An analysis of carbon dioxide and methane exchange at a post-extraction, unrestored peatland in Eastern Québec

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Abstract

Peatlands, in their pristine state, are important long-term sinks of carbon. The extraction of peat for agricultural purposes or for biofuel leads to a drastic shift in the carbon dynamics. Additionally, the change in environmental conditions after extraction could also allow for invasive species to establish and spread across the peatland. Many studies have shown the benefits and advantages of various restoration management practices, but few studies have explored the carbon exchange from unrestored peatlands. This study analyzed the methane (CH_4) and carbon dioxide (CO_2) fluxes from a post-extraction, unrestored peatland in Eastern Québec at both the plant community scale, using chamber methods, and at the ecosystem scale, using eddy covariance techniques. Results indicate that the site is an overall source of carbon rather than a sink, releasing an annual total of 153 g C m^{-2} and 241 g C m^{-2} in CO_2 emissions for 2014 and 2015, respectively and an average annual total of 1 g C m^{-2} in CH_4 emissions. Furthermore, *Phragmites* and *Typha,* both invasive species, have established themselves in the ditches and are sources of methane; partially explaining why the peatland's net carbon flux to the atmosphere has changed. Further research was conducted to examine these findings, including a vegetation survey that provided insight into the relative contributions of each plant community to the total CH_4 and CO_2 fluxes at the peatland site. Results from this study will help managers assess the importance of post-extraction peatland restoration, by comparing the differences in CO_2 and CH_4 exchange between restored and unrestored peatlands.

Résumé

Les tourbières, dans leur état d'origine, sont d'importantes réserves à long terme du carbone. L'extraction de la tourbe utilisée pour les usages agricoles ou pour les biocarburants conduit à un changement radical dans la dynamique du carbone. En outre, le changement des conditions environnementales après l'extraction pourrait également permettre aux espèces envahissantes à s'établir et se répartir dans la tourbière. De nombreuses études ont montré les retombées et les avantages de la restauration, mais peu d'études ont exploré l'échange du carbone dans les tourbières sans la restauration. Cette recherche a analysé les flux de méthane (CH_4) et de dioxyde de carbone (CO_2) d'une tourbière après l'extraction et sans restauration, dans l'Est du Québec, en utilisant les "Chamber methods" à l'échelle de la communauté végétale et en utilisant des techniques "Eddy covariance" à l'échelle de l'écosystème. Les résultats indiquent que le site est une source globale de carbone plutôt qu'un puit, libérant un total annuel de 153 g C m^{-2} et 241 g C m^{-2} en émissions de CO_2 en 2014 et 2015 respectivement et un moyen total annuel de 1 g C m^{-2} en émissions de CH_4. En outre, *Phragmites* et *Typha,* des espèces envahissantes, se sont établies dans les fossés, et sont des sources de CH_4; ce qui explique, en partie, pourquoi les flux nets de carbone de la tourbière ont changé. D'autres recherches ont été mené pour examiner ces résultats, incluant une enquête de la végétation, qui a fourni une perspicacité des contributions relatives de chaque communauté végétale aux flux totaux de la tourbière. Les résultats de cette recherche aideront les gestionnaires à évaluer l'importance de la restauration après l'extraction des tourbières, en comparant les différences de CO_2 et de l'échange de CH_4 entre les sites restaurés et non-restaurés.

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Contribution of Authors

This thesis is presented in the traditional thesis format.

An analysis of carbon dioxide and methane exchange at a post-extraction, unrestored peatland in Eastern Québec

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This thesis is the original work of Tracy Rankin with a few exceptions. The collection of the field and tower data were carried out by the author, Tracy Rankin. Dr. Luc Pelletier performed post-processing, cleaning and gap-filling procedures on the raw CO_2 and CH_4 tower flux data. All data analysis was performed by Tracy Rankin. Dr. Ian Strachan and Dr. Maria Strack provided expert advice and financial support for the project. Dr. Ian Strachan also contributed to the editing process of the thesis as well.

