparable to those in July in some years, highlighting the importance of including the spring thaw period (which has generally not been measured in extracted peatlands) in the annual C budget. We also observed that interannually, CO_2 emissions can vary by as much as 0.5 g C m^{-2} d^{-1}. The environmental controls of these CO_2 emission patterns remain poorly understood, with our results suggesting higher emissions during wetter periods, in contrast to the limited number of previous studies that have shown the opposite. We suggest that because extracted peatlands are deeply drained, dividing the data into categories based on WTD could help better understand these mechanisms. Our results also suggest an interaction between soil temperature and WTD, where surface soil temperature has only a weak effect on CO_2 and CH_4 emissions under deep (> 50 cm below surface) water tables. This means that

process-based modelling studies that use C-temperature relationships alone have the potential to overestimate emissions during drier years. Water table depths at these sites are controlled by both local weather, and site management, including the spacing and the depth of the drainage ditches. Understanding how C emissions vary with fluctuations in WTD will help inform site management plans, allowing companies to balance C emissions with harvesting yields.

At AB, CH_4 emissions were less than a tenth of the emissions reported for natural boreal bogs, suggesting that actively extracted peatland sites are a negligible source of CH_4. However, more work will be needed to understand how well ditch CH_4 emissions are captured by eddy covariance towers. We were unable to quantify nighttime and wintertime CO_2 and CH_4 emissions, leading to a large range of estimated yearly emissions. The current Tier 2 emission factors for CO_2 emissions from Canadian extracted peatlands are well above our estimated 112 to 178 g C m^{-2} yr^{-1}, suggesting that they may not be appropriate for peat extraction sites in Canada.

Chapter 4

Bare peat evaporation dynamics from actively extracted peatlands in Alberta and Quebec under varying water table and management conditions

4.1 Abstract

Peat extraction alters the peat hydrophysical properties of the peat during the 15 to 40 plus years of extraction. However, despite the importance of hydrologic conditions on C emissions from the long-term net C source, few studies in Canada have quantified the impact on energy partitioning and evaporation (E) rates. Vegetation removal alters the controls and mechanisms of water loss, and drainage induced subsidence creates strong vertical capillary connectivity through the peat profile. We thus conducted a multi-year study using eddy covariance to understand the energy balance and daytime E rates from actively extracted sites in Quebec and Alberta, Canada. Continuous measurements of water table depth (WTD), volumetric water content (VWC) and vapour pressure deficit (VPD) were used to understand controls of energy partitioning.

This work confirmed that WTD was not a good predictor of surface moisture conditions, as high E rates were maintained even with WTD of 60 cm below the surface. Site specific WTD thresholds highlighted how the relative importance of atmospheric and surface controls on E rates varied with WTD. During active extraction, site managers harrow (till) the top few cm of peat on a daily to multi-day time scale, forming a dry layer that is hydrologically cut off from to the layer below. A weighing bucket lysimeter experiment found that while harrowing elevates E rates in the first few hours, the newly dry layer then acts as a barrier to further water loss. This research can improve understanding of surface and atmospheric controls on E rates in peatlands due to the absence of vegetation, and can support modelling efforts that attempt to partition evapotranspiration. An understanding of the impact of management on E rates can aid companies in optimizing practices.

4.2 Introduction

Peatland disturbance is increasing worldwide, putting the fate of this long-term carbon (C) sink in jeopardy (Frolking et al., 2011; Harris et al., 2022). Of particular concern is the drainage of peatlands for agricultural and industrial use, which has affected an estimated 10% of peatlands worldwide (UNEP, 2022). Peat extraction for horticultural peat use is a source of peatland drainage in Canada. Despite affecting less than 0.03% of Canadian peatlands (Rochefort et al., 2022), the conversion of

these sites from net sinks to net sources of C during the 15–40 plus years of extraction (Clark et al., 2023; Hunter et al., 2024), along with the expected increase in demand for horticultural peat (Blok et al., 2021) makes them an important disturbance to study. The initial site drainage, along with subsequent management during extraction, alters the peat hydrophysical properties, the energy partitioning, and the sitewide water balance. Given the strong dependence of the production and transport of gaseous and aqueous C fluxes on hydrologic conditions (Evans et al., 2021; Peacock et al., 2021a; Strack et al., 2015), it is vital that we understand the magnitude and controls of energy partitioning and evaporation (E) following drainage.

Prior to extraction, perimeter and interior drainage ditches are cut to drain the site, and the surface vegetation is removed (Cleary et al., 2005). The sites are then actively extracted through a repeating two-step process. First, machinery is driven over the surface to harrow the peat to dry the surface layer (3–5 cm) by disrupting the hydrologic connection with the peat below. Then, this newly dried layer is removed using vacuum harvesting, where it is further dried, processed, and then sold as a horticultural growing medium (Roberts et al., 1998). Depending on precipitation frequency and the surface moisture content, the harrowed peat will remain unharvested for half a day to a couple of days and may be re-harrowed multiple times before the surface peat is extracted (Warren Walker, Sun Gro Horticulture Natural Resource Director, Pers. Comm.)

Peat extraction will alter the hydrophysical properties and processes within the peatland. Drainage-induced subsidence, combined with the physical compression from the harvesting machinery, modifies the pore structure of the peat, including an overall decrease in pore radius, and an increase in bulk density (Kennedy & Price, 2005; Liu et al., 2020). The resulting decreased specific yield and increased water retention at a given matric potential (Price et al., 2003) will affect pore water movement. Additionally, the increased unsaturated hydraulic conductivity will support the upward capillary movement of water in the unsaturated zone (Price & Ketcheson, 2009). Despite being categorized as a drained system, extracted peatlands can have a highly variable water table (WT), due to the decreased specific yield (Holden et al., 2011; Price et al., 2003). In addition, recent measurements show that the WTD can range by over 40 cm between years (Chapter 3). Since changes in soil water storage in the upper peat layers can become disconnected from the WT during periods when WT is deeply drawn down, WTD may not always be a useful indicator of surface moisture conditions (Price, 1997).

Changes to the peat hydrophysical properties will affect energy partitioning, and thus the E rate, at actively extracted sites. While some studies have quantified this in extracted peatlands (Petrone et al., 2004; Price, 1996), the environmental drivers remain poorly understood. The removal of vegetation should increase the solar energy received at the surface (Thompson et al., 2015), increasing the available energy for sensible and latent heat, while also removing the pathway of transpiration (T).

It is also unclear how E rates will vary with WTD. Up until a certain threshold, drainage induced changes in pore structure will create a vertically thick capillary fringe, and support higher vertical unsaturated hydraulic conductivity, allowing for maintenance of a high surface soil moisture content despite the decline in the WT (McCarter & Price, 2014; Price & Ketcheson, 2009). Drying beyond that threshold can disrupt these processes, leading to a large decline in surface soil moisture content, and a decrease in E (Alekseychik et al., 2018; Kettridge & Waddington, 2014). The limited previous research shows that extracted sites can have E rates similar to natural peatlands, despite the absence of T from plants and the deep WTs (Price, 1996; Waddington & Price, 2000). This suggests that extracted peatlands can maintain adequate surface moisture content to meet the atmospheric demand (expressed as VPD). However, it is unclear if this will be true across the wide range of WTDs observed in extracted peatlands (Price 1997; Lai, 2022). The importance of WTD on evapotranspiration (ET) rates in vegetated peatlands is mixed, with studies observing both negative effects, and little to no effect ((Lafleur et al., 2005; Scarlett et al., 2017; Sonnentag et al., 2010). Additionally, it is unclear if this WTD–E relationship will be similar across different climatic zones in Canada with very different VPDs. The absence of vegetation means that at high VPD, there will be no dampening mechanism by plant stomata to reduce ET losses (Waddington et al., 2015). In vegetated peatlands, there has been interest in quantifying how changes in VPD, vegetation cover and surface moisture conditions will drive ET, especially under a changing climate (Helbig et al., 2020; Speranskaya et al., 2024; Wu et al., 2010).

Previous modelling work (He et al., 2023) and some field-based measurements (Petrone et al., 2004; Price, 1996; Price, 1998) suggest that the water budget at extracted peatlands is dominated by E, with He et al. (2023) finding that the interior ditches are only a small source of sitewide water loss. As such, comprehensive field-based measures of E will be essential for understanding water management at these sites. Previous studies using the Priestley-Taylor approach (Price, 1998), lysimeters (Petrone et al., 2004) and the Bowen ratio-energy balance method (Price, 1996) have been conducted at small scales, with their focus to understand the impact of restoration techniques, including ditch blocking

and application of straw mulch, on E rates. To our knowledge, no study in Canada has measured energy budgets or E rates from actively extracted peatlands despite the need to quantify the effects that active management may have on E rates. By harrowing and vacuum harvesting the surface, surface peat moisture and texture conditions will vary hourly to daily. O'Kane (1992) theorized that as the harrowed layer dries, it will act as a barrier to further water loss, reducing the overall E rate as the evaporative plane moves deeper into the peat profile. This aligns with research on tillage and E in agricultural soils (Jalota and Prihar, 1990) and the two-stage conceptual model of bare soil E (Lehmann et al., 2008). However, there is a need to quantify this for an accurate understanding of the water budget at extracted sites, and to aid peat producers in effective site management.

This study thus attempts to further our understanding of bare peat E dynamics through a multi-year study at two actively extracted peatlands in two different climatic zones in Canada. The objectives were to: i) assess the relative importance of VWC, WTD and VPD on E rates at each site, ii) investigate whether the importance of VWC and VPD for predicting E rates changes with WTD, and iii) determine how harrowing impacts E rates in the first 72 hours following harrowing.

4.3 Methods

4.3.1 Study Sites

This study was conducted at two actively extracted peatlands in Canada: an eastern site (QC) near Rivière-du-Loup, Quebec (47.836° N, 69.536° W), and a western site (AB) located near Drayton Valley, Alberta (53.222° N, 114.977° W). The QC (~65 ha) and AB (~8 ha) sites have undergone active extraction since 2007 and 2009, respectively. Prior to extraction, the companies removed the surface vegetation, and dug both perimeter drainage ditches (~1 m deep, 1 m wide) around the sites, and interior ditches (~1 m deep, 0.5 m wide), spaced 30 m apart and running the length of the sites (Figure 4.1). The peat between adjacent interior ditches (termed 'fields') is extracted using the vacuum harvesting method for 15 to 40 years, depending on peat quality and depth. The harvesters mechanically harrow the peat fields to break up the top few cm of peat to aerate and cut it off hydrologically from the peat layer below.

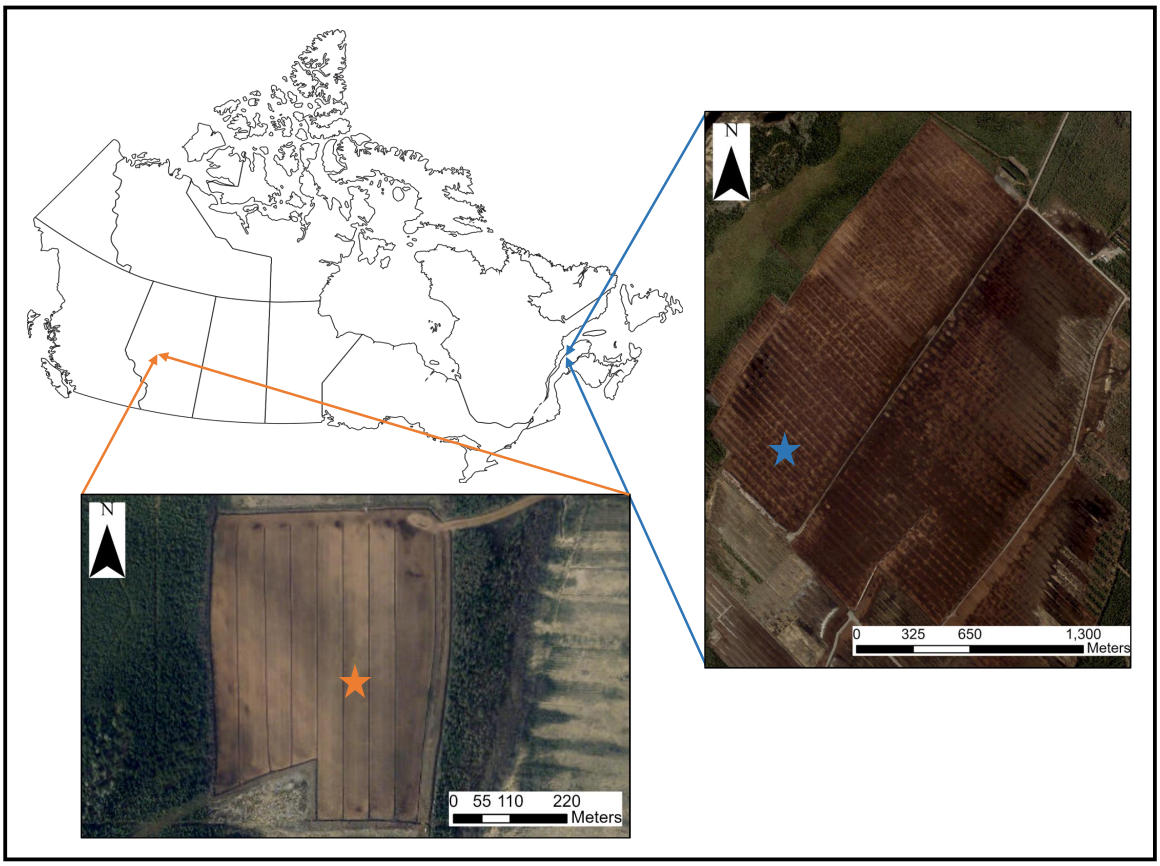


Figure 4.1 Arial images of AB (left) and QC (right) and their locations in Canada. The position of the eddy covariance towers at each site are indicated with stars. The map of Canada is by Craig Clark from Pixabay. The arial images were made using Esri (2024).

4.3.2 Environmental Measurements

A central exclusion zone where harvest machinery did not pass was created and allowed the installation of equipment to continuously measure environmental and flux variables without interfering with management operations. Half-hourly measurements of net radiation (CNR-1, Kipp and Zonen, Netherlands), and relative humidity and air temperature (HMP 45C, Vaisala, Finland) were stored on a CR3000 data logger (Campbell Scientific, Edmonton, Canada). Additionally, soil heat flux was measured using a HFP01-L (QC) and HFT3-L (AB) soil heat flux plate buried at 10 cm depth (Campbell Scientific, Edmonton, Canada) with a TVAC-L averaging soil thermocouple probe placed above it at 5 cm depth (Campbell Scientific, Edmonton, Canada) in the harvesting exclusion

zone. The inability to travel during COVID restrictions led to no viable measurements for the 2020 study period at AB, no soil heat flux measurements during 2021 at AB, and no viable measurements for the 2021 study period at QC.

We used incoming solar radiation (R_g) data from Evansburg 2 AGDM (53.61 °N, 115.06 °W; ACIS, 2023), located ~20 km northwest of AB, and air temperature data from Tomahawk AGDM (53.43 °N, 114.72 °W; ACIS, 2023), located ~11 km southwest of AB, to replace the missing 2020 data. A regression between the measured solar radiation data at AB, and Evansburg radiation data, in 2021 and 2022 was significant ($F_{1,38451}$=103305, $p<0.0001$), with an R^2 of 0.73 (Figure B1). The 15-year average daily air temperatures, and daily precipitation data during the study years, were obtained from the Tomahawk AGDM (ACIS, 2023) and the Rivière-du-Loup Climate Station (47.81° N, 69.55° W; ECCC, 2023a) for AB and QC, respectively. These stations were ~11 km southwest and ~3 km northwest of AB and QC, respectively.

We collected hourly pressure readings from buried Micro-Divers (Van Essen, Waterloo, Canada) located in the harvesting exclusion zones. Water table depth was calculated by correcting the pressure readings with an above ground Micro-Diver at QC, and atmospheric air pressure data from the Stoney Plains climate station near AB (53.56° N, 114.11° W; ECCC, 2023b). We logged hourly soil temperature at 10 cm depth (CR1000X, Campbell Scientific, Edmonton, Canada) using Type T thermocouples (Omega Engineering, Stamford, Connecticut, USA). Volumetric water content (VWC) at 20 cm depth at AB was measured using horizontally inserted CS 655 and CS 616 water content reflectometer probes (Campbell Scientific, Edmonton, Canada), though measurements were sporadic due to equipment failure and power supply issues. Top 14 cm integrated VWC at QC was obtained using CS 616 probes inserted at 45° angles.

4.3.3 Ecosystem Scale Data Collection and Processing

Flux towers were installed near the center of each site in the harvesting exclusion zone. Each consisted of an open path CO_2 and H_2O analyzer (LI-7500A, LI-COR, Nebraska, USA) and sonic anemometer (CSAT-3, Campbell Scientific, Edmonton, Canada) installed at heights of 1.8 m (QC) and 2.1 m (AB). Data sampled at 10 Hz were averaged over 30-minute intervals and stored on a USB in the Auxiliary Interface Unit (LI-7550, LI-COR, Nebraska, USA). We processed the data using EddyPro (version 7.0.9, LI-COR, Nebraska) advanced mode with the WPL correction to account for the effect of temperature and humidity on density (Webb et al., 2007) and used the Massman spectral

correction (Massman, 2000, 2001). The storage corrected fluxes were de-spiked using methods outlined by Papale et al. (2006). All sensible and latent heat fluxes with a quality flag of one and two were removed (Mauder and Foken, 2004). Additionally, year and site-specific u* thresholds were determined using the REddyProc online processing tool (Wutzler et al., 2018). The 2D Flux Footprint Prediction (FFP) online tool (Kljun et al., 2015) was used to ensure that the study sites of interest were within the 80% probability footprint contour (Figure C2, see Chapter 3, Section 3.3.3).

Energy balance closure was assessed in three ways each year at each site. First, the energy balance ratio (EBR) was calculated from

$$EBR = \frac{Q_E + Q_H}{R_n - Q_G} \quad \text{(Equation 4.1)}$$

where Q_E was the evaporation, Q_H was sensible heat, R_n was the net radiation and Q_G the ground heat flux (all in W m^{-2}). Second, we performed a linear regression, with the intercept forced through zero, of the turbulent fluxes ($Q_E + Q_H$) and available energy ($R_n - Q_G$), assessing the slope and R^2 value. Third, we calculated the residuals of the energy balance, using

$$Residuals = R_n - Q_Q - Q_E - Q_H \quad \text{(Equation 4.2)}$$

where Q_G was determined by calculating the heat flux storage (S) in the top 10 cm and adding it to the measured flux from the buried flux plate. S (W m^{-2}) was calculated using

$$S = \frac{\Delta T_s C_s d}{t} \quad \text{(Equation 4.3)}$$

where ΔT_s was the change in soil temperature (K), C_s was the moist peat heat capacity (J m^{-3} K^{-1}), d was the depth of the heat flux plate (m), and t was the number of seconds in a 30-minute period (1800). Moist peat heat capacity was estimated for each half hour period using

$$C_s = \rho_b C_d + \theta_v \rho_w C_w \quad \text{(Equation 4.4)}$$

where ρ_b was the peat dry bulk density (kg m^{-3}, Lai, 2022), C_d was the specific heat capacity of dry peat (J kg^{-1} K^{-1}), θ_v was the volumetric water content, ρ_w was the density of water (kg m^{-3}), and C_w was the specific heat capacity of water (J kg^{-1} K^{-1}). Results of the energy balance closure analysis are provided in Table C1.

The evaporative fraction (EF) was calculated using

$$EF = \frac{Q_E}{Q_E + Q_H} \quad \text{(Equation 4.5)}$$

where, due to limited measurements of Q_G, for this analysis, the sum Q_E and Q_H was used to represent the available energy, as has been done previously (e.g., Speranskaya et al., 2024).

Potential evaporation (PE in mm d^{-1}) was estimated at QC in 2020 and 2022 and at AB in 2022 using half-hourly values by setting the surface resistance to zero in the Penman-Monteith equation as

$$PE = \frac{\Delta(R_n - Q_G) + \rho_a c_p \frac{\mathrm{VPD}}{r_a}}{L_v(\Delta + \gamma)} \times 1000 \quad \text{(Equation 4.6)}$$

where R_n and Q_G were in MJ m^{-2} d^{-1}, ρ_a was the air density (kg m^{-3}), c_p was the specific heat of air (MJ kg^{-1} $°C^{-1}$), VPD was the vapour pressure deficit (kPa), Δ was the slope of the saturation vapour pressure curve (kPa $°C^{-1}$), γ was the psychrometric constant (kPa $°C^{-1}$) and L_v was the latent heat of vaporization (MJ m^{-3}). The aerodynamic resistance (r_a in d m^{-1}) was found using

$$\mathrm{r_a} = \frac{u}{{u_*}^2} - \frac{kB^{-1}}{k u_*} \quad \text{(Equation 4.7)}$$

where u was the wind speed (m/s), u_* was the friction velocity (m/s), kB^{-1} was a unitless parameter, and k was von Karman's constant (=0.41). We assumed a kB^{-1} parameter of -0.9 for bare soil (Verhoef et al., 1996).

4.3.4 Plot Scale Evaporation

Ten weighing bucket lysimeters were installed at the AB site, flush with the surrounding peat surface, in the same harvesting exclusion zone as the tower. Peat cores (31 cm diameter, ~ 30 cm tall) were collected in plastic rings, the bottoms were covered in plastic mesh and then the cores were placed in plastic buckets with at least 10 cm between the peat and the bucket bottom to allow for vertical drainage and avoid oversaturation. We conducted three 72-hour lysimeter experiments in 2022, running from July 20th and 23rd, July 26th and 29th, and August 6th and 9th, to evaluate the impact of harrowing on E. The absence of local precipitation during each experimental period was verified with a manual rain gauge. At the start of each round, we applied a harrowing treatment to five of the lysimeters by raking the top few cm of peat. The other five lysimeters were left untouched as a control. The buckets were weighed using a portable battery power scale (Model H-1651, ULINE, Wisconsin, USA) to track water loss. We made five measurements of the bucket weight during the first experiment. For each of the next two experiments, the harrowing treatment lysimeters were re-

harrowed. We added additional measurements in the next two 72-hour rounds, taking six and seven measurements during the second and third rounds, respectively.

4.3.5 Effect of Harrowing on Peat Moisture Content

We collected 24 peat grab samples of known volume in August 2021 at AB to understand the effect of harrowing on peat VWC. We sampled both the harrowed layer, and the layer of peat directly below. The harrowed layer had been present for at least half a day prior to data collection, and samples were collected on three different days. The peat was weighed, dried at 80 °C for 48 hours, and then weighed again to calculate the VWC.

4.3.6 Statistical Analysis

All statistical analysis, and the creation of plots, were performed in Rstudio using the ggplot2 (Wickham et al., 2016) and dplyr (Wickham et al., 2023) packages. Only daytime ($R_n > 20$ W m^{-2}) data was used for analysis in this study. Due to the highly stable conditions overnight, few half-hourly nighttime periods remained after our strict quality control procedures. We thus decided to compute monthly averages of the data. There were an unequal number of fluxes between half-hourly periods each day, with a higher proportion of fluxes during the middle of the day. To estimate monthly fluxes, we first took averages of E, Q_E and Q_H for each hour-hourly period each month. These were then all averaged monthly to obtain a monthly average half-hourly E, Q_E and Q_H. There were multiple half-hourly daytime periods, mainly before 7 am and after 8 pm, where no E, Q_H or Q_E measurements passed our quality control procedures during a given month. If a time period in a given month was missing data, we used data from the other months for the same time period by taking the ratio of the flux value for that period to the flux value at noon and then applying that ratio accordingly to replace the missing value. A series of two-way analysis of variance (ANOVA) tests were performed to determine the effect of the interaction of VPD and WTD on ecosystem scale E rates. Linear and non-linear regressions were performed to determine the effect of VPD, WTD, wind speed and VWC on ecosystem scale E rates, and the relationship between potential and actual E. One-way ANOVAs were performed, using linear mixed effects (LME) models with bucket identity as a random effect to account for the repeated measurements of lysimeter buckets and meet the assumption of independent samples, to assess the effect of harrowing on E loss from the lysimeters.

4.4 Results

4.4.1 Meteorological Conditions

The average QC study period (May to October) air temperature was within 0.2 °C of the 15-year average of 13.2 °C, with average monthly temperatures peaking at 19.1 and 18.7 °C in July 2020 and 2022, respectively (Figure 4.2; ECCC, 2023a). Average monthly temperatures varied by only a few degrees between years. At AB, average study period (March to October) air temperature was higher in 2021 (10.3 °C) and 2022 (9.9 °C), and lower in 2020 (7.6 °C) than the 15-year average of 9.1 °C (ACIS, 2023). Peak average monthly temperatures occurred in July in 2020 (16 °C) and 2021 (18.6 °C), while it peaked in August in 2022 (18.3 °C). Temperatures rose above freezing earliest in 2021, with average temperatures of -7.3 °C in March 2020 compared to 1.4 °C in March 2021.

March to October total precipitation was just above 600 mm in 2020 and 2022 at QC, very similar to the 15-year average of 596 mm (Figure 4.2, ECCC, 2023a). Interannually, the largest precipitation events occurred during September and October in 2020, compared to in March and May in 2022. Monthly average WTD ranged from 48 to 89 cm below the surface, with few periods when the WTD was less than 30 cm. Total precipitation was lower at AB than QC in all three study years. There was considerable interannual variability, with precipitation sums of 482, 341 and 350 mm in 2020, 2021 and 2022 respectively, compared to the 15-year average of 377 mm (ACIS, 2023). Like at QC, the timing of precipitation was different between years, with 27% of study period precipitation occurring during June and July in 2021, compared to 65% in 2022. The WTD was smallest in 2020, with frequent fluctuations, and the WT was in the top 40 cm of the peat for most of the study period. Water table depth patterns were similar in 2022, except for a dry period in May and June 2022. Despite some large precipitation events in 2021, the WT position was steady and ranged from ~ 70 cm to 60 cm below the surface from April to August, suggesting a disconnect between moisture content at the surface and the WT.

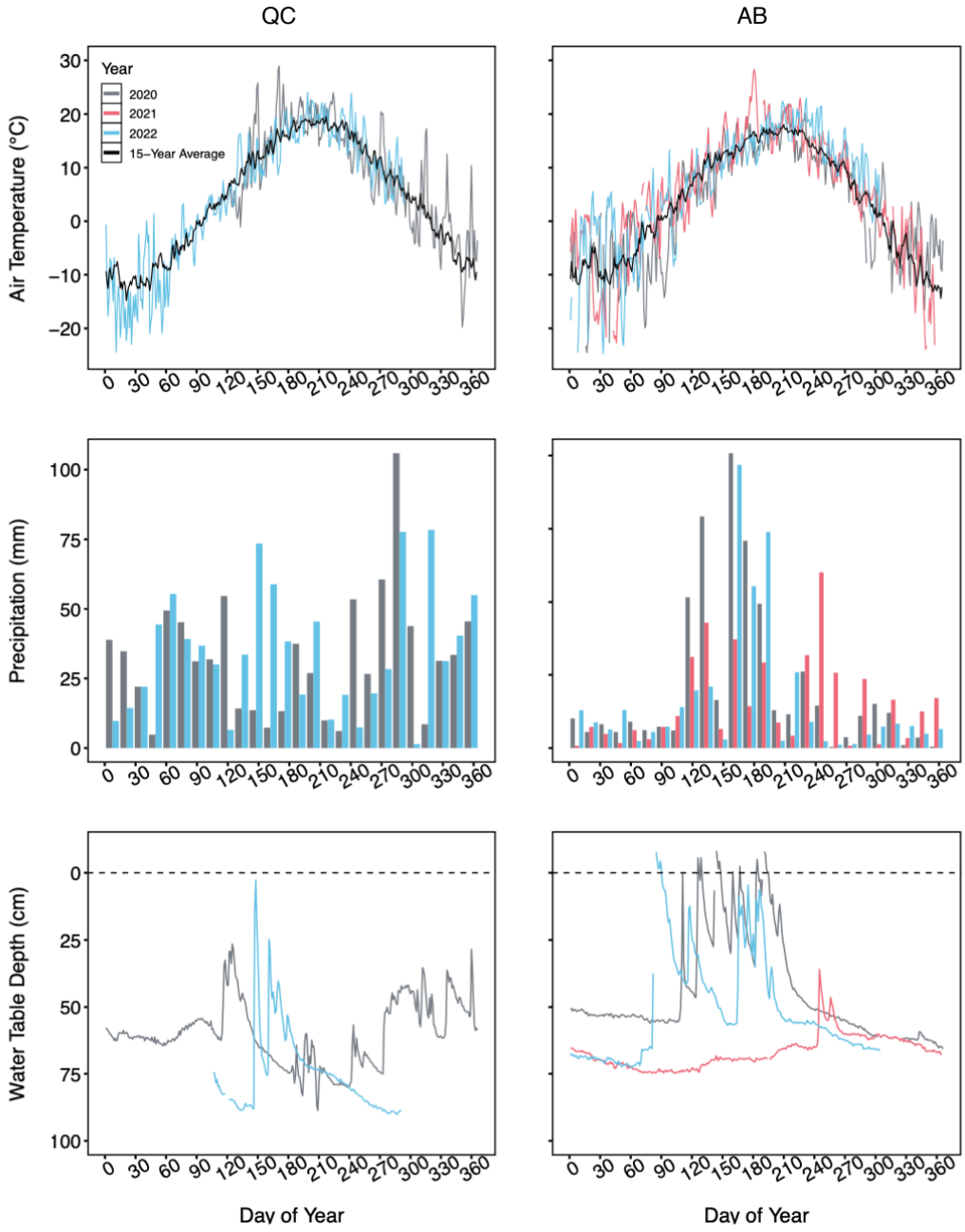


Figure 4.2 Daily air temperature (°C), bi-weekly precipitation (mm), and daily WTD (cm) at QC (left) and AB (right) during the study years. Water table depth was measured near the eddy covariance tower in the harvesting exclusion zone (see Figure 4.1), with a positive value indicating that the WT was below the peat surface. The 15-year average air temperature (2008 to 2022) at AB and QC is shown in black. Daily air temperature in 2020, 15-year average daily air temperature, and bi-weekly precipitation data for AB was obtained from the Tomahawk AGDM weather station (ACIS, 2023). The 15-year average daily air temperature, and bi-weekly precipitation data for QC was obtained from the Rivière-du-Loup climate station (ECCC, 2023a).

4.4.2 Energy Balance

Daytime (R_g > 20 W m^{-2}) Q_E and Q_H fluxes followed the diurnal pattern of incoming solar radiation, peaking between 11 am and 2 pm each day. During the early morning and late evening, we observed small positive Q_E , and small negative Q_H. Latent heat flux dominated the total available energy during daytime periods, with the evaporative fraction less than 0.5 only occurring in October 2022 at AB (Table 4.1). The partitioning of this daytime energy into Q_H and Q_E, and the magnitude of these fluxes, varied monthly and annually (Figure 4.3, Table 4.1).

At AB, and at QC in 2022, daytime Q_E displayed a seasonal pattern, peaking in June and July, though the amplitude of this pattern was greatest in 2022 at AB (Figure 4.3). In contrast, Q_E were similar across the study months at QC in 2020 and peaked in September. Sensible heat fluxes did not vary as much monthly or between years at each site, though there was a general pattern of fluxes increasing throughout the summer. At AB, evaporative fractions ranged between 0.47 and 0.87, and were on average highest in 2020 and lowest in 2021 (Table 4.1). E fractions at QC ranged from 0.66 to 0.90. They were highest in July and August in 2022 compared to in 2021. However, this pattern was reversed in September and October, with higher evaporative fractions in 2020 compared to 2022. Average daytime Q_G at both sites was similar between months, and were always less than Q_H, except for in October in 2020 at QC. Values ranged from 20 to 30 W m^{-2} and 30 to 40 W m^{-2} at QC and AB, respectively (Figure 4.3).

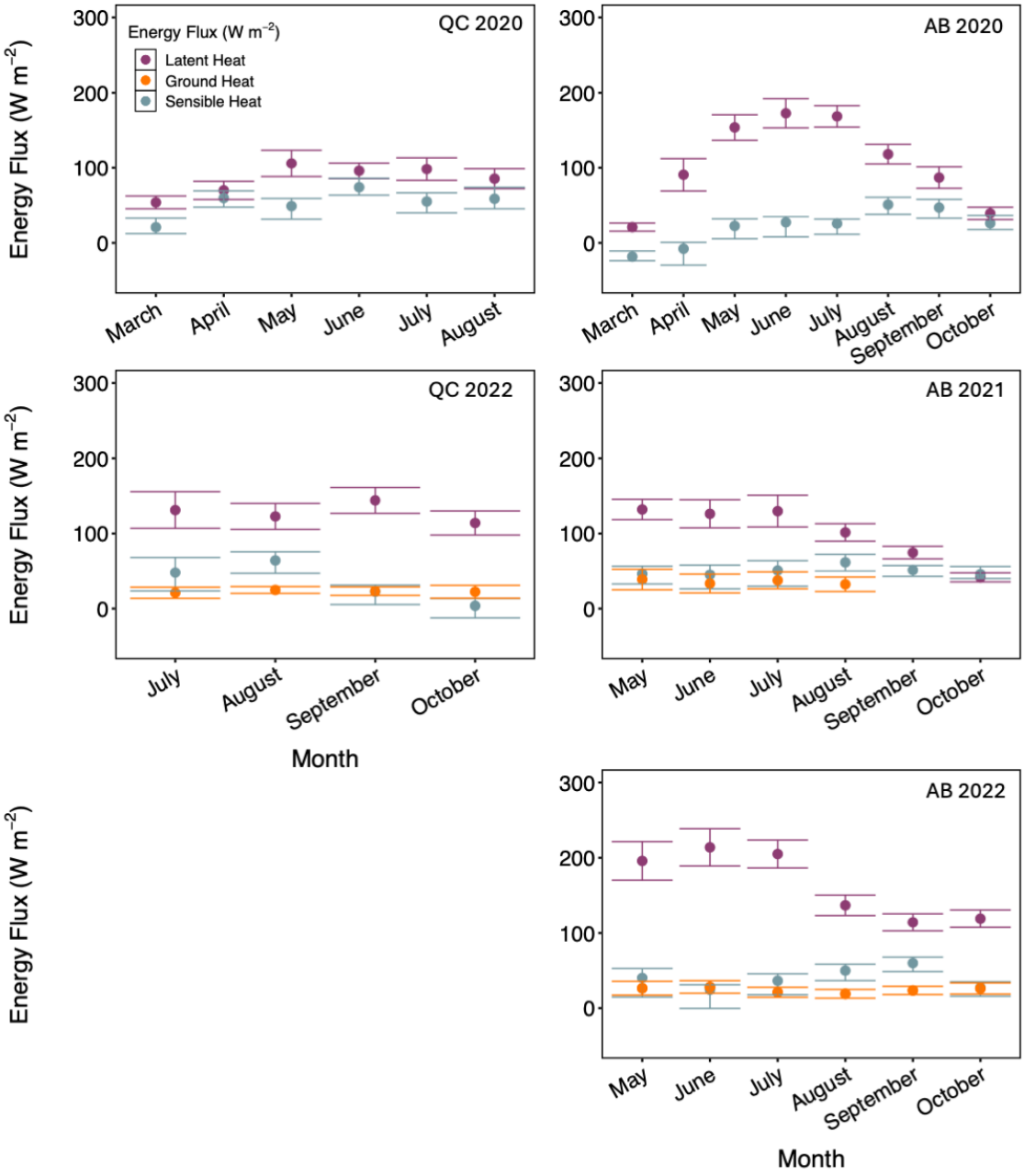


Figure 4.3 Monthly average daytime latent heat (Q_E), sensible heat (Q_H) and ground heat (Q_G) flux across the study years at QC and AB during each study year. Error bars represent the standard error.

Table 4.1 Monthly average daytime (R_g > 20 W m^{-2}) evaporative fractions at AB and QC in 2020, 2021 and 2022. A value of n. m. indicates that measurements were not made that month.

	AB			QC	
	2020	2021	2022	2020	2022
March	0.53	0.72	n. m	n. m	n. m
April	0.92	0.54	n. m	n. m	n. m
May	0.87	0.68	0.74	n. m	0.83
June	0.86	0.56	0.74	n. m	0.90
July	0.87	0.64	0.72	0.73	0.85
August	0.70	0.59	0.62	0.66	0.73
September	0.65	n. m	0.59	0.86	0.66
October	0.60	n. m	0.47	0.97	0.81

4.4.3 Sitewide Evaporation

Average monthly daytime (R_g > 20 W m^{-2}) E ranged from 0.03 to 0.28 mm hr^{-1} and 0.16 to 0.31 mm hr^{-1} at AB and QC, respectively, with the lower limit at AB due to the inclusion of March and April E rates (Table 4.2). E generally followed the precipitation and WTD patterns. At AB, E rates were lowest in early spring and fall, with limited interannual variability during those months. Peak E occurred between May and July each year, though the magnitude of E was different, with the highest rates on average occurring in 2020, and the lowest in 2021. This difference could be large, with double the E rate in 2020 compared to 2021. At QC, both the timing and magnitude of peak E rates differed between years (Table 4.2). Peak E rates of 0.31 mm hr^{-1} occurred in July in 2022, compared to 0.21 mm hr^{-1} in September 2020. E rates were on average higher at QC than AB, with the largest differences occurring in September and October.

Table 4.2 Average monthly daytime (R_g > 20 W m^{-2}) evaporation (mm/hr) at AB and QC in 2020, 2021 and 2022. A value of n. m. indicates that measurements were not made that month.

	AB			QC	
	2020	2021	2022	2020	2022
March	0.03	0.08	n. m.	n. m.	n. m.
April	0.14	0.10	n. m.	n. m.	n. m.
May	0.23	0.16	0.20	n. m.	0.27
June	0.28	0.14	0.19	n. m.	0.30
July	0.28	0.16	0.22	0.20	0.31
August	0.17	0.13	0.15	0.18	0.20
September	0.13	n. m.	0.11	0.21	0.16
October	0.06	n. m.	0.06	0.17	0.18

4.4.4 Surface and Atmospheric Controls on Evaporation

Surface moisture conditions were an important control on E rates. There was a significant positive relationship between 20 cm depth VWC and E rates for periods with incoming solar radiation > 500 W m^{-2} at AB in 2021 and 2022 ($F_{1, 161}$=17.41, $p < 0.0001$) across a range of WTDs (Figure 4.4). A threshold of 500 W m^{-2} was chosen to exclude periods when incoming radiation was limiting. We did not have VWC measurements in 2020. Volumetric water content explained 49% of the variation in E. The effect size was large, with E rates averaging ~ 0.15 mm hr^{-1} at VWCs of 40% compared to ~0.40 mm hr^{-1} at VWCs of 80%. At QC, top 14 cm integrated VWC was also significant ($F_{1, 1242}$=628.6, $p<0.0001$) explaining 34% of the variation in E. However, there was a large range of E rates when the VWC was above 30% (Figure 4.4).

It is difficult to obtain continuous surface VWC at these sites, as equipment cannot be left on the actively extracted portions of the sites. We wanted to know if WTD was an appropriate proxy for VWC to understand E rates. We found that the strength and direction of the effect of WTD on E rates during sunny periods (R_g > 500 W m^{-2}) varied with WTD (Figure 4.5). AT AB, when the WT was

within the top 40 cm, there was no significant linear relationship with E rates ($F_{1, 603}$=0.98, p=0.32) and a large range in E from around 0.2 to 0.7 mm hr^{-1}. When the WT was deeper than 40 cm, we observed a significant negative relationship ($F_{1, 2006}$=173, p<0.0001), though it had low explanatory power (R^2=0.08) and E ranged from around 0 to 0.6 mm hr^{-1}. We observed a similar overall pattern at QC in 2020 and 2022, though the threshold occurred at around 60 cm WTD (Figure 4.5). There was a significant negative relationship ($F_{1, 1142}$ =167, p<0.0001), explaining 12% of the variation in E when the WTD was greater than 60 cm. There was no effect of WTD on E rates when the WTD was less than this threshold ($F_{1, 198}$ = 1.22, p=0.27).

The importance of the atmospheric demand for water vapour (VPD) for E rates varied with WTD. There was a significant effect of the interaction between VPD and the WTD on E rates during periods with R_g > 500 W m^{-2} at both sites (AB: $F_{3,2608}$=488.1, p<0.0001; QC: $F_{3,761}$=72.39, p<0.0001). We thus divided the WT data into five levels: 0–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80 plus cm below the surface and fit rectangular hyperbola relationships of the form

$$y = \frac{a \cdot b \cdot VPD}{a \cdot VPD + b} \qquad \text{(Equation 4.8)}$$

to understand the impact of VPD on E rates. The shape of this relationship assumes that near saturation, increases in VPD will result in increasing E but that a maximum rate is reached at higher VPD due to limitations in the rate of liquid and vapour water flux through the soil. At AB, there was a significant positive effect of VPD on half-hourly E at each factor level (Figure 4.6, Table C2). However, the degree of scatter increased, and the explanatory power decreased as WTD increased from 0–20 cm to 60–80 cm below the surface. Additionally, *b*, the parameter representing the maximum E rate, decreased from 0.719 to 0.240 mm hr^{-1} as WTD increased. At QC, the WT was rarely in the top 40 cm of the peat, making it difficult to assess the effect of VPD on E during these periods (Figure 4.6). When the WT was deeper than 40 cm, VPDs were associated with a wide range of E rates, and the rectangular hyperbola relationship explained less than 6% of the variation in E (Table C2). Wind speed was not an important control on E rates at either site. While significant, the positive linear relationship between wind speed and sunny period E rates (Figure C2) had an explanatory power of less than 1.5% of the variation in E (*AB*: $F_{1, 2630}$=51.76, p<0.0001; *QC*: $F_{1, 1048}$=10.57, p<0.0001).