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Chapter 1. Introduction

1.1 Peatlands and Greenhouse Gases

Northern peatlands play a significant role in the global carbon cycle, covering about 3% of the earth's surface, but storing up to 30% of the world's carbon (C) (Gorham 1991). Following the last glaciation, Canadian peatlands have accumulated C at an average rate of 18-30 g m^{-2} yr^{-1} (Clymo 1984) with Frolking et al. (2010) finding C accumulation rates that reached as high as 35 g m^{-2} yr^{-1} in a temperate bog in Eastern Ontario. Natural peatlands are therefore considered long-term sinks of carbon.

Peatlands take up carbon dioxide (CO_2) through photosynthesis by the surface vegetation and release CO_2 through respiration and peat decomposition. On average, photosynthesis is greater than respiration and decomposition rates, where low temperatures and anoxic conditions both lead to the favoring of biomass production over decomposition (Blodau 2002; Humphreys et al. 2006). Another important greenhouse gas released from peatlands is methane (CH_4). CH_4 is produced under the water table surface, in the anaerobic zone. Methane gas is released from the surface of peatlands to the atmosphere by the process of methanogenesis through various means of transport; molecular diffusion, ebullition, and by plant mediated transport through the aerenchymous tissue of vascular plant species. It has been estimated that northern peatlands contribute between 40 and 155 Tg annually to global CH_4 emissions (Neef et al. 2010; Turetsky et al. 2014; Waddington and Roulet 1996). Although the atmospheric concentration of CH_4 is lower than that of CO_2, it has a global warming potential twenty-five times more potent.

Anthropogenic disturbances (e.g. peat extraction) can drastically alter the carbon dynamics of a peatland; consequently, they may change from a sink of carbon to a source (Strack and Zuback 2013). Although CH_4 emissions decrease following the extraction of peat due to an increase in the aerobic zone (Waddington and Price 2000), if left unrestored, extracted peatlands can remain a persistent source of carbon, releasing a large amount of CO_2 to the atmosphere (Waddington et al. 2002).

1.2 Extraction and Restoration of Peatlands

The extraction of peat has increased over the last decade. An estimated 28,000 hectares of peatlands have been extracted in Canada (mainly for horticultural purposes), which is an increase of 4000 hectares since 2010 (Canadian Sphagnum Peat Moss Association 2016). In other countries, like Russia and Sweden, peat is also extracted for energy usage or biofuel (Peat and Peatlands 2016). The negative impacts that follow the extraction of peat are an increase in organic matter decomposition and a decrease in soil moisture. Plant species that are adapted for wetter areas, such as *Sphagnum* mosses, that would normally uptake a large amount of CO_2, are not able to re-establish as easily in the resulting conditions (Glatzel et al. 2004; Waddington et al. 2010; Waddington et al. 2002).

The carbon and water budgets of peatlands are intricately linked (Waddington et al. 2015). As the water table drops beyond a certain depth, plant productivity substantially decreases, decomposition increases and the peatland becomes a source of carbon rather than a sink (Seters and Price 2001). This emphasizes the need for a balance between the economic benefits of peat extraction and minimizing the amount of greenhouse gases released to the atmosphere. One way of achieving this balance is through the restoration of extracted peatlands.

In Canada, over 6400 hectares of extracted peatlands have been restored in recent years as an attempt to restore the carbon sink function (Graf et al. 2012; Tarnocai et al. 2011). Some of the restoration efforts have been successful where peatlands have returned to a state of carbon-sink and high biodiversity in fewer than 20 years (Lucchese et al. 2010). The restoration process includes the blocking of the ditches, which causes a rise in the water table. This allows for more water to be available to a variety of plants that lack complex root structures, thereby providing the potential for an increase in biodiversity and a greater potential for increased uptake of CO_2.

If a peatland is extracted, but remains unrestored, it may never return to its initial state before extraction, and will continue to be a persistent source of CO_2 (Lavoie et al. 2003; Waddington et al. 2002). Furthermore, if the ditches are not blocked, the new hydrological and nutrient conditions following extraction can lead to the establishment of invasive species, which can outcompete the other vegetation and spread across the peatland (Strack et al. 2009). Consequently, restoration efforts are needed to ensure the re-establishment of the desired moss species native to peatlands, and to restore the carbon sink function.