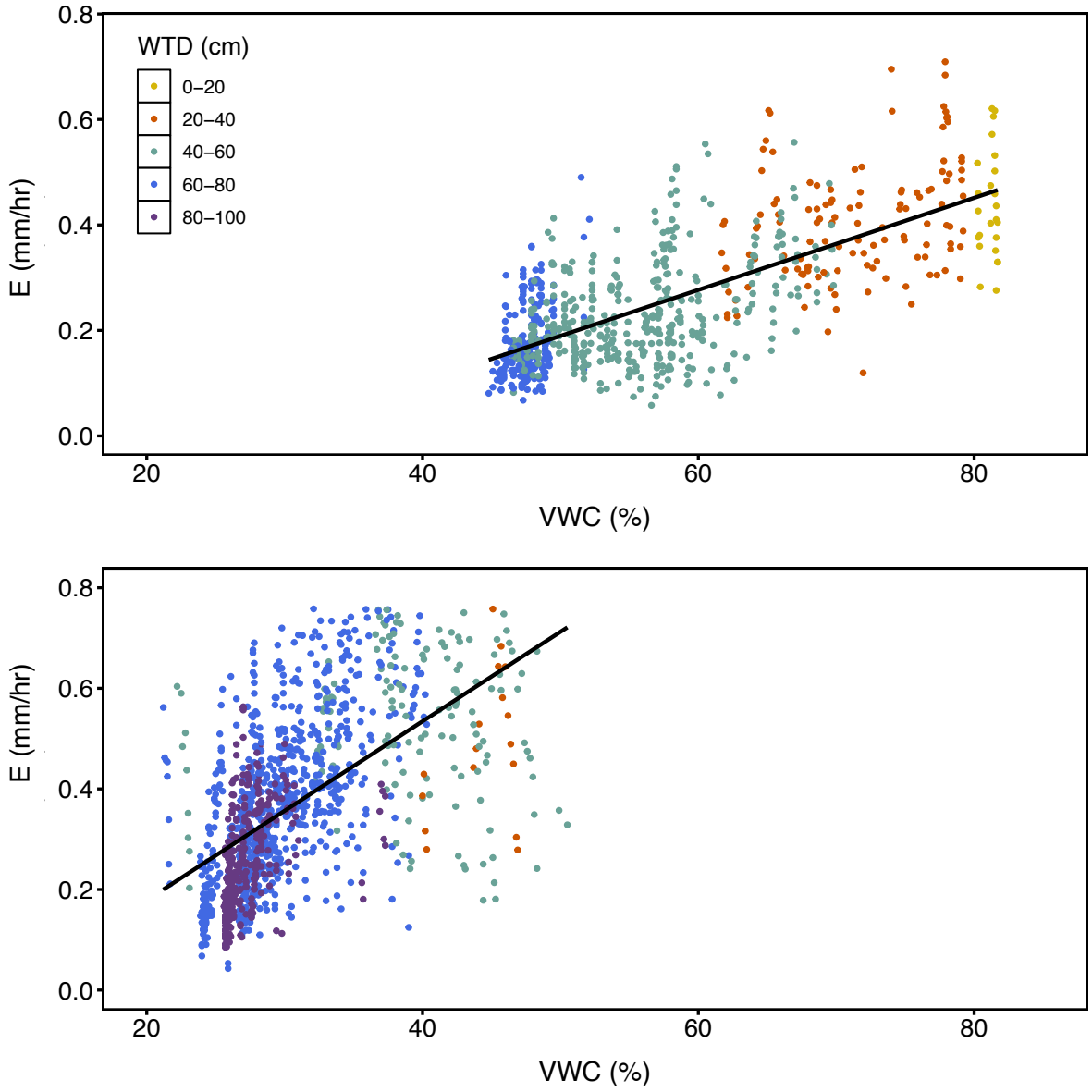


Figure 4.4 Relationship between 20 cm depth (at AB) and top 14 cm integrated (at QC) volumetric water content (VWC) and evaporation at AB (top) and QC (bottom). Only evaporation values when the incoming shortwave radiation was above 500 W m^{-2} were included. Scatter point colours indicate the corresponding WTD.

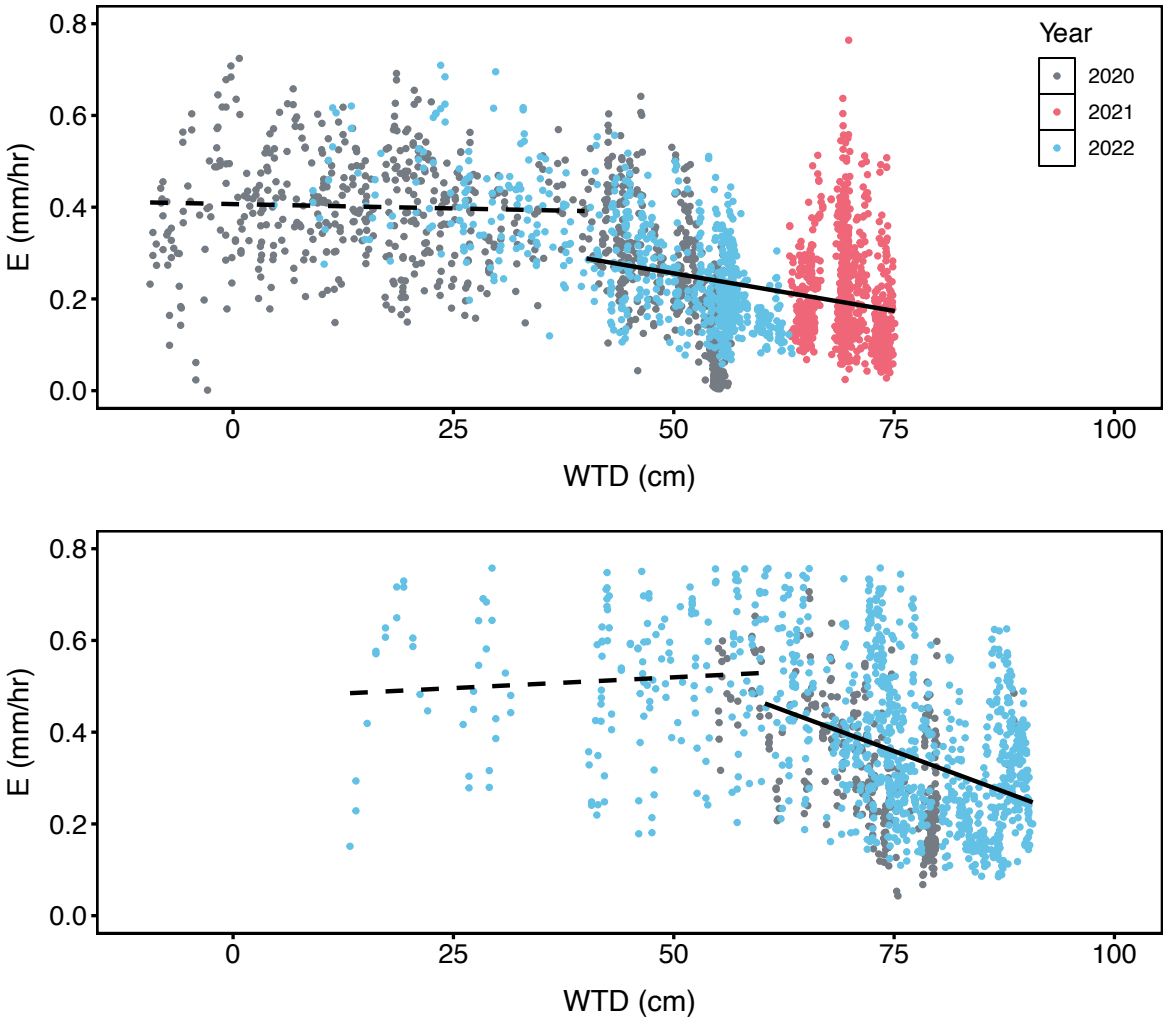


Figure 4.5 Relationship between WTD and average hourly evaporation rates each year at AB (top) and QC (bottom). Only periods when R_g was above 500 W m^{-2} were included. Best fit linear lines show the relationship when WTD is greater and less than 40 cm at AB, and greater and less than 60 cm at QC. A dashed line indicates a non-significant relationship.

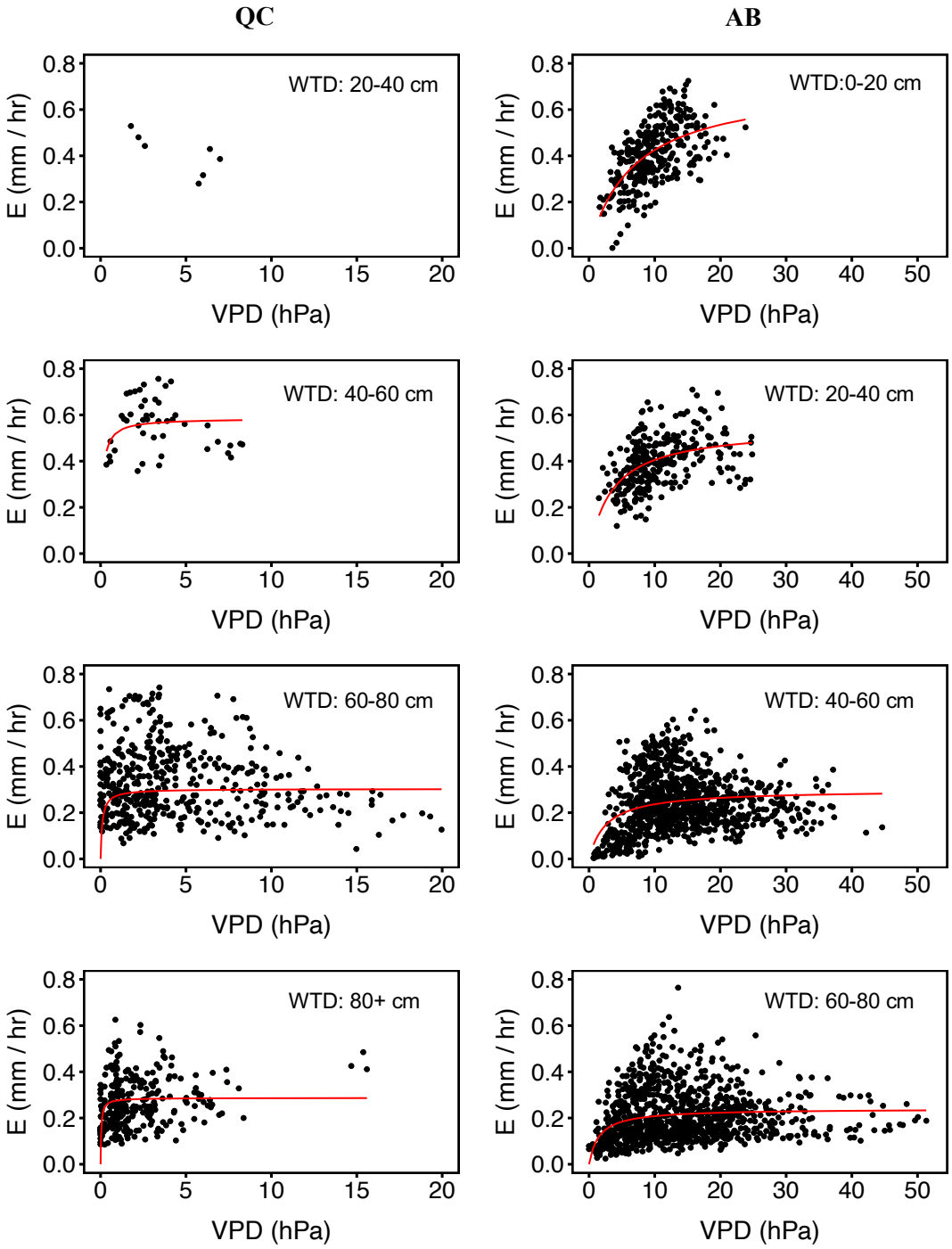


Figure 4.6 Relationship between VPD and E at QC (left) and AB (right) during periods with the indicated WTD range (cm). Only evaporation rates from periods when R_g was above 500 W m^{-2} were included. Best fit curves using a rectangular hyperbola relationship are shown. See Table C2 for information on model parameters and fit results.

4.4.5 Impact of Precipitation on Evaporation

We observed an increase in half-hourly E rates following precipitation events, despite limited change in WTD. This is illustrated by an example from August 16th to 21st, 2020 at QC (Figure 4.7). There was no precipitation during the previous 10 days. August 16th and 17th were clear sunny days, and E rates peaked below 0.3 mm hr^{-1}. August 18th and 19th were cloudy, with a two-day total precipitation of 6.1 mm. The following two days had comparable incoming shortwave radiation to the pre-precipitation days, but had observably higher E rates, peaking between 0.4 and 0.5 mm hr^{-1}. This coincided with a less than 3 cm change in WTD across this period.

4.4.6 Potential Evaporation

Half-hourly actual E rates were lower than potential rates, and the strength and slope of the relationship depended on the WTD (Figure 4.8). There was a significant effect of the interaction between half-hourly potential E and WTD level (*AB*: <40 cm and > 40 cm; *QC*: <60 cm and > 60 cm) on actual E at QC ($F_{1, 1841}$=89.0, p<0.0001) and AB ($F_{1, 719}$=88.3, p<0.0001). Linear regressions were significant at each site at both WTD levels (Table C3). The slopes between the two WT conditions were significantly different (AB: t-ratio=9.4, p<0.0001; *QC*: t-ratio =9.43, p<0.0001), with actual E rates closer to potential when the WTD was closer to the surface as expected. The strength of the relationship also varied with WTD, as potential E explained at least 70% of the variation when the WTD was above the thresholds, and less than 45% of the variation when the WTD was deeper than the thresholds (Figure 4.8, Table C3). Additionally, the regressions at AB and QC were significantly different than the 1:1 line (Figure 3.8, Table C4).

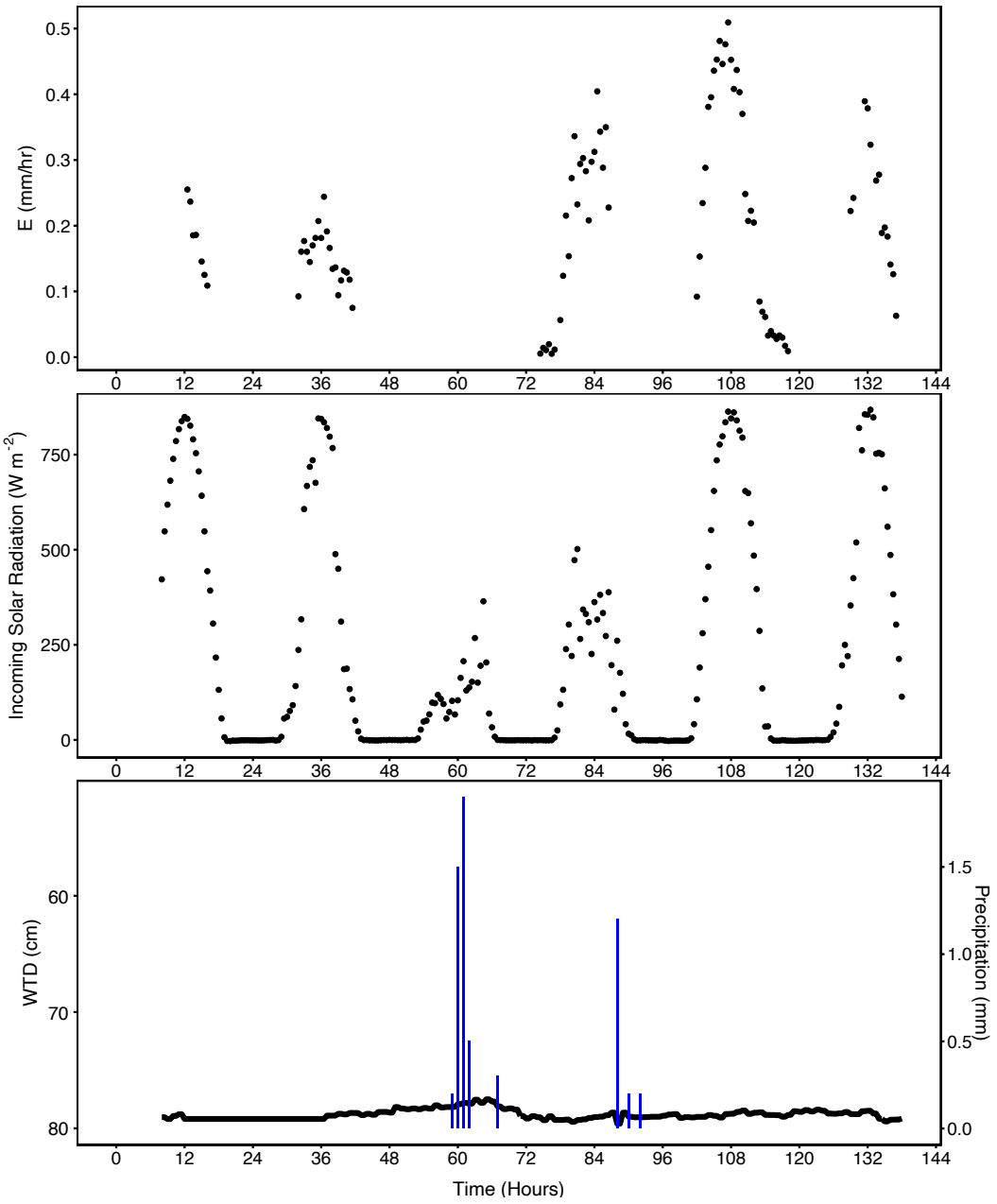


Figure 4.7 Half-hourly evaporation rates, half-hourly incoming solar radiation, hourly WTD, and hourly precipitation (on secondary axis) from a 6-day period from August 16th to 21st, 2021 at QC.

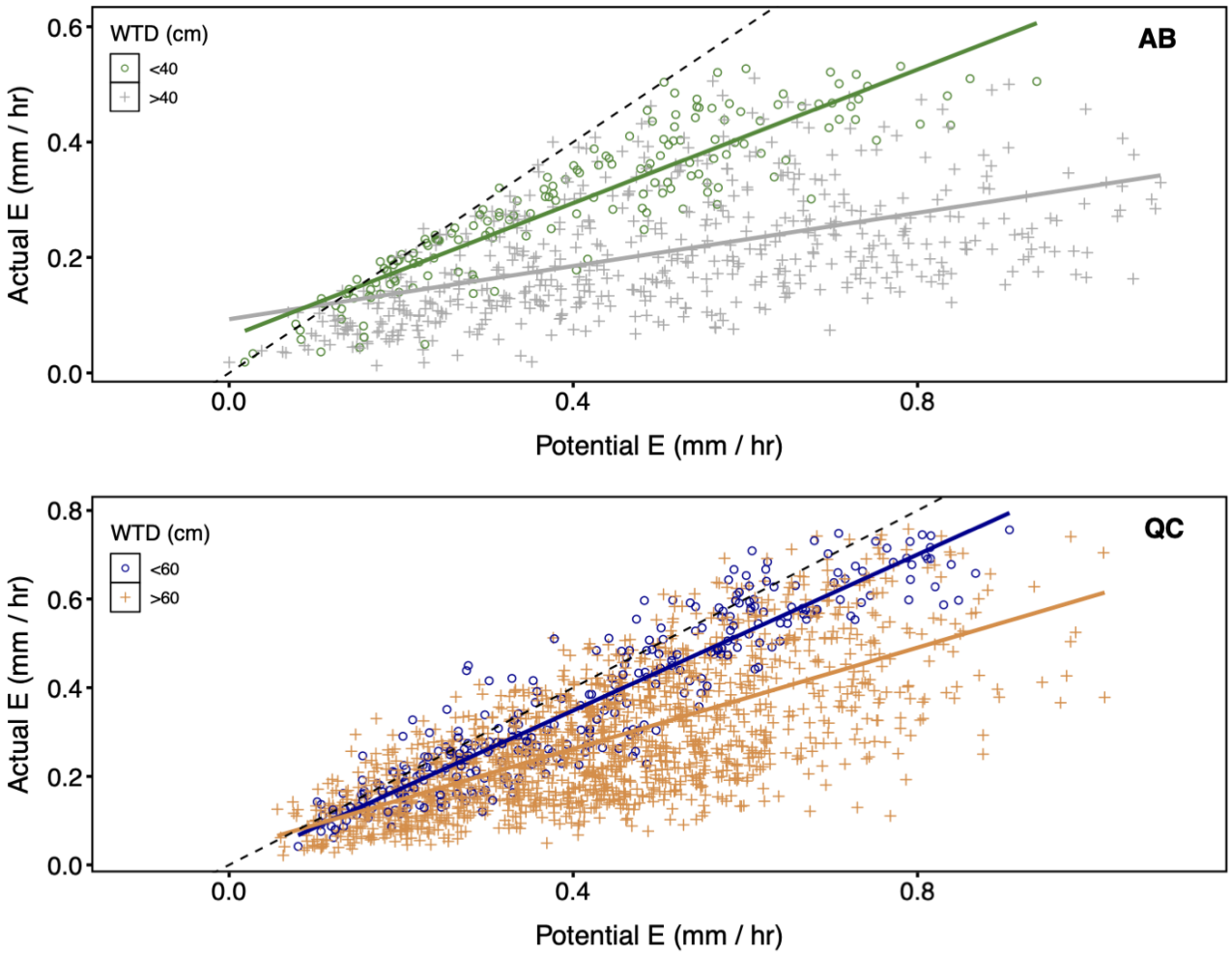


Figure 4.8 Relationship between half-hourly potential and actual E rates at AB in 2022 (top), and QC in 2020 and 2022 (bottom). The dashed line represents the 1:1 line. At AB, regressions explained 77% and 29% of the variation in actual E when the WTD was less than and greater then 40 cm, respectively. At QC, regressions explained 88% and 43% of the variation in actual E when the WTD was less than and greater then 60 cm, respectively (Table C3).