1.3 Research Question and Objectives

Previous studies have analyzed the benefits and advantages of various restoration management practices from extracted peatlands (Girard et al. 2002; Lavoie et al. 2003; Waddington et al. 2010), but the carbon exchange that results from unrestored peatlands is not well understood. To the best of our knowledge, continuous measurements of carbon dioxide and methane exchange from unrestored peatlands, where no restoration efforts have been made (i.e. no blocking of the ditches), or where there has been no spontaneous regeneration of *Sphagnum* species, have yet to be obtained. Furthermore, CO_2 and CH_4 flux measurements from unrestored

peatlands reported in previous studies have almost exclusively been obtained at the plant community level, using chamber methods. Therefore, there is a need to determine the fluxes of the two dominant greenhouse gases at both the ecosystem and plant community scales from unrestored peatlands, as this will provide the trajectory that a peatland may follow if no restoration efforts are implemented following extraction. The measurements would also provide the peat industry with a baseline case with which to compare the results from restored peatlands, thereby depicting the true net benefit (from a carbon uptake perspective) of implementing restoration practices.

Thus, the overarching question is, "What is the net carbon exchange to the atmosphere of a post-extraction, unrestored peatland?" To answer this question, we selected a peatland in eastern Quebec, where extraction ended in 1999 and where no restoration efforts have been implemented.

The study has the following objectives:

1)   To continuously measure CO_2 and CH_4 fluxes at the ecosystem scale over two years and,

2)   To determine which plant communities contribute to CO_2 and CH_4 exchange.

We used the eddy covariance technique to measure the net CO_2 and CH_4 fluxes from the peatland (Objective 1). Data were collected from 2014-2015 and results were compared with a restored peatland of the same age in the same area. Environmental variables were measured to determine the controls on the inter-annual variability in carbon dioxide and methane exchange.

While the tower fluxes provide an ecosystem-scale measurement of the net carbon exchange, plant community scale measurements help determine the processes and mechanisms controlling carbon uptake and release (Objective 2). Static chamber measurements were made throughout

the snow-free seasons over the same two years (May-September, 2014 and 2015), to determine how shifts in vegetation and peatland function can affect the hydrology-carbon feedback. It is anticipated that the results from this study will help managers assess the importance of post-extraction peatland restoration by comparing the differences in carbon dioxide and methane exchange between restored and unrestored peatlands.

Chapter 2: Literature Review

2.1 Peatlands and Greenhouse Gases

Wetland ecosystems cover roughly 6-9% of the earth's terrestrial surface. They are present in multiple regions across the globe, but are most abundant in the boreal and subarctic regions, where temperatures are cooler and precipitation amounts are favourable (Baldwin and Batzer 2012). In Canada, peatlands comprise 76% of the total area of wetlands, which translates into a surface area of roughly 11×10^7 hectares. The widely accepted definition of a peatland is, "a wetland on which extensive organic material has accumulated" (Daigle and Gautreau-Daigle 2001). This is a very broad definition and since peatlands have an extensive distribution among various environmental regimes, the development of a standard classification scheme was necessary (Zoltai and Vitt 1995).

Most peatlands can be categorized as either bogs or fens, according to their geographical location, hydrology, chemical and physical characteristics (Mitsch and Gosselink 1993). Ombrotrophic bogs receive water inputs solely from precipitation, and consequently tend to be more acidic than fens, where pH values range between 3 and 5. Bogs are often dome-shaped due to the accumulation of peat over thousands of years, which allows the incoming water to flow away from the peatland into the surrounding areas; whereas fens usually reside in areas of lower elevation (e.g. valleys) (Gorham 1991). Fens are influenced by drainage flows through the system and hence receive water inputs from both precipitation and from the groundwater below. This causes increased oxygenation of the organic substrate which enhances decomposition, explaining the smaller thickness of fens compared to that of bogs (Baldwin and Batzer 2012). Furthermore, the pore water and peat in the deeper layers of fens tend to take on the

characteristics of the underlying bedrock and calcareous soil, translating into fens being less acidic, with pH values ranging between 5 and 7 (Holden 2005). Fens and bogs also tend to be host to different species of vegetation that are better adapted to the different nutrient and hydrological conditions (Gorham 1991).