4.4.7 Effect of Harrowing on Evaporation

Gravimetric sampling from AB in 2021 confirmed that harrowing reduces surface peat VWC.

Harrowing is applied to dry out surface peat to facilitate vacuum-harvesting, but measuring changes

in VWC accurately in this thin surface layer using moisture probes is not possible given high porosity and exposure at the surface. The ~3 cm thick harrowed layer had an average VWC of 8% ± 6%, while the layer directly below had an average VWC of 40% ± 8%.

The lysimeter experiment demonstrated that the effect of harrowing on E changes over time. The three experiment rounds were conducted during sunny, precipitation free periods, with average midday (10 am to 2 pm) incoming solar radiation between 715 and 745 Wm^{-2}. During each round, there was a significant effect of time since harrowing (24, 48 and 72 hours), and the interaction between time since harrowing and treatment, on cumulative E (Figure 4.9, Table C5). After 24 hours, the non-harrowed treatment evaporated between 0.14 and 0.53 mm more water than the harrowed treatment. By 48 hours, the non-harrowed treatment had evaporated between 1 and 1.4 mm more water than the harrowed treatment, though these differences were not statistically significant (Figure 4.9, Table C5). However, after 72 hours, the cumulative E differences between the treatments were significant for rounds 2 and 3, with the non-harrowed treatment cumulatively evaporating at least 67% more water than the harrowed treatments (Figure 4.9, Table C5).

Multiple measurements were made during the first 24 hours following harrowing to understand the initial effects of harrowing on E (Figure 4.9). We observed similar patterns during rounds 1 and 3, where the harrowed treatment had a higher E rate in the first ~4 hours, but after that, the non-harrowed treatment had a higher E rate. During round 2, the first measurement was made ~5.5 hours after harrowing, and by this point, the non-harrowed treatment had evaporated 0.1 mm more water than the harrowed (Figure 4.9).

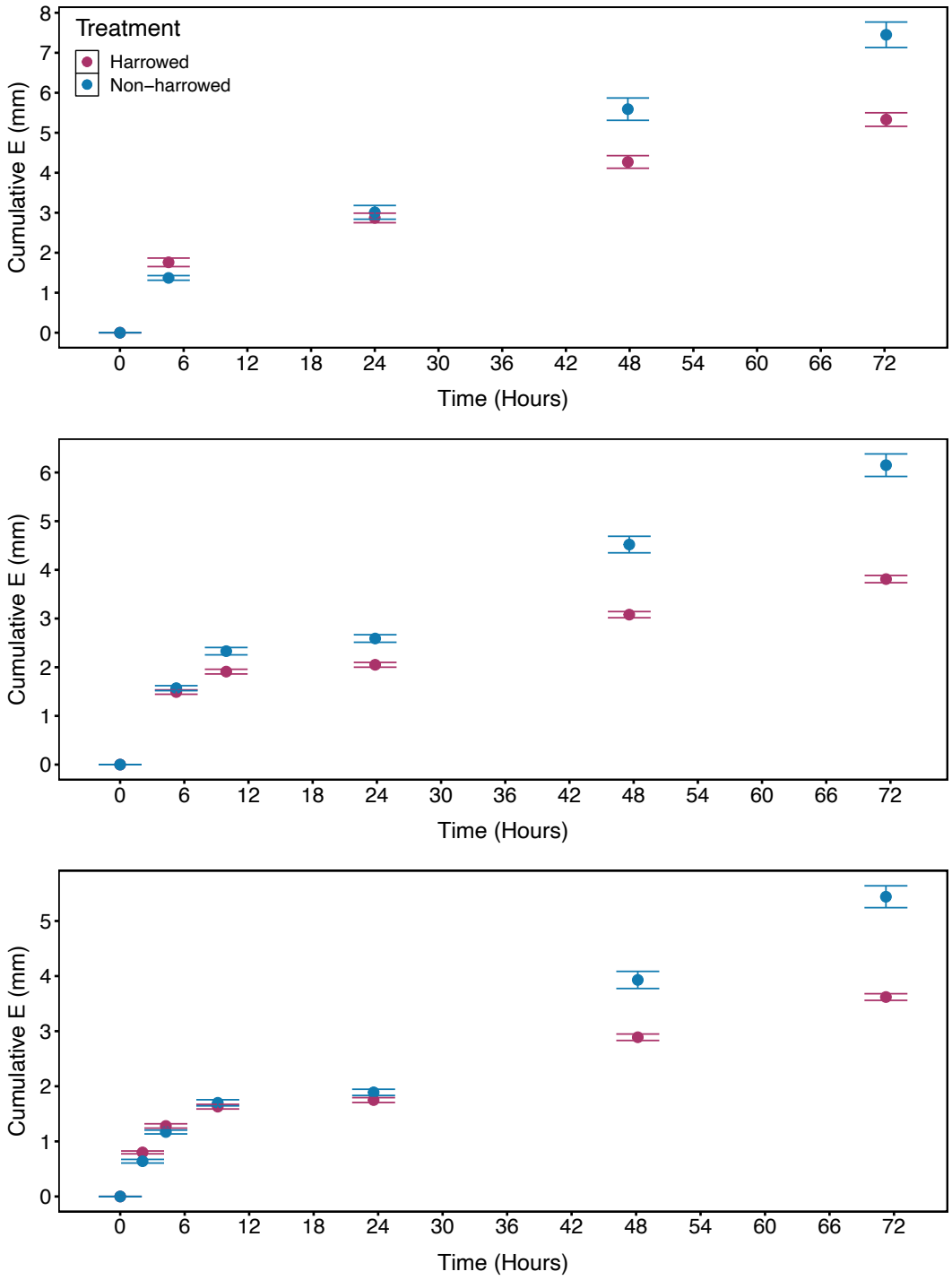


Figure 4.9 Cumulative evaporation (mm) from the harrowed and non-harrowed lysimeter bucket treatments over 72 hours during round 1 (top), round 2 (middle) and round 3 (bottom) at AB in 2022. The error bars represent the standard error.

4.5 Discussion

This was the first study in Canada to measure the energy balance and E rates at actively extracted peatlands. Our results confirm previous research that the available energy is largely partitioned into Q_E in actively extracted peatlands, despite being an actively drained site (Petrone et al., 2004; Price, 1996; Price et al., 1998; Waddington & Price, 2000). This finding was true across WTDs ranging from 0–80 plus cm below the surface, with the only evaporative fraction of less than 0.5 occurring in October 2022 at AB. The pattern in E fraction generally followed patterns in WTD, with the highest fractions coinciding with the shallowest WTs, highlighting the importance of moisture conditions on E rates (see Section 4.5.1.1). Ground heat flux was between 9 and 15% of R_n, in agreement with previous studies suggesting that Q_G is a small component of the sitewide energy balance (Price, 1996).

Evaporation rates were similar between sites, and seasonal patterns follow the observed WTD. The large interannual variability, with close to double the daytime E rate in June and July 2020 compared to 2021, highlights the importance of hydrologic conditions on E rates (see Section 4.3.1). When we summed the average half-hourly daytime (R_g> 20 W m^{-2}) E rates each month, average daytime May to August E was 2.97 and 3.60 mm at AB and QC, respectively (Table C6). This is a slight underestimation of daily E rates, as non-daytime periods will have low, but not zero, E. These rates are comparable to previous measurements at post-extracted unrestored peatlands. Price (1996) observed an evaporation rate of 2.9 mm d^{-1}, while Petrone et al. (2004) measured seasonal (May 17th to October 11th) evaporation of 2.7 mm d^{-1} and 3.4 mm d^{-1} using lysimeters during 2000 and 2001, respectively. Price (1998) estimated May to August evaporation using the Priestley-Taylor method of 3.1 mm d^{-1}. Unlike in our study, these studies did not include E from drainage ditches, which, when full of water, would likely evaporate at potential evaporation (PE) rates. Despite being a drained system with no T, previous studies have suggested that extracted peatland E will be similar to ET from boreal bogs, due to the efficient capillary movement of water (Price, 1996; Waddington & Price, 2000). Our results support this, with our E rates at the lower end of other boreal bogs (Moore et al., 2013; Wu et al., 2010). Given the lack of vegetation on these extracted peatland sites, they provide a unique opportunity to investigate environmental controls on E from peat soils.

4.5.1 Environmental Controls on Evaporation

4.5.1.1 Threshold Effect of WTD on Evaporation Rates

Surface moisture conditions, and thus the availability of water for E, was the most important control on E rates, and this was true across the observed range of WTD. This explanatory power of VWC is large considering that these sensors were in a harvesting exclusion zone. The peat above the sensors did not undergo harrowing and harvesting processes that would alter the pore structure and hydrologic properties of the peat, making them not true representations of surface conditions (see Section 4.5.4 for discussion of the effects of harrowing on E).

Water table depth, a common proxy for surface moisture content, was not as strong of a predictor of E rates as was VWC. However, the observed WTD threshold provides information on water movement within the peat profile, and the relative importance of surface and atmospheric conditions on E rates. Average evaporation rates were largest when the WTD was less than thresholds of 40 cm and 60 cm at AB and QC, respectively. During these periods, there was no significant relationship between WTD and E, suggesting the upward capillary transport of water was able to maintain sufficient surface moisture conditions to meet the atmospheric demand (McCarter & Price, 2012; Price & Ketcheson, 2009). When WTD was deeper than this threshold, we observed an overall decrease in E rates, suggesting potential for reduced surface moisture conditions that cannot meet the atmospheric demand. This is part of a well-documented negative feedback in peatlands, where, when upward capillary water movement is not able to replace the evaporated water from near surface storage, the surface layer dries out and surface resistance increases, thereby decreasing E (Kettridge & Waddington, 2014; Price & Whittington, 2010). The importance of the deeper threshold at QC compared to AB is discussed in Section 4.5.3.

However, the large range of E rates we observed at these deep WTs suggests that high surface moisture contents can occur alongside a deep WT. Our observed increase in E rates following rain events, despite similar R_g and WTD pre- and post-rain events, indicates that the importance of precipitation and the resulting surface water storage changes for E rates in these drained systems. During periods of deep WT, E rates may be more influenced by rates of atmospheric water input than by the upward capillary movement of water from below. In the absence of reliable surface moisture measurements, care needs to be taken when using WTD to predict E rates when the WTD is greater than our observed thresholds.

Our observed thresholds align with previous observations of WTD being disconnected from surface moisture conditions when the WT is greater than 60 cm in other drained peatlands (Price, 1997; Strack et al., 2009). As in extracted peatlands, in natural peatlands, most studies have not observed a clear negative relationship between WTD and ET rates (Lafleur et al., 2005; Moore et al., 2022; Moore et al., 2013; Scarlett et al., 2017; Wu et al., 2010). This is due to the influence of vegetation community composition on ET rates, and the presence of negative feedbacks, such as the WTD–albedo feedback (Waddington et al., 2015) and WTD–surface resistance feedback (Kettridge & Waddington, 2014), which reduce E rates as WTD increases.

4.5.1.2 Importance of VPD Depends on WTD

We observed that the importance of VPD on E rates varied with WTD, aligning with the WTD threshold as a delimiter between the importance of surface conditions and atmospheric conditions. We found that VPD explained ~25% of the variation in E when the WT was in the top 40 cm at AB, but then was not a strong control under greater WTD. This is expected, as water vapour moves along a VPD gradient from the surface. Our observed rectangular hyperbola relationships suggest that as the WTD increases, the atmospheric control is overridden by the lack of available water for E. Values of unsaturated hydraulic conductivity of catotelmic peat range from less than 1 mm/hr to over 50 mm/hr, with larger values at higher moisture contents (Balliston and Price, 2020; McCarter and Price, 2014). These values are larger than the asymptote reached in the E–VPD curve, suggesting that upward vertical transport can in some instances be sustained beyond our modelled asymptote.

In vegetated peatlands, the relationship between VPD and ET is wide ranging. A near logarithmic relationship, similar to our finding, is often observed (Alekseychik et al., 2018; Popović et al., 2024; Peichl et al., 2013). However, in this case, the plateauing of ET rates at high VPDs is likely due to the ET–stomatal control feedback whereby vascular vegetation can close their stomata under high VPDs to conserve water (Waddington et al., 2015). Studies have also observed strong positive relationships when moss was the dominant vegetation cover (Admiral et al., 2006), as well as weak to no relationship between VPD and ET Brown et al. (2010). Vapour pressure deficits are expected to increase under anthropogenic climate change (Ficklin & Novick, 2017), making it important to understand how peatland water balances will respond. Water loss from natural peatlands is expected to increase, leading to overall increases in WTD (Helbig et al., 2020). Our research suggests that any

increase in VPD will not be a primary driver of water loss in extracted peatlands when WTD are greater than 40 cm.

While wind speed, the main component of aerodynamic resistance, is often positively correlated with E rates (Lafleur, 2008), it was not an important control in this study. Our sites are wide open fields, and thus there is likely always sufficient wind to transport the saturated air away from the peat surface. Additionally, bare soil E research suggests that wind speed will not be an important control when the evaporating plane has moved below the soil surface (Davarzani et al., 2014). Harrowing practices at these sites, and the associated dry surface layer, means that the evaporating plane may be below the surface for large periods of time at extracted peatlands (see Section 4.5.4).

4.5.2 Potential Versus Actual Evaporation Rates

Actual E was well-correlated with potential E, and the slope and strength of the relationship varied with WTD. Actual E was closer to potential E when the WTD was within the top 40 cm and 60 cm at AB and QC, respectively. Under these conditions, surface VWC was greater, and closer to the condition of no water limitation. The high degree of scatter when WTDs were greater than these thresholds, as evidenced by the lower explanatory power of the linear relationships, again suggests that WTD is not an ideal predictor of surface moisture conditions in drained systems (e.g., Price, 1997). We observed actual E rates close to potential rates even with WTDs greater than 60 cm. The overall high correlation between potential and actual E suggests that the Penman-Monteith equation is a suitable choice for modelling E from extracted peatlands. Estimates can be improved by differentiating periods when the WT is above and below our identified thresholds.

4.5.3 Importance of VPD Differences Between Sites

Multiple results suggest the importance of VPD differences between AB and QC for understanding E rates. The 20 cm deeper WTD threshold delimiting the relative importance of atmospheric versus surface controls on E at QC compared to AB suggests that high surface moisture conditions can be sustained under a greater range of WTD at QC. This is additionally supported by the potential versus actual E figures (Figure 4.8), where there was a greater overlap in the scatter of points above and below the WTD threshold at QC compared to AB. These differences are likely due to the greater VPD at AB, highlighting the importance of considering climatic zones when understanding E dynamics. However, differences in vertical pore connection could have also contribute to these

differences. With greater pore connectivity, the upward capillary movement of water will be able to maintain a given surface moisture content at a greater WTD (Price & Ketcheson, 2009). This result is supported by the distinct patterns in the soil water retention curves at each site (Figure B5, Lai, 2022). At AB, there is a breakpoint at ~55 cm WTD, while the relationship between VWC and WTD is more linear at QC. This finding is important for future research in drained peatlands. Within a site, WTD measurements will be useful for understanding E rates. But care needs to be taken when assuming that E rates will be comparable between sites in different climatic zones, even with similar WTD.

4.5.4 Effect of Harrowing on Evaporation

The importance of the presence or absence of a harrowed layer for E rates depended on the length of time that layer was present. During sunny periods with low cloud coverage (average midday incoming solar radiation > 500 W m^{-2}), there was an initial increase in E from the harrowed treatment, likely due to the increased peat surface area exposure to wind following harrowing, facilitating the drying of the surface (O'Kane, 1992). By 24 hours, we found that the non-harrowed treatments had evaporated more, but not significantly more, water after 24 hours. Therefore, when companies vacuum harvest the harrowed layer after one day, the harrowing process likely has a limited effect on sitewide daily water loss. However, the newly dry harrowed layer acted as a barrier to further E, as suggested by the higher E rates from the non-harrowed treatment, as soon as 5.5 hours post harrowing. This is presumed to be through an increase in the surface resistance to E. This reduction was large, with the non-harrowed treatment evaporating on average 73% and 67% more water than the harrowed treatment 48 and 72 hours post harrowing, respectively.

While no field study has looked at the effect of harrowing on E at extracted peatlands, our findings agree with the early modelling work of O'Kane's (1992). In agricultural settings, no-till procedures are often combined with the addition of crop residues on top of the soil to reduce water loss (Verhulst et al., 2011), making those systems difficult to compare to our sites. However, field studies have found that tillage can increase evaporative losses (Jalota and Prihar, 1990), while also, depending on the surface characteristics, increasing infiltration rates (Burns, 1971).

Harrowing is an acceleration of a negative feedback process that would have occurred naturally. Under the two-stage theory of E, during stage one, E occurs at the soil surface, and water for E is supplied by the upward capillary movement of water. The main control on E rates is the vapour pressure gradient between the surface and the air (Lehmann et al., 2018; Or & Lehmann, 2019). As

the surface dries, the system will shift to stage two, where the E plane has moved below the surface, and E rates also now depend on vapour diffusion rates through the peat (Lehmann et al., 2008). This negative feedback, where the resistance to E increasing when the surface peat moisture cannot meet the atmospheric demand, thereby reducing E rates, has been observed in moss dominated peatlands (Waddington et al., 2015). This study therefore adds to the peatland literature on this mechanism, which, due to the presence of both vascular and non-vascular vegetation at most peatland sites, is hard to isolate.

Understanding how the harrowed layer alters E rates in the hours and days following harrowing will be important from a site management perspective. Our work suggests that during sunny periods, by 10 hours post harrowing, the surface layer has dried sufficiently such that it is now acting as a barrier to further E from below. From a site water management perspective, harvesting should occur by this point. Leaving the harrowed layer present for longer will limit evaporative water loss from the layers below, which is counteractive to their management goals. Companies harrow the surface layer to dry it and make it easier to vacuum harvest, and cheaper to transport (Warren Walker, Sun Gro Horticulture Natural Resource Director, Pers. Comm.). From a site C perspective, limited research, at both the plot (Velosa, 2024, unpublished) and ecosystem scale (Chapter 3), suggests that the presence of a dry harrowed layer will not significantly alter CO_2 emissions from these sites.

4.6 Conclusion

This was the first study in Canada to conduct measurements at actively extracted peatlands to understand the impact of surface and atmospheric conditions, and active site management, on energy partitioning and E. Despite being drained, available energy is largely partitioned into Q_E, and E rates are comparable to those from boreal bogs. Potential E rates calculated using the Penman-Monteith equation with surface resistance set to zero were well correlated with actual E rates, especially when the WT was within the top 40 and 60 cm at AB and QC, respectively, making it a suitable choice for modelling E at these sites.

This work showed the WTD will not always be a good predictor of E rates at these sites. We confirmed that the drainage induced changes to peat hydrophysical properties at these sites can support strong capillary connectivity between the WT and the surface, even under periods of deep WT. Our WTD thresholds of ~ 40 cm and 60 cm at AB and QC, respectively, can help inform the relative importance of atmospheric and surface conditions on E rates. When WTD was above these

thresholds, VPD and VWC were important controls, while WTD was not. Under deeper WT conditions, VWC and WTD were the strongest predictors. The difference in the WTD threshold between the two sites highlighted that peat hydrophysical properties cannot be assumed to be similar across actively extracted peatlands, despite the peat all being highly decomposed. Care therefore needs to be taken when extrapolating E rates to other extracted sites with different hydrologic conditions.

We found that the impact of management practices on E depended on the length of time between harrowing and harvesting, and its impact should be considered when the harrowed layer is left for multiple days. This work can aid site managers in effective harrowing procedures by considering how soon after harrowing they should harvest. Due to the absence of vegetation at extracted sites, this work can contribute to furthering our understanding of the controls on E from bare peat. Additionally, our evaporation–WTD findings can help inform restoration practices at these sites by providing information on the WTD required to maintain capillary connectivity between the surface and WT to support moss establishment.

Chapter 5

Conclusions

This research employed incubation and field methods at the plot and ecosystem scale to improve our limited knowledge around C cycling and E dynamics in actively extracted peatlands in Canada. This multi-year and multi-site research will have applications for scientific inquiry (Section 5.1), industry partners (Section 5.2), national C accounting and policy (Section 5.3).