Peatlands offer a variety of valuable ecosystem services (Ramsar 2006). They represent areas of high biodiversity, being host to over 600 species of flora and fauna. They also generally have a high water quality (Waddington et al. 2015), where they possess the ability to decrease and even remove pollutants from the water and the atmosphere, which provides them with great socio-ecological and environmental value. Peatlands also possess significant economic value, where peat moss is often extracted for horticultural purposes or as an alternative energy source (Kennedy and Mayer 2002).

The various characteristics of peatlands have been examined in multiple studies (Andersen et al. 2009; Van der Valk 1981; Kennedy and Mayer 2002). However, most of the literature tends to be focused on peatland ecosystem function, especially their ability to sequester a large amount of C in the soil (Lafleur 2009). Dead plant material in undisturbed peatlands does not decompose as rapidly as it accumulates as peat; making natural peatlands long term sinks of carbon. However, on shorter time scales, a natural peatland may be a source or a sink of carbon depending on the weather conditions of a given year (Roulet et al. 2007; Strack 2008). The carbon and water budgets of peatlands are intricately linked (Waddington et al. 2015). Disturbances that impact water storage and flows such as climate change or anthropogenic activities (e.g. peat extraction) lead to changes in peatland carbon cycle processes.

The two dominant greenhouse gases exchanged with the atmosphere from the surface of peatlands are carbon dioxide (CO_2) and methane (CH_4), both of which contribute significantly to global warming (Kayranli et al. 2009). The release of greenhouse gases is exacerbated following a disturbance. The extent of this shift in carbon cycle processes and the resulting carbon exchange depends highly on various environmental factors such as water table depth and temperature. Therefore, there is a need for a better understanding of the processes and environmental conditions that drive the variability in carbon exchange in peatlands in order to determine how such disturbances can alter the hydrology-carbon feedback.

2.2 CO_2 Exchange

The difference between Gross ecosystem photosynthesis (GEP) and Ecosystem respiration (ER) of a vegetated surface equals the Net ecosystem exchange (NEE) of a peatland. During the growing season, GEP is higher than ER resulting in an uptake of CO_2. Average NEE values from northern bogs range between -3 g C $m^{-2}d^{-1}$ and -8 g C $m^{-2}d^{-1}$ (Lafleur 2009; Pelletier et al. 2011), where the sign convention is such that negative values of NEE represent uptake of CO_2 and positive values of NEE represent emissions. CO_2 exchange depends on many factors including light intensity or Photosynthetically active radiation (PAR), water table depth and temperature, with water table being the most prominent driver (Lafleur et al. 2005; Sonnentag et al. 2009).

2.2.1 Photosynthesis

Carbon dioxide is taken in by the vegetation through the process of photosynthesis. Uptake increases with light intensity until it reaches saturation (between 1000 μmol m^{-2} s^{-1} and

1500 μmol m^{-2} s^{-1}). The relationship between photosynthesis and light intensity may differ temporally (across years and within seasons) as well as with the type of ecosystem and vegetation structure. For example, Frolking et al. (1998) found that fens tend to have higher CO_2 emissions than bogs, where fens show almost 15 g C $m^{-2}d^{-1}$ higher emissions for the same light intensity of 1800 μmol m^{-2} s^{-1}.

Air temperature and soil moisture are also important controls of photosynthesis. Petrone et al. (2001) found that photosynthesis, measured by plant productivity, was highest during the growing season when temperatures were high, and Moore (1989a) found that a large portion of variability in Net primary productivity (NPP) can be attributed to higher mean annual temperatures. Strack and Price (2009) found that moist sites were sinks of CO_2 and dry sites were sources of CO_2. However, the soil moisture to photosynthesis relationship depends highly on the type of vegetation, which react to changing water levels differently. For example, in an ombrotrophic bog in Eastern Ontario, Bubier et al. (2003) found higher CO_2 uptake in the drier summer for the shrub species, but a same or lower uptake for the sedge species in the same drier summer.

2.2.2 Ecosystem Respiration

ER (CO_2 emitted to the atmosphere) is highly dependent on the vegetation type and community composition of the landscape. ER is composed of both autotrophic respiration by plant parts (including roots and belowground biomass) and heterotrophic respiration by microbial bacteria in the soil and decomposition of organic matter (Lafleur 2008; Blodau 2002). It has been shown that water table and temperature affect both the decomposition rates and respiration rates from the plants (Lafleur et al. 2005; Moore 1989). For example, Bubier et al.