5.1 Key Findings

5.1.1 Effect of Soil Temperature and Hydrologic Conditions on C Emissions

The absence of vegetation provides a unique opportunity to understand heterotrophic respiration in peatlands without the confounding impact of vegetation. This work furthers our understanding of the impact of soil temperature, hydrologic conditions, and substrate quality, and their interactions, on C emissions through its multiple measurement techniques at different scales. To date, most research at extracted peatlands has focused on quantifying emissions to understand the C benefit of restoration. Ecosystem scale measurements (Chapter 3) demonstrated that the conventional positive relationship between heterotrophic CO_2 emissions and WTD does not hold in heavily drained peatlands, and that CO_2 emissions can be reduced under very deep WT conditions. At AB, emissions were at least 25% lower in a dry year (WTD greater than 60 cm), compared to during wet years where the WT was regularly in the top 40 cm of the peat profile. Plot scale chamber measurements (Chapter 2) confirmed that the well-known temperature–CO_2 relationships are still relevant in the catotelmic peat of extracted peatlands. With the ecosystem scale measurements (Chapter 3), we showed that in deeply drained systems, the temperature dependence of CO_2 emissions may be greater under wet conditions (WTD <50 cm) compared to dry conditions. With the drainage of peatlands for industrial use increasing in Canada, it is essential to understand the temperature–WT interactions driving CO_2 emissions under severe drainage. We suggest that future research in drained peatlands divide data by WTD to better understand these mechanisms. Additionally, processed-based models need to consider these relationships to avoid overestimating emissions during drier years.

Through the incubation and chamber-based measurements (Chapter 2), we were able to understand the importance of substrate quality as a driver of C emissions. Our chamber measurements indicated

that CO_2 emissions were significantly higher from the drainage ditches than the fields, despite the often-higher moisture content in the ditches. Through an incubation experiment, we demonstrated that these differences were due to the greater importance of substrate quality versus moisture content on C emissions. This research also lays the groundwork for future research to investigate whether porewater drainage from the fields to the ditches is providing a labile substrate for decomposition. Our multi-year chamber measurements (Chapter 2) at different peatland complexes of varying extraction durations highlighted the need to consider both extraction duration, and substrate quality, when creating emission factors for extracted peatlands (see Section 5.3).

5.1.2 Spatial Distribution of CO_2 and CH_4 Emissions

This study indicated that there was limited spatial variability of CO_2 and CH_4 emissions across the non-contoured peat fields that are common in western Canada (Chapter 2). At sites with a range of extraction durations, and across both wet and dry years, we did not observe a significant difference in emissions with distance from the drainage ditches. This research was the first multi-year study on non-contoured fields to assess this, filling an important research gap. We suggest that future measurements at non-contoured sites do not need to consider distance from the drainage ditches when estimating field emissions.

We found that emissions from intact ditches were significantly greater than field emissions, highlighting the importance of including ditch measurements in future sampling plans to accurately quantify site wide emissions (Chapter 2). Our chamber emissions demonstrated that CO_2 and CH_4 ditch emissions were around double, and ten times the field emissions, receptively. This finding was confirmed by our incubation experiment, in which we observed significantly greater CO_2 production potential from the ditches compared to the fields. To date, research has been focused on field emissions, with only a handful of studies measuring C emissions from drainage ditches. The current estimates are wide ranging, especially for CH_4. Given that the drainage ditches in this study emitted disproportionally to their surface area, understanding how ditch emissions will vary with environmental and management conditions will be necessary to understand sitewide emissions.

5.1.3 Surface Versus Atmospheric Controls on Evaporation

The ecosystem scale measurements of E (Chapter 4) confirmed that despite being a drained site, available energy at extracted sites is largely partitioned into latent heat (Q_G). We observed that WTD

was not always a good predictor of E rates, as the highly compacted peat likely had a strong capillary connection between the WT and the surface, allowing for maintenance of high E rates even as the WT dropped to 60 cm below the surface. However, this study showed that WTD thresholds can be used to further our understanding of controls on available energy partitioning and E rates. Surface moisture was the most important control on E rates across the whole range of WTDs. Atmospheric VPD was important when WTD was less than ~40 cm at AB, but not at greater depths. Evaporation rates did not significantly vary with WTD when WTs were within the top ~40 and 60 cm at AB and QC, respectively, but we found a negative effect under deeper WTs. The non-linear relationship between E and WTD, and the effect of the VPD−WTD interaction on E, can help inform our understanding of E rates in vegetated peatlands. The absence of vegetation provides a unique opportunity to understand the drivers of E dynamics from peatlands without having to partition the effects of surface and atmospheric controls on E and T.

5.1.4 Catotelmic Peat is not the Same at all Peatlands

Results from Chapters 2, 3 and 4 all suggest that there are inherent biogeochemical and hydrophysical differences in the catotelmic peat across actively extracted peatlands, and that they need to be considered when evaluating C and hydrologic cycling. Analysis of grab samples (Chapter 2) showed the peat age and C:N were quite different between sites, with the Young site having more bog-like peat, compared to more fen-like peat at the Intermediate Site. Ecosystem scale CO_2 emissions (Chapter 3) were higher at QC compared to AB in most months, despite a similar extraction duration. When investigating the effect of WTD on E (Chapter 4), we observed a WT control on E rates at a shallower depth at AB compared to QC. The higher atmospheric demand for water vapour at AB is likely responsible for this, but differences in peat hydrophysical properties between sites could have also contributed. A greater vertical pore connectivity at QC compared to AB would have allowed for maintenance of high E rates at a greater WTD. These results all suggest that we must be careful when generalizing C and E fluxes across sites with different peat quality, peat structure, and hydrologic conditions.

5.2 Recommendations for Industry Site Management

Our plot scale chamber CO_2 and CH_4 measurements (Chapter 2) under a variety of environmental and management conditions will guide peat extraction companies on how best to manage their ditches to balance harvesting yields with C emissions. We showed that maintaining some standing water in the

interior ditches would significantly reduce C emissions from these sites. Our work also suggests that increasing the spacing of the drainage ditches at extracted sites would reduce sitewide C emissions. We found that CH_4 emissions were significantly reduced when drainage ditches were partially to fully filled in with peat at the Young Site. This could be applied at newly extracted sites where the surface peat has an adequate saturated hydraulic conductivity to support sufficient lateral drainage, even with the reduced number of ditches. The plot scale lysimeter E measurements (Chapter 4) can inform harrowing and harvesting practices at peat extraction sites to optimize peat extraction efficiency. Our work suggests that the surface layer has dried out within 9 hours post harrowing during sunny periods, and sometimes within 5.5 hours. The prolonged presence of a dry harrowed layer on the surface will limit water loss from the peat below, which is counteractive to their management goals.

5.3 Recommendations for Carbon Accounting

This work was part of a larger project to quantify and understand mechanisms of C cycling during the life cycle of peat extraction to inform Canadian C emission factors. While previous studies have compiled components of this life cycle (Cleary et al., 2005; Waddington et al., 2009), they have been limited in part by a lack of measurements of C emissions during active extraction. Previous estimates were based on C emissions from post-extracted unrestored peatlands, with some of these sites having non-functioning drainage ditches and or partial natural revegetation. Previous measurements were all based in Quebec with bog-like peat, and thus may not be representative of the fen-like peat found at many extraction sites in Alberta and Manitoba. Additionally, measurements were generally conducted in only one year, limiting our understanding of interannual variability. This study's multi- year chamber (Chapter 2) and eddy covariance (Chapter 3) measurements will help answer a few key questions: i) how much CO_2 and CH_4 is emitted from actively extracted peatlands and what is the interannual variability, and ii) should emission factors change with time since the start of extraction?

The measured multi-year plot and ecosystem scale CO_2 and CH_4 emissions can be added to existing measurements to estimate C emissions at actively extracted peatlands in Canada. The collection of data at both spatial scales allows for multiple methods of estimating the contribution of both fields and ditches to sitewide emissions. This study was unfortunately not able to quantify November to February emissions, due to instrument failure and COVID-related travel restrictions. However, it was able to measure March and April emissions at AB in two years, which are historically not included in chamber studies due to fieldwork logistics. These measurements illustrate the importance of capturing

springtime conditions for annual estimates of C loss as C emissions in these months were variable but could be as high as summer emissions in some years.

This research builds on work by Clark et al. (2023) on determining if and how emission factors should change with time since extraction. Clark et al. (2023), completing their work at one peatland complex, proposed one CO_2 value for newly open sites, and a second value for sites that have been extracted for longer than ~5 years. The basis for this was that during the 15 to 40 plus years of extraction, surface peat is removed, exposing older and more recalcitrant peat at the surface that will have lower decomposition rates. Through measurements at extraction sites of multiple ages in multiple peatland complexes, our research showed that this relationship is not universal. Differences in CO_2 emissions between peatlands can be greater than changes in CO_2 with time since extraction. Emissions factors should consider both the time since extraction, as well as the substrate quality of the peat. More measurements will need to be made but grouping extracted sites by C:N ratio could be an inexpensive and easy way to categorize emission factors. Our work confirms Clark et al.'s (2023) finding that CH_4 emissions do not vary significantly with time since extraction began.

5.4 Future Research

Based on the research presented in this thesis, and the limitations of our study, I have a few suggestions for future research at extracted peatlands, in addition to what has been presented above.

The first is that future research at extracted peatlands needs to better characterize the surface moisture content of the peat to understand C cycling and E. While easy to measure, our results highlighted how WTD was not a good predictor of surface moisture conditions under heavily drained conditions. Due to COVID-19 restricting site access, our continuous VWC measurements in the harvesting exclusion zone were sporadic, making them difficult to use as a predictive variable. Obtaining continuous surface moisture measurements that include the effect of harrowing and harvesting will be difficult due to the need to not interfere with site management. Remote sensing options, such as synthetic aperture radar from satellite data has been used in peatlands to estimate surface moisture content at a 10 m spatial resolution (Lees et al., 2021), and could be applied at these sites. For future measurements in harvesting exclusion zones, surface tension measurements (e.g., Moore et al., 2021) may provide a better idea of the water availability at the surface for E compared to VWC.

The second recommendation is that future research should look at production and transport of CO_2 and CH_4 during WT fluctuations. To date, most studies, including our own, have sought to understand

how longer term (monthly to seasonally) increases or decreases in average WTD affect C emissions. Given the large WT fluctuations at these sites, there is a need to understand how hourly or daily changes in WTD, such as after large rain events, affect C cycling. Our soil water retention curves indicated the presence of hysteresis, but we lacked the temporal resolution in our ecosystem scale data to understand the effect of this hysteresis on C dynamics. Additionally, we were unable to quantify the time lags between a change in redox conditions, and the start of CO_2 and CH_4 production. A column experiment with artificial manipulation of the WT could be a viable method to assess the impact of WT fluctuations on C dynamics.

The third recommendation is for future studies to verify our findings on the effect of harrowing on E rates at the ecosystem scale. In our lysimeter experiment, we raked the surface of the peat to replicate the impact of harrowing. However, we were unable to assess E rates on pre- and post-harrowing on the actual peat fields. We attempted to use a hunting camera to detect when harvesting and harrowing practices were occurring, and then link this timing with our ecosystem scale E measurements. However, due to the low percentage of data that passed quality control at our AB site, and the limited time span of camera measurements, we lacked the temporal resolution in our data to assess this. Evaporation is a major source of water loss from these sites, and there is a tight link between hydrologic conditions and C cycling. Understanding the impact of harrowing on daily to multi-day E rates will be vital for understanding the sitewide water balance and will support future modelling efforts at these sites.

Finally, our results also highlight a need for additional measurements of CO_2 emissions from heavily drained (WTD > 60 cm) peatlands in general, to better understand the non-linear relationship between WTD and CO_2 emissions, and the interacting effect of soil moisture and temperature, on CO_2 emissions. Given that an estimated 10% of global peatlands have been drained (UNEP, 2022), and the potential for future WT drawdown in response to a warming climate (Swindles et al., 2019), understanding these processes will be key to predicting the fate of peat C (Zhong et al., 2020). Our study observed that under deep WTs, a further increase in WTD can lead to a decrease in CO_2 emissions, likely due to limitations of moisture content for decomposition. We also found a reduced dependence of CO_2 emissions on temperature under drier conditions. Few studies have conducted measurements in such heavily drained systems, making it hard to understand mechanistic processes behind these patterns. Research that also incorporated measurements of microbial community structure and abundance shifts following drainage, as well as assessing changes in peat chemistry

associated with drainage, could improve our understanding of mechanistic pathways of production and consumption of these gases.

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Appendix A

Table of Contents:

Text A1 to A4

Figures A1 to A6

Tables A1 to A6

Text A1 Field transect layout.

Eight transects were placed across two fields at the Old and Intermediate Sites in May 2019 (Figure 1). The company decommissioned the Old Site prior to the start of our measurements in 2021. As the site was no longer under active extraction, we removed it from our sampling. The transects at the Intermediate Site continued to be used for measurements in 2021 and 2022. We placed five transects along one field at the Young Site in May 2019. Two transects were removed from our data analysis due to shallow peat depth and visible clay at the surface, making them not representation of average peat extraction sites across Canada. In 2021, we placed eight new transects across two fields to follow a similar sampling design to the other sites (Figure 1). Equipment supply chain issues meant that the ditches were not re-dug out by the peat producer prior to harvesting in 2021. Ditch measurements were made where the dug-out ditches would have been. In 2022, only half of the ditches were dug, creating a spacing of 60 m between each drainage ditch. We shifted a set of four transects to the other site of the field so that all eight transects had a dug-out ditch. The ditches were gradually filled in with surface peat over the course of the 2022 study period by equipment during harvesting activities.

Text A2 Calculation of steady and episodic carbon gas fluxes.

The steady portion of gas fluxes (i.e., diffusive fluxes, CO_2: g C m^{-2} d^{-1}; CH_4: mg C m^{-2} d^{-1}) was calculated using the ideal gas law and was corrected for chamber volume and temperature, using equation 1, where f_x is the linear rate of change of the steady gas flux (ppm min^{-1}), V is the chamber volume (L), R is the ideal gas constant (0.0821 L atm K^{-1} mol^{-1}), T is the average chamber air temperature (K), n is the molecular mass of the gas (kg mol^{-1}), t is the number of minutes in a day (1400 min day^{-1}), and S is the surface area of the chamber (m^2).

$$Flux = \frac{f_x \times (\frac{V}{R \times T}) \times n \times t}{S \times 1000} \qquad (Equation\ 1)$$

A visual inspection of the data was used to highlight possible periods of CH_4 ebullition. First, we calculated a first difference for each CH_4 measurement in a flux by dividing each measurement by the previous one. We then calculated a mean rate of change for the steady flux. The steady flux period needed to occur for at least 20 seconds and have an R^2 greater than 0.9. In the case of there being multiple identified periods of steady flux, an average was used. Finally, we calculated a percent difference between the first difference and the mean steady rate of change. To be categorized as an episodic event, we required there to be three or more continuous seconds with a percent difference of 100% or more. There also needed to be at least 10 seconds of steady flux slope prior to and following the ebullition event.

Steady, episodic, and a total CH_4 flux were calculated for each of the identified fluxes with ebullition events. The steady flux was calculated using equation 1, with the rate of the change being the mean steady flux rate of change. For the episodic flux, we calculated the rate of change of CH_4 concentration during the ebullition period, and then subtracted the mean steady rate of change from it, with the assumption that diffusion had been occurring during the episodic event. This number was multiplied by the duration in seconds of the episodic flux, and was then divided by the total flux length to determine an average rate of change in ppb/s. This rate of change was then used in equation 1 to determine an episodic flux contribution during the chamber closure. The total CH_4 flux was calculated by adding the steady and episodic portions together.

Text A3 Methods for ^{14}C and FTIR analysis of peat samples.

The samples were stored in a freezer prior to analysis. They were thawed and homogenized, and then a subsample was dried in an oven at 80 °C for 48 hours. To determine the age of the organic matter, 20 mg subsamples of the dried peat were processed for ^{14}C dating at the AEL AMS Laboratory at the University of Ottawa, using a 3MV accelerator mass spectrometer. The samples underwent physical and chemical pre-treatment (see Crann et al., 2017 for methods), and then graphitization (St-Jean et

al., 2017). The IntCal 2020 (Reimer et al., 2020) and OxCal 4.4 (Bronk Ramsey, 2009) curves were used for calibrating the carbon data.

Another subsample of the peat was ground using a mortar and pestle, and then filtered through a 500 μm mesh sieve for two additional analyses. Fourier transform infrared (FTIR) spectrometry was conducted at the Ecohydrology and Biogeochemistry Research Group at the University of Münster to assess substrate quality of the peat samples. Humification indices (HI) were calculated using the 1630 cm^{-1} and 1090 cm^{-1} bands, which represent aromatic and aliphatic carboxylates, and carbohydrates, respectively (Broder et al., 2012). A baseline correction was performed (Beleites and Sergo, 2020) with the "ir" R package (Teickner, 2022). Further analysis was performed using the "irpeat" R package (Teickner and Hodgkins, 2022). Carbon to nitrogen analysis was conducted at the Whalen Lab at the McGill Macdonald Campus in Quebec, Canada, using the FLASH1112 Elemental Analyzer.

Text A4 Calculation of nonlinear CO_2-soil temperature relationship and Q10 values.

We chose an exponential non-linear model (Equation 2), where T was the soil temperature, and a and b were fitted parameters. We used AIC values during model selection to assess the non-linear model performance compared to the linear model. Q10 values for the soil temperature CO_2 relationship were calculated using Equation 3, where b was the fitted parameter from Equation 2. For a 10 ˚C increase in temperature, Q10 represents the factor by which soil respiration will increase (Liu et al., 2024).

$$Carbon\ Flux = ae^{b \times T} \qquad (Equation\ 2)$$

$$Q10 = e^{10 \times b} \qquad (Equation\ 3)$$

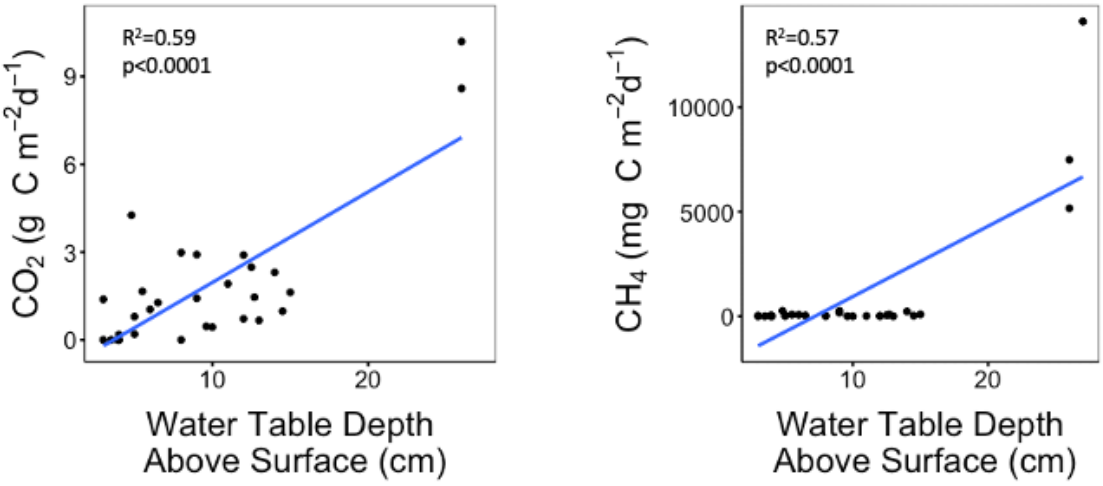


Figure A1 Scatterplots showing effect of depth of standing water in the ditches on CO_2 and CH_4 emissions at the Old Site in 2019.

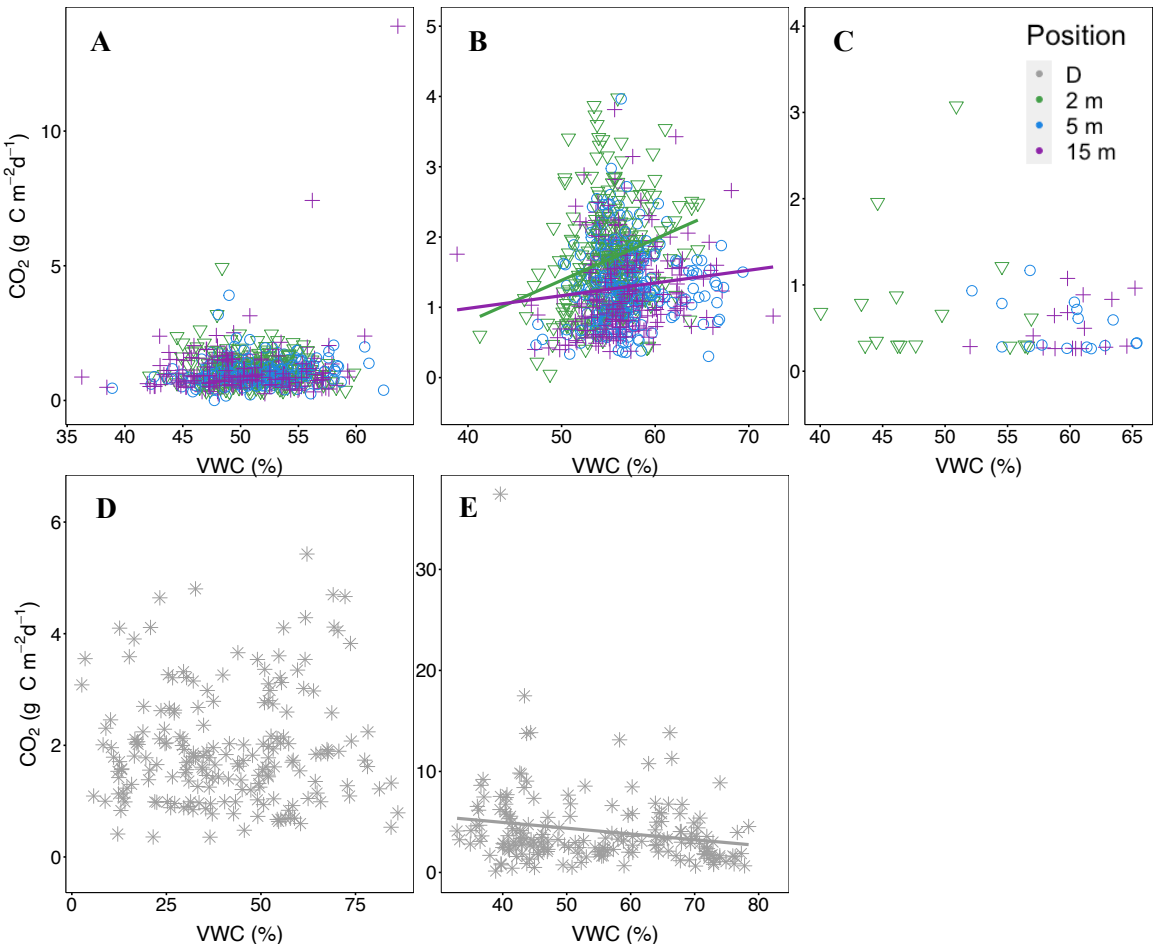


Figure A2 Scatterplots showing effect of peat top 20 cm depth VWC on CO_2 field emissions at the Young (A), Intermediate (B) and Old (C) sites, and CO_2 ditch emissions at the Young (D) and Intermediate (E) sites. The marginal R^2 values were 0.036, 0.071 and 0.029 for the Intermediate Site ditch, 2 m and 15 m positions respectfully. The field and ditch position emissions were plotted separately for ease of visualization.

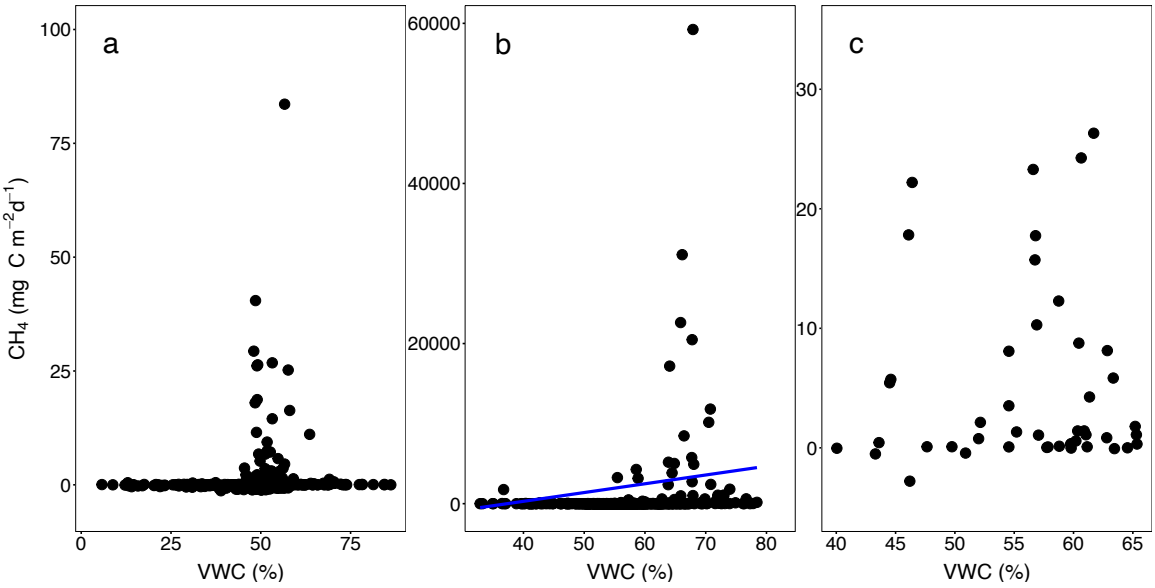


Figure A3 Scatterplots showing effect of peat top 20 cm depth VWC on CH_4 emissions across all measurement Positions at the Young (a), Intermediate (b) and Old (c) sites.

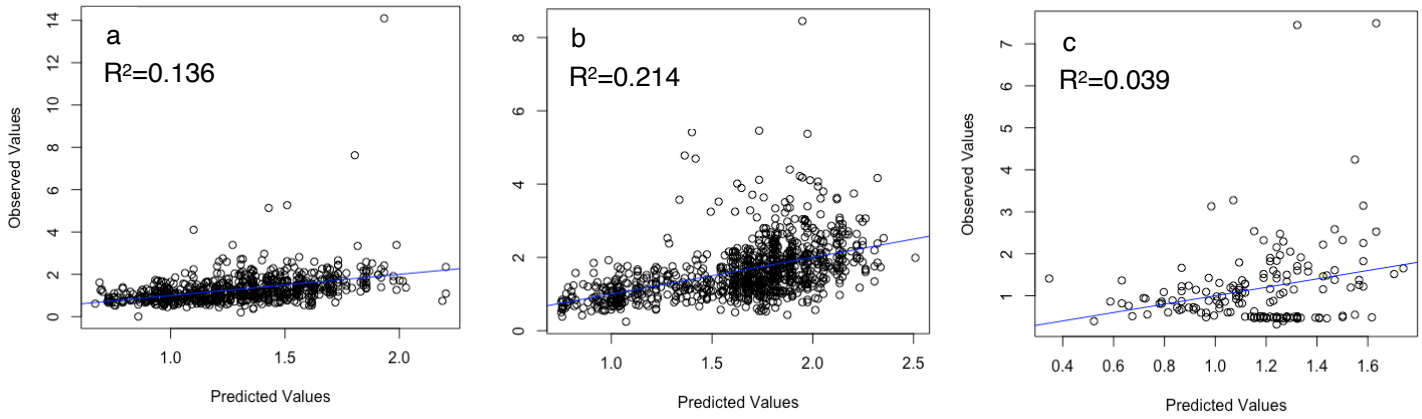


Figure A4 Plots of the predicted vs observed values from the nonlinear model of the effect of soil temperature at 10 cm depth on field CO_2 emissions at the Young (a), Intermediate (b) and Old (c) Sites across the three study years. The blue line plotted is the 1:1 line.

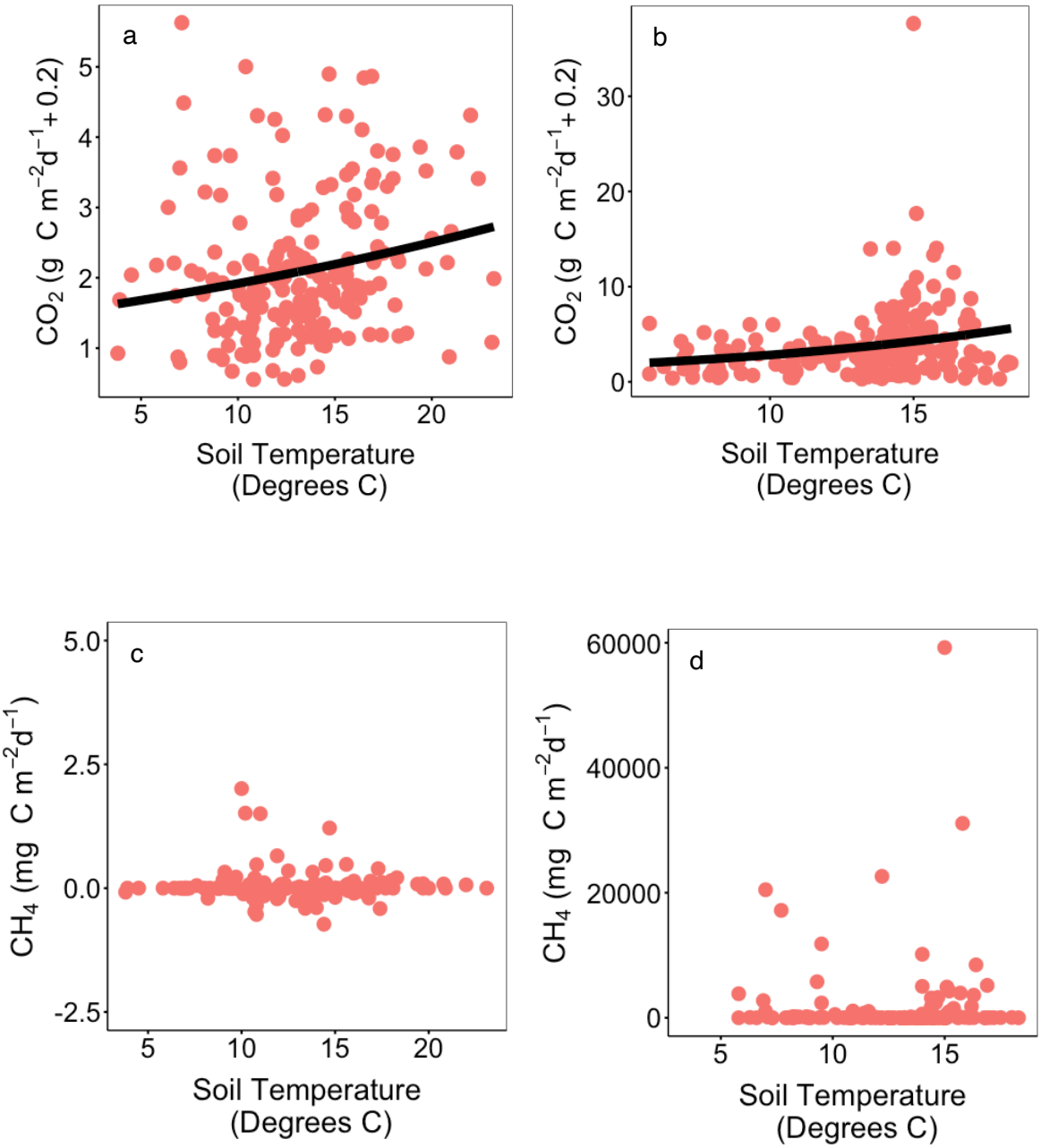


Figure A5 Scatterplots showing the effect of ditch soil temperature at 10 cm depth on CO_2 and CH_4 emissions Intermediate (b, d) and Young (a, c) Sites. The nonlinear regression model was included when significant. The correlations between the observed and predicted values and residual sum of squares for the regression models were 0.19 and 195.3, and 0.22 and 2814 for the Young and Intermediate Site respectively. The regression equations were $y=1.471e^{0.0266x}$ and $y=1.244e^{0.0819x}$ for the Young the Intermediate Site.

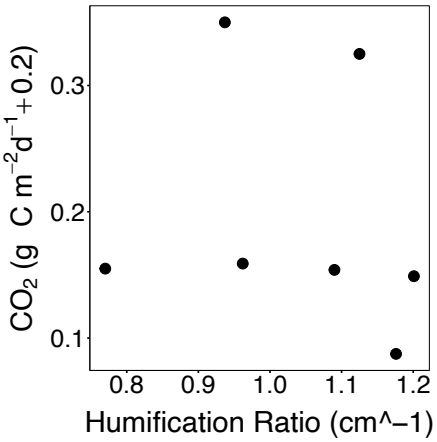


Figure A6 Scatterplot showing the effect of peat humification on CO_2 fluxes from the incubation experiment.

Table A1 Results of LME model, with collar as a random factor, for the effect of Site and Position on 10 cm depth soil temperature and top 20 cm VWC in 2019, 2021 and 2022.

		Soil Temperature			VWC		
		DF	F	P	DF	F	P
2019							
	Site	2, 61	3.41	0.040	2, 48	23.12	<0.0001
	Position	2, 61	0.46	0.633	2, 48	33.31	<0.0001
	Site: Position	4, 61	0.15	0.956	4, 48	4.56	0.0034
2021							
	Site	1, 91	0.12	0.735	1, 91	115.88	<0.0001
	Position	3, 91	1.39	0.251	3, 91	5.85	0.0011
	Site: Position	3, 91	1.68	0.178	3, 91	17.90	<0.0001
2022							
	Site	1, 56	1.07	0.305	1, 55	70.88	<0.0001
	Position	3, 56	1.85	0.149	3, 55	29.77	<0.0001
	Site: Position	3, 56	0.37	0.775	3, 55	10.14	<0.0001

Table A2 Mean and median (in brackets) CO_2 (g C m^{-2} d^{-1}) and CH_4 (mg C m^{-2} d^{-1}) emissions from the measurement positions at the Young, Intermediate and Old Site in 2019, 2021 and 2022.

	2019		2021		2022	
	CO2	CH4	CO2	CH4	CO2	CH4
Young Site						
D	11.213 (11.749)	19743 (8775)	1.7045 (1.5715)	-0.11779 (-0.08596)	2.0621 (1.8349)	0.1009 (0.00)
2 m	1.6099 (1.5275)	7.4152 (0.5209)	1.2701 (1.1273)	0.1356 (0.00)	0.9708 (0.9594)	0.64285 (0.00)
5 m	1.628 (1.509)	3.5617 (-0.5398)	1.1124 (1.0377)	2.11764 (0.00)	0.9283 (0.8661)	-0.08946 (0.00)
15 m	2.5373 (1.4237)	1.5409 (-0.3760)	1.0733 (0.9816)	0.5066 (0.00)	0.9507 (0.8622)	0.1072 (0.00)
Intermediate Site						
D	1.0526 (0.6033)	424.749 (86.485)	4.3634 (3.2405)	242.273 (4.659)	2.29136 (1.39745)	2583.85 (53.18)
2 m	1.7223 (1.6481)	2.6831 (0.4247)	1.7942 (1.7044)	-0.03356 (0.00)	1.44355 (1.37371)	0.1632 (0.00)
5 m	1.2733 (1.5036)	1.9882 (0.5285)	1.5184 (1.4179)	2.9191 (0.00)	1.1130 (1.0671)	6.9734 (0.1679)
15 m	2.1167 (1.4943)	17.2639 (0.1534)	1.3799 (1.2926)	0.2563 (-0.1456)	1.1048 (0.9687)	2.1499 (0.00)
Old Site						
D	2.0563 (1.2727)	2234.792 (24.281)	n.m	n.m	n.m	n.m
2 m	0.9547 (0.7325)	3.35008 (0.03332)	n.m	n.m	n.m	n.m
5 m	0.8447 (0.6973)	4.05433 (0.15990)	n.m	n.m	n.m	n.m
15 m	1.1265 (0.6703)	1.41022 (0.13573)	n.m	n.m	n.m	n.m

Table A3 Statistical results of linear mixed effects model, with collar as a random factor, on the effect of depth of ditch standing water on CO_2 and CH_4 ditch emissions at the Young, Intermediate and Old Sites in 2019, 2021 and 2022. Sites with no standing water present were indicated with n.a (not applicable).

		CO_2			CH_4		
		DF	F	P	DF	F	P
2019							
	Young	1, 4	1.148	0.3443	1, 5	0.1998	0.6736
	Intermediate	1, 18	3.417	0.0751	1, 25	1.087	0.3071
	Old	1, 28	42.53	<0.0001	1, 29	40.51	<0.0001
2021							
	Young	n. a	n. a	n. a	n. a	n. a	n. a
	Intermediate	n. a	n. a	n. a	n. a	n. a	n. a
2022							
	Young	n. a	n. a	n. a	n. a	n. a	n. a
	Intermediate	1, 52	0.395	0.5324	1, 50	1.096	0.3001

Table A4 Statistical results of linear mixed effects model on the effect of VWC on CO_2 emissions from each measurement position across the three study years at the Young, Intermediate and Old Sites.

		DF	F	P
Young Site				
	Ditch	1,169	0.054	0.817
	2 m	1,162	1.640	0.202
	5 m	1,186	2.692	0.103
	15 m	1,183	1.061	0.304
Intermediate Site				
	Ditch	1,165	6.539	0.016
	2 m	1,249	24.438	<0.0001
	5 m	1,251	0.355	0.552
	15 m	1,246	10.300	0.0015
Old Site				
	Ditch	n.m.	n.m.	n.m.
	2 m	1,6	0.00940	0.926
	5 m	1,6	3.151	0.126
	15 m	1,7	1.541	0.254

Table A5 Statistical results of linear mixed effects model on the effect of VWC on CH_4 emissions from each Site across the three study years.

	DF	F	P
Young	1,169	0.0535	0.817
Intermediate	1,651	8.674	0.0033
Old	1,21	0.0218	0.884

Table A6 Statistic outputs from one-way ANOVA looking at predicted modeled data versus observed CO_2 and CH_4 emissions, where predicted emissions were calculated using the non-linear (NL) regression models at each Site. See Figure 5 in the main text for the regression equations. There is no model for the Old Site ditch data because no measurements of ditch soil temperature were taken.

		CO_2			CH_4		
	Site	DF	F	P	DF	F	P
Field Position							
	Young	1, 703	111.58	<0.0001	1, 549	1.251	0.264
	Intermediate	1, 915	250.29	<0.0001	1, 653	1.172	0.280
	Old	1, 161	7.589	0.0066	1, 88	1.154	0.286
Ditch Data							
	Young	1, 197	7.882	0.0055	1,152	0.0613	0.805
	Intermediate	1, 213	10.917	0.0011	1,143	0.154	0.696

Appendix B

Table of Contents:

Text B1

Figures B1-B7

Tables B1-B6

Text B1 Methods for assessing the effect of extraction activities on CO_2 emissions.

The trail camera was oriented westward and programmed to take a photo when it detected motion in front of it, with a detection range of ~ two peat fields. The photos were visually inspected to confirm the presence of extraction activity. Vacuum harvesting and harrowing activities were distinguished by the unique equipment required for each. To account for the unknown activity on the eastern half of the site, we added thirty minutes before and after the observed extraction activity time window. This was based on personal observations of extraction activity.

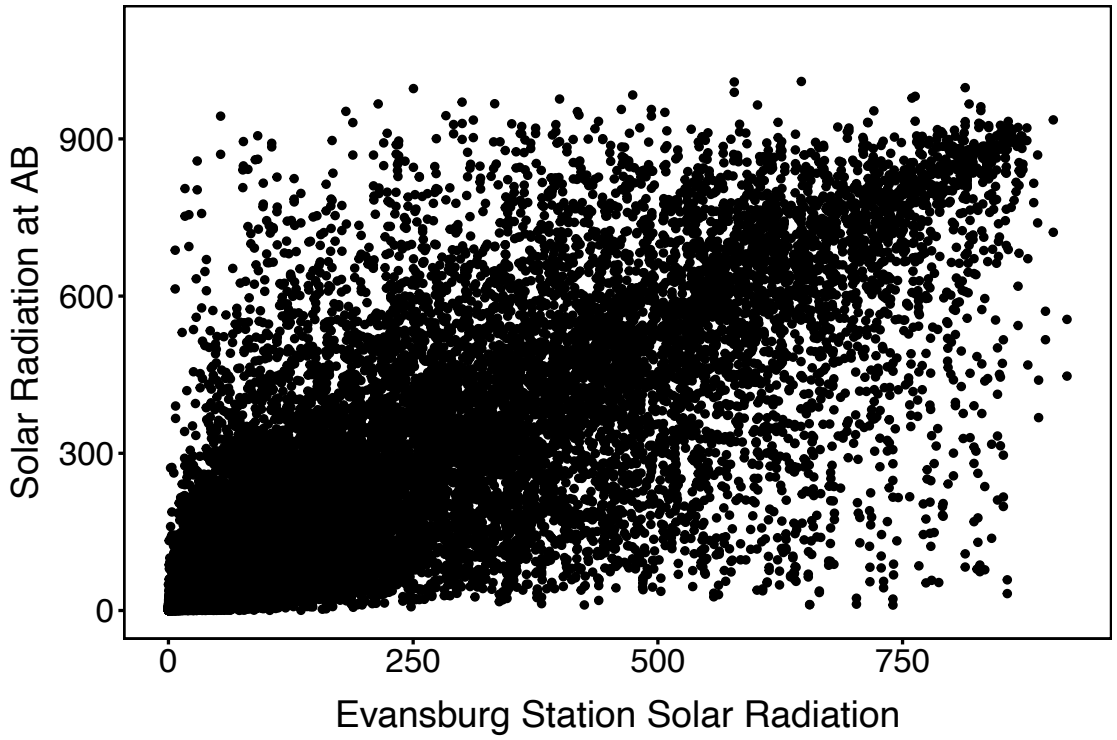


Figure B1 Scatter plot of the incoming solar radiation data (W m^{-2}) obtained from Evansburg 2 ADGM weather station (ACIS, 2023) versus the measured incoming solar radiation (W m^{-2}) at AB. The linear regression is significant ($F_{1,38451}$=103305, $p<0.0001$), with an R^2 of 0.73.