(1998) found that lower water tables corresponded with higher CO_2 emissions. Peat temperature seems to also have an effect on CO_2 emissions. Moore and Dalva (1993) found that with an increase in temperature from 10 °C to 23 °C, CO_2 emissions had increased by a factor of 2.4.

2.3 CH$_4$ exchange

Northern peatlands contribute between 40 and 155 Tg annually to global CH_4 emissions (Neef et al. 2010; Turetsky et al. 2014; Waddington and Roulet 1996). CH_4 emissions can vary both spatially among different peatlands and temporally within the same peatland (Moore et al. 1998). Many studies suggest that the main driver of variability in CH_4 emissions is a change in water table position, where a higher water table (closer to the surface) is usually followed by an increase in CH_4 emissions (Roulet et al. 1992; Seters and Price 2001). Other controls include air temperature and peat temperature (Mahmood and Strack 2011; Pelletier et al. 2007), vegetation biomass, net primary production and a shift in vegetation community (Mahmood and Strack 2011; Whalen 2005).

2.3.1 CH$_4$ Production

CH_4 is produced under the water table surface (in the anaerobic zone) by bacteria that act as methanogens, by either using acetate as a substrate to produce methane gas or by reducing CO_2 using hydrogen gas as an electron donor. The latter form of methanogenesis is found in deeper layers of peat where labile organic carbon is not as abundant (Blodau 2002; Edwards et al. 1998; Whalen 2005). The process of acetate as a substrate is increased in more acidic environments, such that pH plays a role in CH_4 production as well (Blodau 2002; Lafleur 2008). Other controls that drive a variation in CH_4 production are changes in water table depth, which

changes the thickness of the anoxic and oxic peat layers, peat temperature and surface vegetation (Roulet et al. 1992; Bellisario et al. 1999). Surface vegetation plays a role in the variation of methane production by providing labile C through root decay, enhancing the acetate substrate process.

2.3.2 CH_4 Consumption

Not all of the methane gas produced by the methanogens is released to the atmosphere. Some of the CH_4 is consumed by the methanotrophs in the aerobic zone for use in growth and maintenance. This is where oxygen (O_2) plays a crucial role. All methanotrophs have the monooxygenase enzyme which can break the bonds in O_2 molecules. This reduces one oxygen atom to a water molecule (H_2O) and one to CH_3OH using the incorporated methane. Oxygen can also serve as an electron acceptor in the process of methane oxidation to form CO_2 gas. The highest activity of methanotrophs occurs closest to the water table, which translates into higher CH_4 fluxes from areas where the water table is closer to the surface (Blodau 2002; Edwards et al. 1998; Whalen 2005). Since changes in water table depth often mirror changes in the topography of the landscape, mictrotopography plays a significant role in CH_4 exchange in northern peatlands (Lafleur 2008; Bubier and Crill 2003; Bubier 1995). Furthermore, it has been shown that temperature can also affect the amount of methanotrophic activity as well (Lafleur 2008; Bubier 1995), but water table seems to be the more prominent driver.