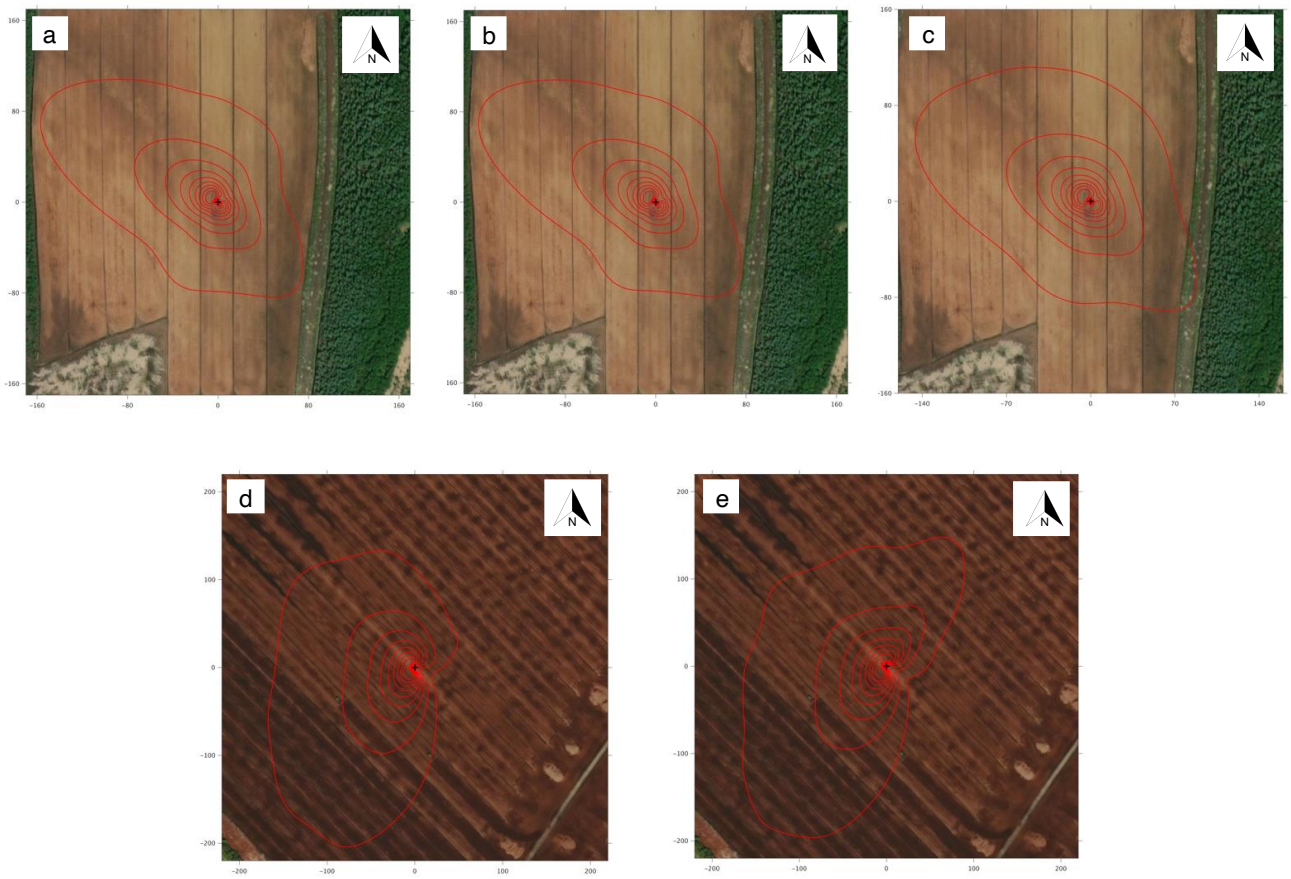


Figure B2 2D flux footprint probability contour maps of CO_2 at the AB site in 2020, (a) 2021 (b) and 2022 (c), and at the QC site in 2020 (d) and 2022 (e). The plots were created using the online tool by Kljun and others (2015). The scale bars along the x and y axes are in meters. The outermost contour line represents the 90th percentile probability. The North Arrow is from FreeImages.com.

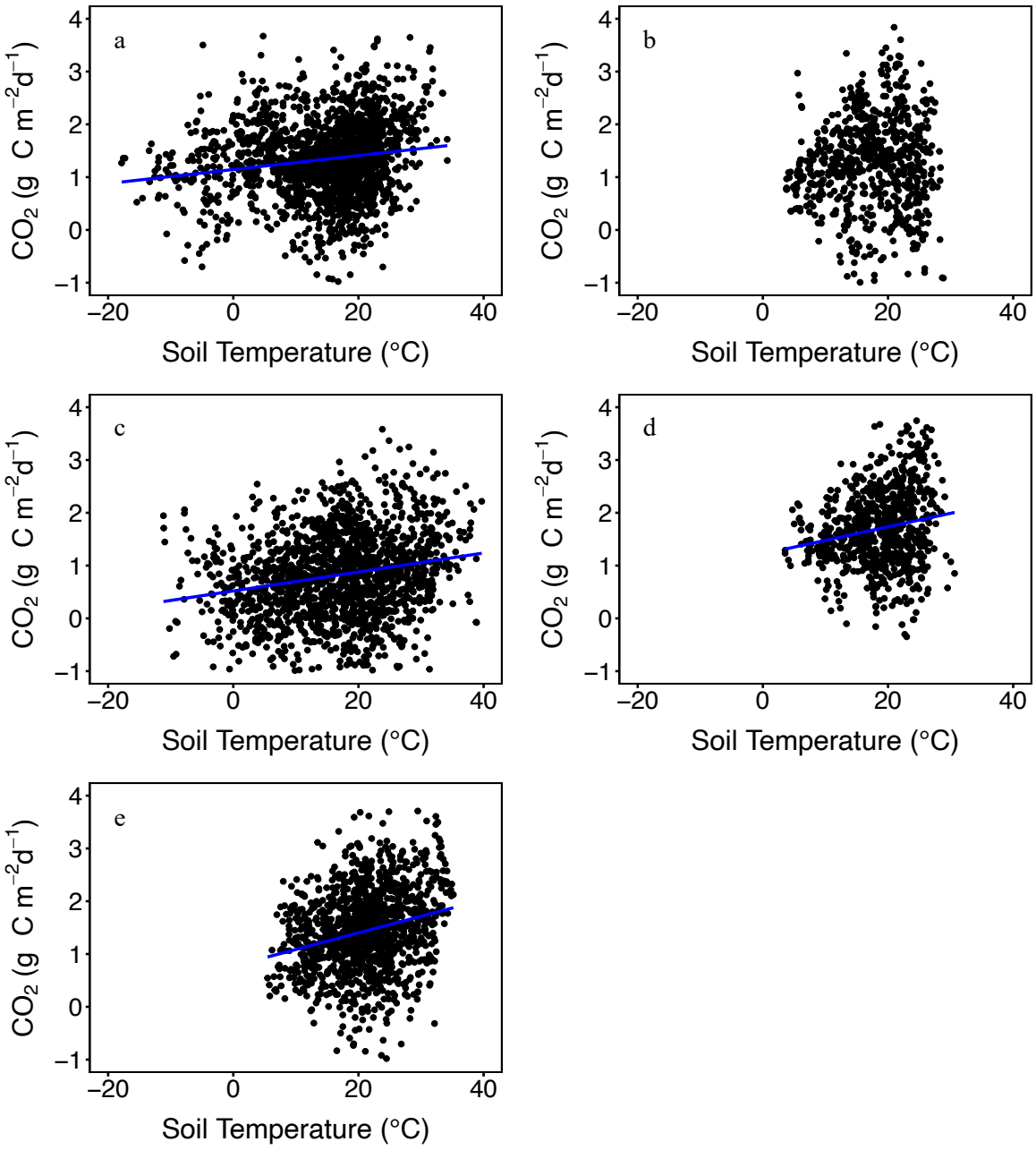


Figure B3 Scatterplots showing the effect of air temperature on CO_2 emissions in 2020 (a), 2021 (c) and 2022 (e) at AB, and in 2020 (b) and 2022 (d) at QC. Best fit lines were shown when linear regressions were significant. The explanatory power was 2.5%, 4.3% and 6.3% at AB in 2020, 2021 and 2022 respectively, and 3.5% at QC in 2022.

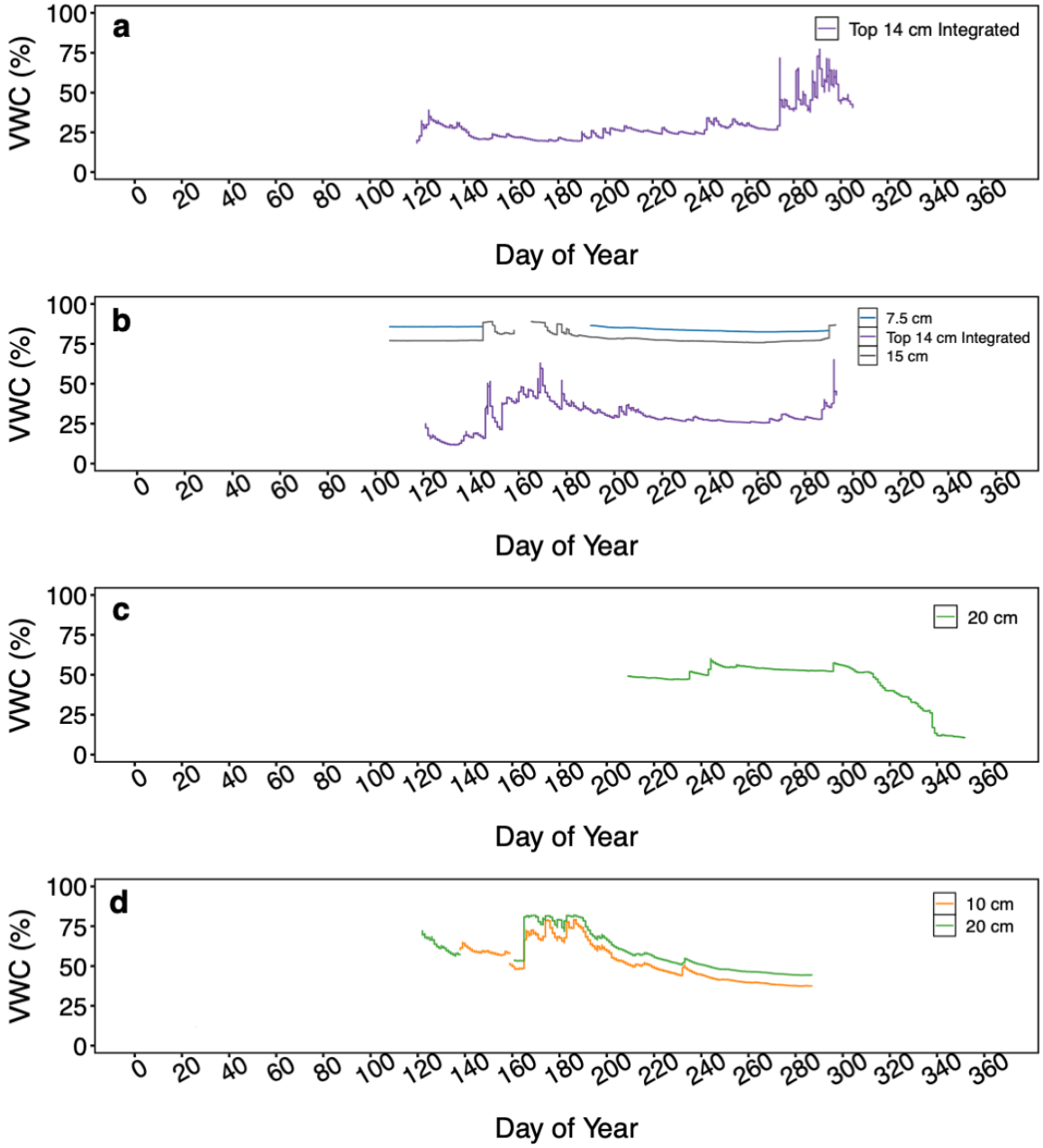


Figure B4 Volumetric water content (%) at QC during 2020 (a) and 2022 (b) and at AB in 2021 (c) and 2022 (d). We measured top 14 cm integrated VWC, and horizontal 7.5 cm and 15 cm depth VWC at QC, and horizontal 10 cm and 20 cm depth VWC measurements at AB.

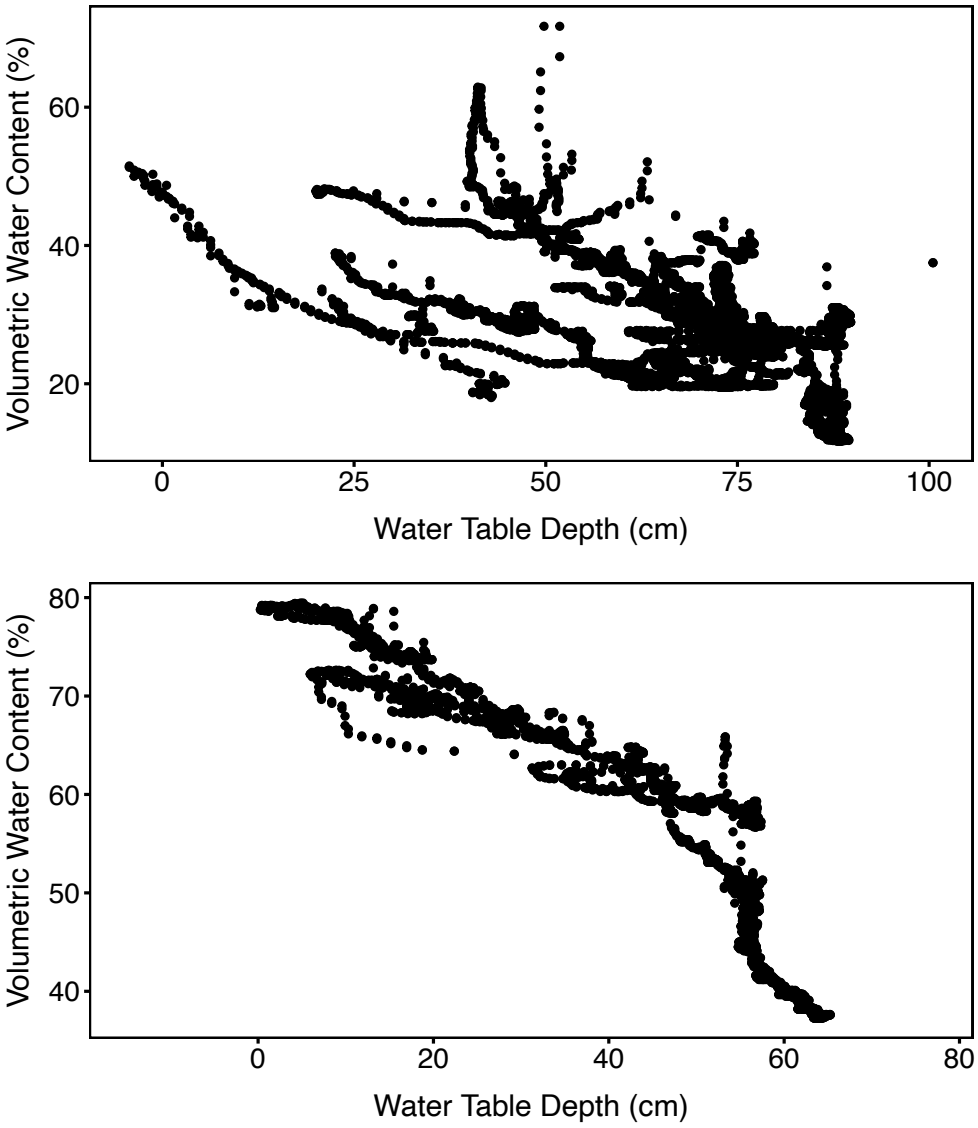


Figure B5 Soil water retention curves at QC (top) and AB (bottom). A 20 cm depth horizontal measurement of VWC and a top 14 cm integrated measurement of VWC were used at AB and QC respectively. A positive WTD indicates that the water table was below the peat surface.

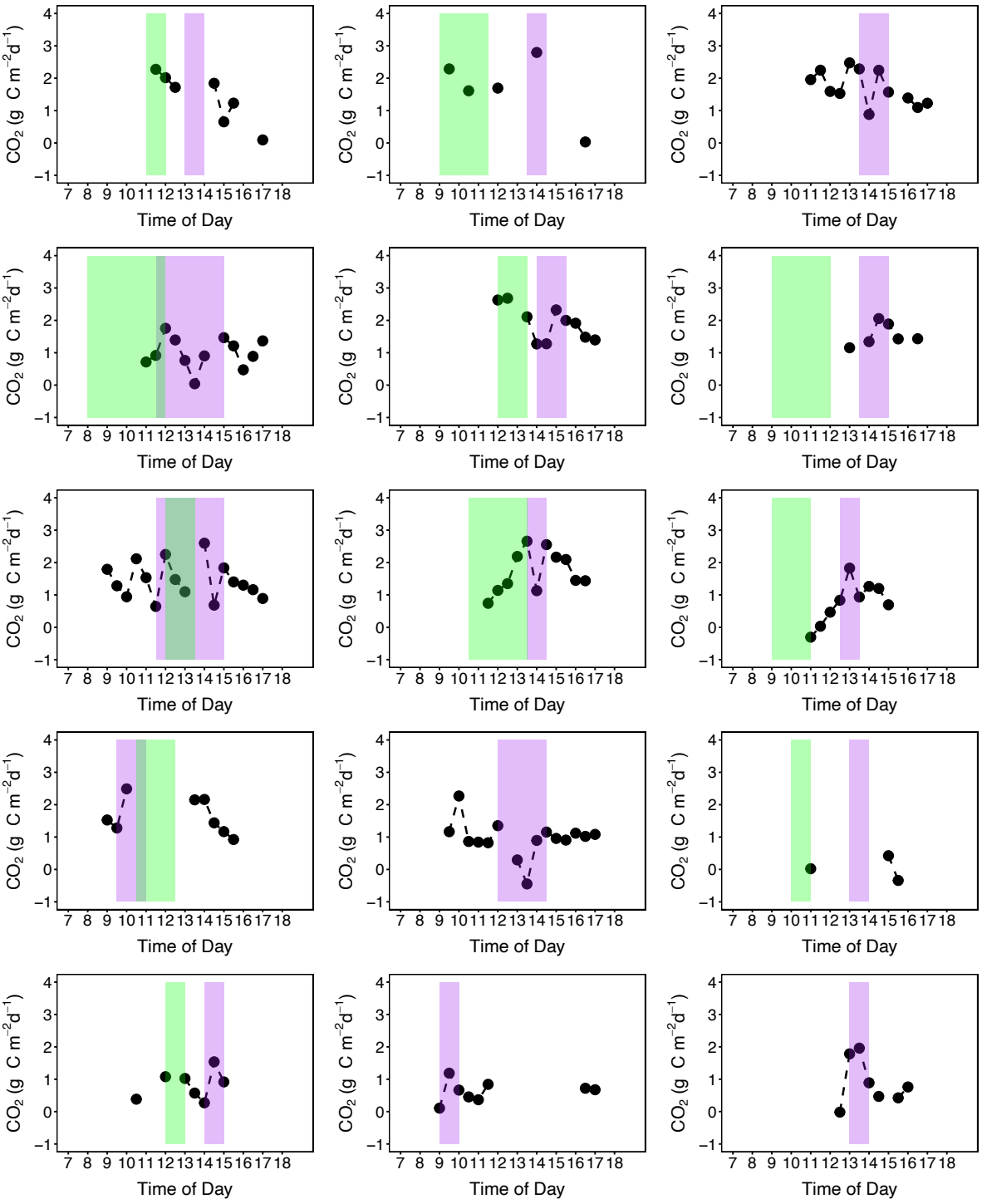


Figure B6 Plot of CO_2 emissions over the course of 15 days during the period of August 17th to September 23rd, 2022 at AB. The purple and green boxes indicate periods when harrowing and vacuum harvesting occurred respectively.