2.3.3 CH_4 Transport Mechanisms

CH_4 can be transported from beneath the surface to the atmosphere in three ways: by molecular diffusion, ebullition, and by plant-mediated transport (Figure 2.1).
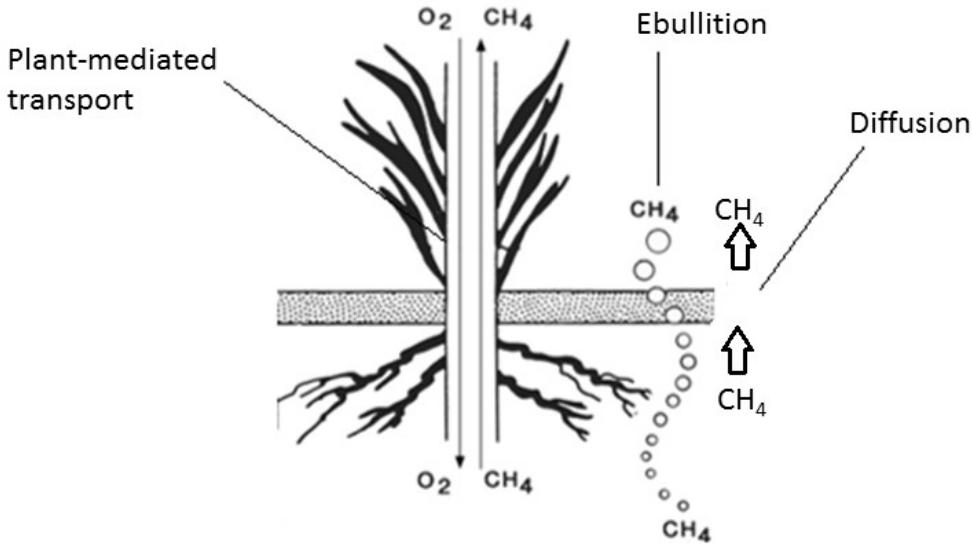
Figure 2.1. Production and efflux of CH₄ from a peatland surface (Adapted from Couwenberg and Fritz 2012).

Molecular diffusion is driven by a gradient in CH_4 concentration between the anaerobic layer, where CH_4 gas is produced, and the surface. The flux of CH_4 transported to the surface by diffusion can be described as

$$J = -D * \frac{dC}{dz} \; , \qquad (1)$$

where J is the diffusive flux, D is the diffusion coefficient, C is the CH_4 concentration and z is the depth of peat. Molecular diffusion is an important transport mechanism to consider because it affects the rate of CH_4 consumption by methanotrophs (Lai 2009).

Ebullition refers to the release of methane gas to the atmosphere by gas bubbles. Gas bubbles are created when the partial pressure of the dissolved gases in the pore water is greater than the hydrostatic pressure in the peat. The bubbles get attached by adhesive forces to the pore walls in the peat, and start to grow. The accumulation of the growing gas bubbles increases the

pressure and when a certain pressure threshold is met, the gas bubbles are released all at once. Ebullition can be triggered by a change in atmospheric or hydrostatic pressure, or a rise in temperature. Ebullition happens very rapidly and large amounts of CH_4 gas can be released into the atmosphere in a short period of time (Whalen 2005).

Many vascular plant species contain aerenchyma, which act as internal gas-space ventilation systems. They allow O_2 to reach the roots of the plants submerged under the water table and serve as direct conduits of methane gas from the roots and rhizomes of the plants to the atmosphere. A difference in temperature or water vapour pressure between the gas spaces in the aerenchyma and the atmosphere creates a pressure gradient. Plant-mediated transport is therefore driven by convective flow within the plants themselves, and CH_4 is released to the atmosphere in large amounts with minimal resistance, while also bypassing the aerobic, methane-oxidizing peat layer (Lai 2009; Shannon et al. 1996).

2.4 Extraction and Restoration Practices

One of the main anthropogenic disturbances to peatlands is the extraction of peat (e.g. for horticultural purposes or for biofuel). In this process, the hydrology-carbon feedback in peatlands is drastically altered. While the extraction of peat has high economic value, large amounts of greenhouse gases are released to the atmosphere as a result. The extraction of peat has increased in the last few decades. The most common method of extraction used in Quebec, where 10% of Canadian peatlands are located (Daigle and Gautreau-Daigle 2001), is "vacuum harvesting" (Poulin et al. 2005). Several meters of peat may be removed in the process. A detailed description of the "vacuum harvesting" method is described in Graf et al. (2012). Briefly, the

method consists of first removing the surface vegetation and creating ditches that drain the site, facilitating the peat removal process. Then the extraction of peat is carried out, often for many decades, until the remaining peat is no longer suitable for industry needs.

Another extraction technique is the "block-cut" method, but this is more labour intensive and time consuming as the peat is removed by hand. Therefore, it is not as widely used as the "vacuum-harvesting" method, especially with the increasing demand for peat (Girard et al. 2002).