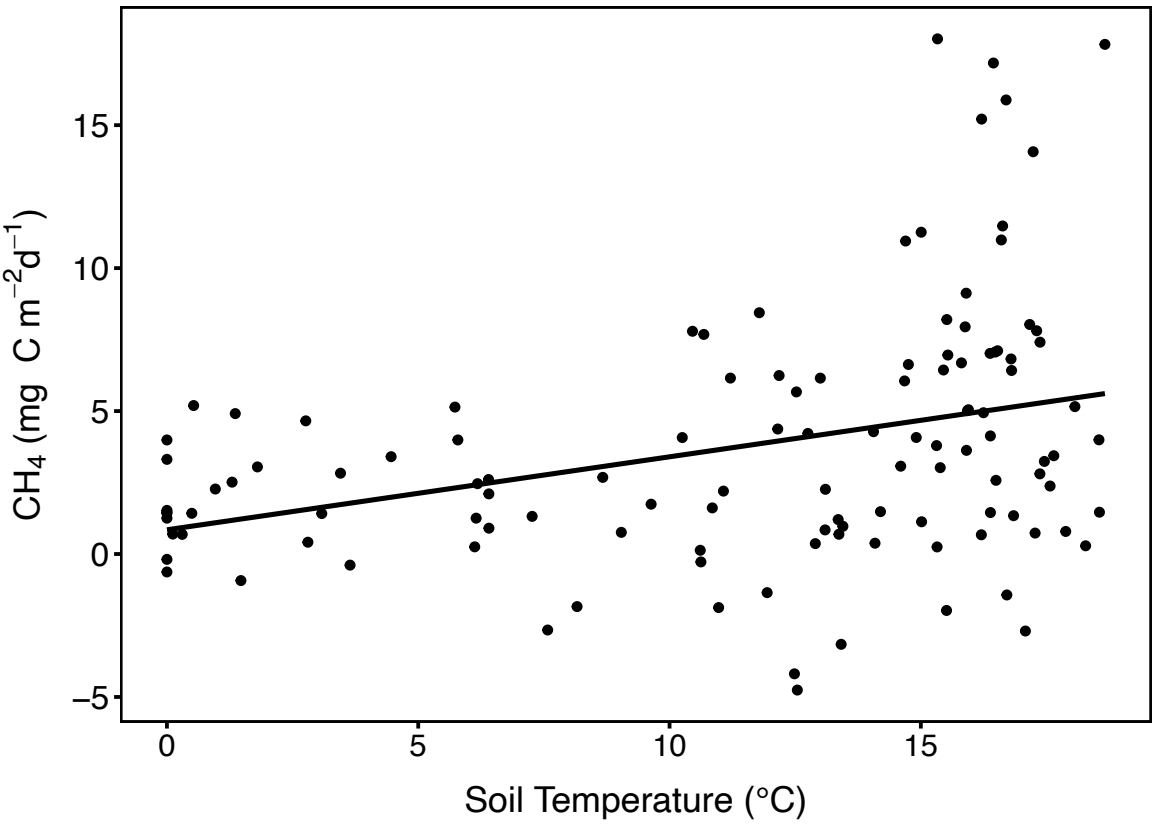


Figure B7 Scatterplot showing the effect of soil temperature at 20 cm depth on daily daytime (9 am to 5:30 pm) CH_4 emissions at AB in 2022. The linear regression shown is significant (see Section 3.4.3).

Table B1 Percent of daytime and nighttime CO_2 fluxes per month in each year at AB. A value of n.m indicates that data was not measured that month. Fluxes were categorized as daytime and nighttime when the incoming solar radiation was greater than 20 W m^{-2} and less than 20 W m^{-2}, respectively. In cases when the eddy covariance tower was not operating for the full month, only periods when it was operating were used for calculations.

	2020		2021		2022	
	Percent Daytime Fluxes	Percent Nighttime Fluxes	Percent Daytime Fluxes	Percent Nighttime Fluxes	Percent Daytime Fluxes	Percent Nighttime Fluxes
January	1.4	2.1	20.5	4.7	n.m	n.m
February	14.8	2.8	18.5	5.3	n.m	n.m
March	34.2	8.4	39.1	15.0	n.m	n.m
April	41.7	10.8	50.8	13.3	n.m	n.m
May	37.8	10.3	42.9	10.2	46.5	15.0
June	38.8	4.6	50.1	9.1	27.8	6.1
July	33.2	5.5	37.7	8.2	27.7	2.1
August	38.0	8.1	31.0	3.7	34.9	4.6
September	43.5	4.4	n.m	n.m	40.8	6.2
October	30.8	8.2	n.m	n.m	44.4	7.1
November	13.9	2.8	n.m	n.m	n.m	n.m
December	13.4	2.0	n.m	n.m	n.m	n.m

Table B2 Percent of daytime and nighttime CO_2 fluxes per month in each year at QC. A value of n.m indicates that data was not measured that month. Fluxes were categorized as daytime and nighttime when the incoming solar radiation was greater than 20 W $m^{-2}$and less than 20 W m^{-2}, respectively. In cases when the eddy covariance tower was not operating for the full month, only periods when it was operating were used for calculations.

	2020		2022	
	Percent Daytime Fluxes	Percent Nighttime Fluxes	Percent Daytime Fluxes	Percent Nighttime Fluxes
January	n.m	n.m	n.m	n.m
February	n.m	n.m	n.m	n.m
March	n.m	n.m	n.m	n.m
April	n.m	n.m	n.m	n.m
May	n.m	n.m	44.7	17.1
June	n.m	n.m	38.8	13.2
July	45.3	12.8	42.7	21.3
August	36.5	12.8	38.8	15.4
September	44.6	17.2	50.5	23.9
October	29.7	13.9	45.8	18.8
November	n.m	n.m	n.m	n.m
December	n.m	n.m	n.m	n.m

Table B3 Percent of daytime and nighttime CH_4 fluxes per month in each year at AB. A value of n.m indicates that data was not measured that month. Fluxes were categorized as daytime and nighttime when the incoming solar radiation was greater than 20 W $m^{-2}$and less than 20 W m^{-2}, respectively. In cases when the eddy covariance tower was not operating for the full month, only periods when it was operating were used for calculations.

	Alberta	
	Percent Daytime Fluxes	Percent Nighttime Fluxes
January	n. m	n. m
February	n. m	n. m
March	n. m	n. m
April	n. m	n. m
May	19.3	5.3
June	12.1	1.9
July	10.6	0.2
August	12.4	0.3
September	0	0
October	0	0
November	n. m	n. m
December	n. m	n. m

Table B4 Monthly 9 am to 5:30 pm local time average and median (in brackets) CO_2 emissions (g C m^{-2} d^{-1}) at AB and QC in 2020, 2021 and 2022. A value of n.m indicates that data was not measured that month. Letters indicated when fluxes are significantly different, based on a linear model on the effect of location and month on CO_2 fluxes.

	Quebec Site		Alberta Site		
	2020	2022	2020	2021	2022
March	n.m	n.m	0.13^{bc}	-0.01^{ab}	n.m
			(0.17)	(0.01)	
April	n.m	n.m	0.45^{efgh}	-0.19^{a}	n.m
			(0.43)	(-0.19)	
May	n.m	0.91^{klm}	0.34^{cdef}	0.16^{bcd}	0.50^{efghi}
		(0.87)	(0.42)	(0.25)	(0.45)
June	n.m	1.09^{m}	0.40^{cdefg}	0.22^{bcde}	0.68^{ghijk}
		(1.18)	(0.45)	(0.20)	(0.66)
July	0.74^{ijkl}	1.02^{lm}	0.71^{hijk}	0.76^{ijkl}	1.00^{lm}
	(0.75)	(1.07)	(0.68)	(0.78)	(1.01)
August	0.39^{cdef}	0.84^{jklm}	0.74^{ijkl}	0.43^{defgh}	1.00^{lm}
	(0.37)	(0.81)	(0.79)	(0.37)	(1.07)
September	0.55^{fghi}	0.56^{fghij}	0.54^{fghi}	n.m	0.84^{jklm}
	(0.54)	(0.56)	(0.49)		(0.90)
October	0.31^{cdef}	0.53^{fghi}	0.34^{cdef}	n.m	0.52^{fghi}
	(0.21)	(0.55)	(0.37)		(0.59)

Table B5 Monthly 9 am to 5:30 pm local time average and median (in brackets) CH_4 emissions (mg C m^{-2} d^{-1}) at AB 2022. Letters indicated when fluxes are significantly different, based on a linear model on the effect month on CH_4 fluxes.

Month	CH_4 Flux
May	6.54^{a} (7.17)
June	5.48^{a} (5.89)
July	7.73^{ab} (7.63)
August	9.13^{b} (9.35)

Table B6 Outputs of linear model on the effect of soil temperature at 20 cm depth (at AB) and 10 cm depth (at QC) on daily CO_2 emissions across the study years. The CO_2 data has been divided in wet periods (WTD < 25 cm), moderate periods (50 cm > WTD >25 cm), dry periods (75 cm > WTD > 50 cm) and very dry periods (WTD > 75 cm).

	Depth	DF	F	P	R^2
AB site					
	Wet	1, 11	2.62	0.12	0.12
	Moderate	1, 31	47.13	<0.0001	0.59
	Dry	1, 56	0.17	0.68	-0.02
QC site					
	Moderate	1, 18	42.68	<0.0001	0.69
	Dry	1, 92	24.34	<0.0001	0.20
	Very Dry	1, 97	9.95	0.0021	0.08

Appendix C

Table of Contents:

Figures C1-C2

Tables C1-C6

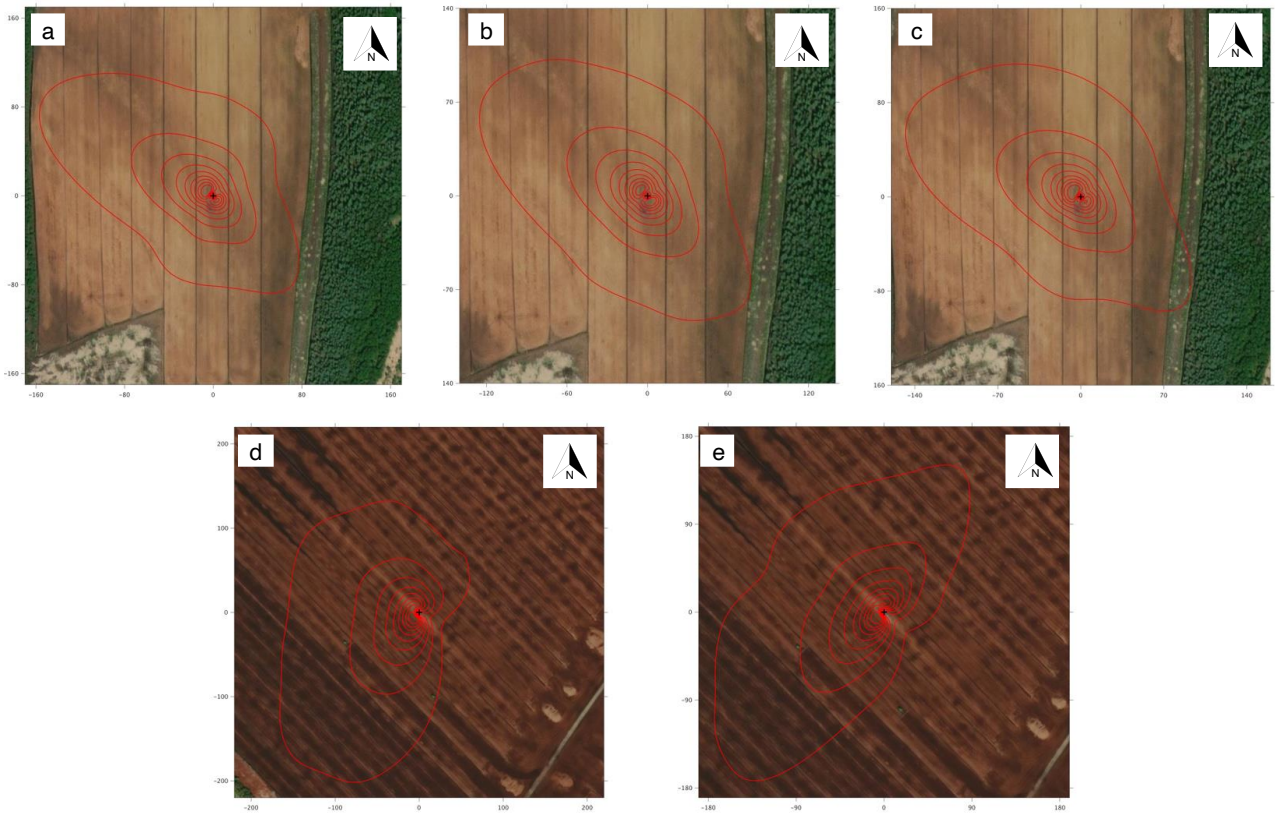


Figure C1 2D flux footprint probability contour maps of ET at the AB site in 2020, (a) 2021 (b) and 2022 (c), and at the QC site in 2020 (d) and 2022 (e). The plots were created using the online tool by Kljun and others (2015). The scale bars along the x and y axes are in meters. The outermost contour line represents the 90th percentile probability. The North Arrow is from FreeImages.com.

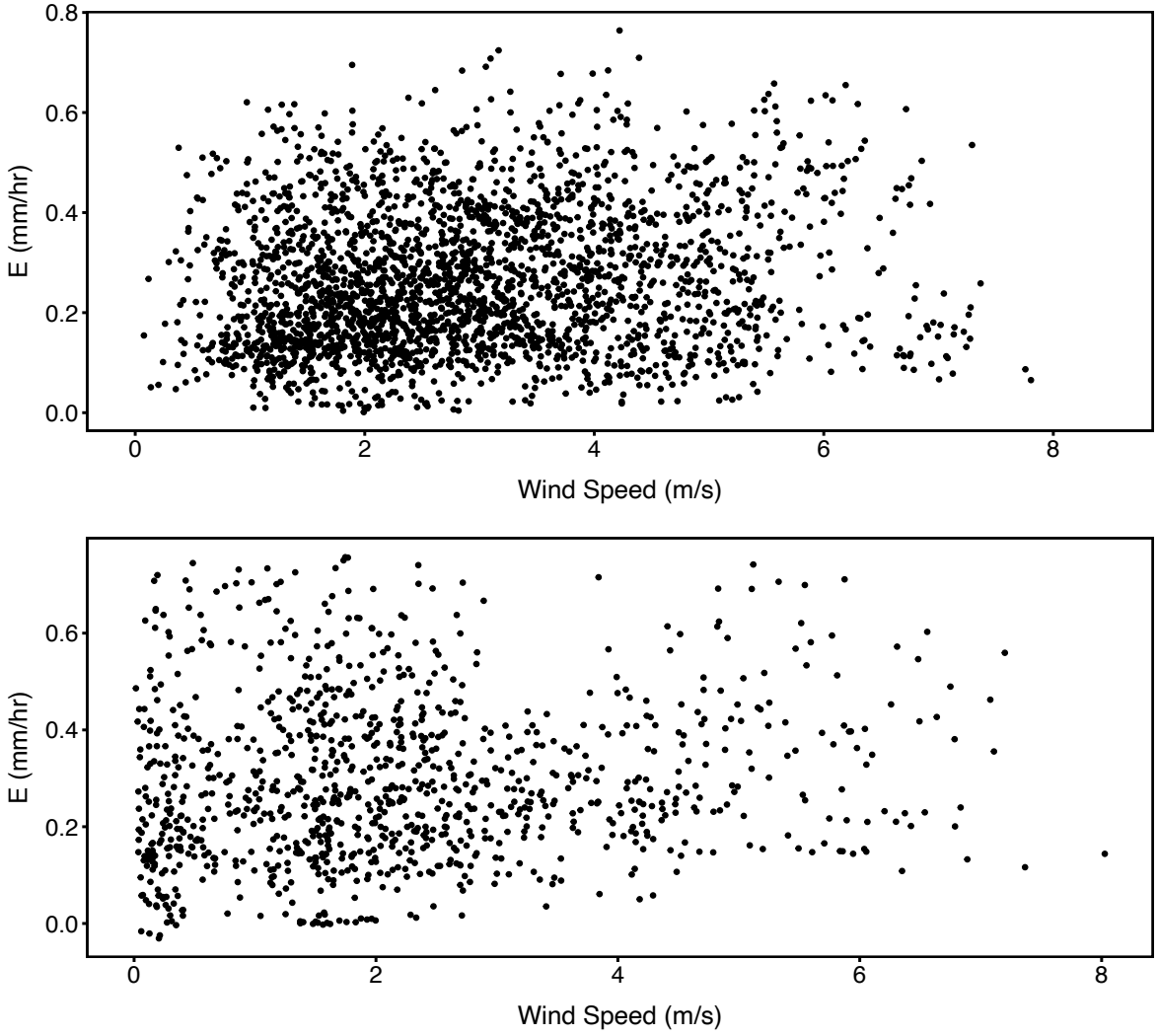


Figure C2 Relationship between wind speed and half-hourly evaporation rates (expressed in units of mm/hr) at AB (top) and QC (bottom).

Table C1 Energy balance closure at AB in 2022 and QC in 2020 and 2022. The energy balance was assessed three ways: the slope and R^2 of the linear regression of the energy balance ratio (EBR), the average energy EBR, and the energy balance residuals. See Section 4.3.3 for details of the calculations.

	AB	QC
Slope	0.83	0.76
R^2	0.94	0.95
EBR	0.58	0.66
Residuals (W m^{-2})	48	66

Table C2 Model results from using a rectangular hyperbola relationships [y= (a*b*VPD)/(a*VPD + b)] to understand the effect of VPD on dunny period (R_g > 500 W m^{-2}) half-hourly E rates at AB and QC across the study years. Data was divided in five WTD levels: 0–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80 plus cm below the surface. Values for parameter a and b are shown, along with the correlation between the observed and predicted data (corr), the model AIC, and the degrees of freedom (df), F statistic, p value and R^2 of the ANOVA run on the linear model of the observed versus predicted values. There were only 7 half-hourly periods when the WTD at QC was between 20 and 40 cm below the surface, and so no modelling was attempted, indicated by a value of not applicable (n.a.).

	a	b	corr	AIC	df	F-value	p-value	R^2
AB								
0-20 cm	0.104	0.719	0.61	-562	1, 321	186.6	<0.0001	0.37
20-40 cm	0.156	0.547	0.48	-519	1, 280	81.77	<0.0001	0.22
40-60 cm	0.113	0.299	0.36	-1532	1, 995	148.4	<0.0001	0.13
60-80 cm	0.157	0.240	0.25	-1692	1, 1008	69.74	<0.0001	0.063
QC								
20-40 cm	n. a.	n. a.	n. a.	n. a.	n. a.	n. a.	n. a.	n. a.
40-60 cm	5.27	0.585	0.27	-70	1, 44	3.56	0.06	0.054
60-80 cm	12.4	0.335	0.02	-338	1, 441	0.101	0.75	0.002
80-100 cm	7.20	0.287	0.23	-399	1, 267	15.66	<0.0001	0.052

Table C3 Results of linear model on the effect of half-hourly potential E (mm hr^{-1}) on actual E (mm hr^{-1}) at AB and QC at two WTD levels.

	WTD	DF	F-value	p-value	R^2
AB					
	< 40 cm	1, 170	597	<0.0001	0.78
	> 40 cm	1, 549	223	<0.0001	0.29
QC					
	< 60 cm	1, 307	1945	<0.0001	0.86
	> 60 cm	1, 1534	1475	<0.0001	0.49

Table C4 Results of ANOVA comparing the linear model on the effect of half-hourly potential E (mm hr^{-1}) on actual E (mm hr^{-1}) at AB and QC at two WTD levels with the 1:1 line.

	WTD	DF	F-value	p-value
AB				
	< 40 cm	1, 170	441	<0.0001
	> 40 cm	1, 549	673	<0.0001
QC				
	< 60 cm	1, 307	37	<0.0001
	> 60 cm	1, 1534	832	<0.0001

Table C5 Results of the ANOVA run on a linear mixed effects model of the effect of treatment, time since harrowing, and their interaction, on E rates in the three rounds of the lysimeter experiment. The lysimeter bucket was used as a random factor to account for the repeated measurements.

Effect	df	F-value	p-value
Round 1			
Treatment	1, 24	18.43	<0.0001
Time	2, 24	6.57	0.02
Time x Treatment	2, 24	1.53	0.24
Round 2			
Treatment	1, 8	1.14	0.32
Time	2, 16	23.38	<0.0001
Time x Treatment	2, 16	12.36	0.0006
Round 3			
Treatment	1, 8	0.13	0.72
Time	2, 16	35.74	<0.0001
Time x Treatment	2, 16	14.32	0.0003

Table C6 Average monthly total daytime (R_g > 20 W m^{-2}) E (mm) at AB and QC in 2020, 2021 and 2022. A value of n. m. indicates that measurements were not made that month.

	AB			QC	
	2020	2021	2022	2020	2022
March	0.4	1.0	n. m.	n. m.	n. m.
April	2.0	1.4	n. m.	n. m.	n. m.
May	3.6	2.5	3.0	n. m.	4.1
June	4.6	2.3	3.1	n. m.	4.7
July	4.4	2.3	3.2	2.9	4.6
August	2.6	1.8	2.2	2.4	2.9
September	1.7	n. m.	1.4	2.6	2.0
October	0.6	n. m.	0.7	1.7	1.9