Efforts to remediate the peatland following extraction hope to restore peatland functioning in terms of hydrology and carbon exchange. A typical restoration technique following extraction by the "vacuum harvesting" method is described in Lavoie and Rochefort (1996). Briefly, restoration includes rewetting the soil by blocking the ditches, which causes a rise in the water table. A few studies have seen spontaneous re-vegetation occur in post-extracted peatlands, whereby *Sphagnum* mosses spread on their own after only rewetting the site (Girard et al. 2002; Lavoie and Rochefort 1996), thereby eliminating the need for any further restoration efforts. However, spontaneous regeneration is more likely to occur in peatlands that have been extracted using the "block-cut" method. Peatlands extracted through the "vacuum-harvesting" technique, which uses heavy machinery, are less likely to revegetate on their own (Poulin et al. 2005). Restoration therefore includes the transplanting of *Sphagnum* moss seeds from a donor site. The peatland would then be covered in a layer of mulch to protect the newly transplanted seeds from desiccation, and to facilitate plant growth (Rochefort et al. 1990). It has been suggested that filling in the ditches with peat, which provides a flatter surface, may help facilitate the growth of mosses as well (Graf et al. 2012).

2.5 A Focus on the Species of Interest

According to Graf et al. (2012), the new environmental conditions following extraction may be favourable for invasive species (*Phragmites australis*, also known as Common Reed, and *Typha latifolia*, also known as Cattail) to establish. *Typha* and *Phragmites* are more commonly found where the water table is constantly above the surface (e.g. marshes). Consequently, the literature focuses more on these wetter sites (Rice et al. 2000; Christensen et al. 2009). However, studies that measured the productivity of *Typha* and *Phragmites* where there was a drawdown or a large fluctuation in the water table (Christensen et al. 2009; Galinato and Van der Valk 1986), show that the hydrological niche in which these invasive species can thrive is much more expansive and that both invasive species can adapt to changing water table positions better than other plant species native to peatlands. Although they can both live in water that is between 50 and 100 cm above the surface, they can also survive in environments where the water table is 60 cm *below* the surface (Walker 1965; Zhao et al. 2013). However, Squires and Van der Valk (1992) found that *Typha* may be slightly more restrictive than *Phragmites*, and do not seem to be present where water tables drop more than 30 cm below the surface.

The invasive species also seem to have a distinct range of temperatures in which they can survive. Galinato and Van der Valk (1986) suggest that both *Typha* and *Phragmites* thrive best in air temperatures ranging from 20-30 °C, whereas other studies suggest that *Phragmites* can survive in a slightly larger range of temperatures (10-30 °C) than *Typha* (Ekstam & Forseby, 1999).

The invasive species play an important role in the biodiversity of peatlands. A post-extraction, unrestored peatland is devoid of natural vegetation that would normally occupy the

space, and if the ditches are not blocked, the site is also prone to fluctuating water table positions (Rochefort et al. 1990; Graf et al. 2012). The ditches provide a wetter environment than the surrounding drier peat fields, allowing the invasive species to establish themselves. Then, with their competitive nature and their ability to adapt more easily to the fluctuating water levels, they can outcompete the other vegetation and spread across the peatland (Van der Valk 1981; Seabloom et al. 2001; Tulbure et al. 2007).

The presence of *Typha* and *Phragmites* can also affect the carbon dynamics of a peatland. They possess aerenchymous tissue that act as direct conduits of methane gas (plant-mediated transport) to the atmosphere through convective flow through the plant itself, often bypassing the anoxic peat layer without being oxidized (Grünfeld and Brix 1999; Joabsson et al. 1999). Values of CH_4 release from these species range between 5 mg CH_4 m^{-2} hr^{-1} and 12 mg CH_4 m^{-2} hr^{-1} (Arkebauer et al. 2001; Günther et al. 2014). Ditches themselves are already considered "methane hotspots" (Roulet and Moore 1995). Therefore, the establishment of invasive species in ditches increases the overall methane emissions from unrestored, extracted peatlands.

CH_4 emissions from *Phragmites* and *Typha* are reported numerous times in the literature, but the effect of these plants on the CO_2 dynamics of a peatland is not well documented. It has been estimated that CO_2 uptake from *Phragmites* and *Typha* can range between -2 g C $m^{-2}d^{-1}$ and -4 g C $m^{-2}d^{-1}$ (Zhou et al. 2009). However, due to the lack of studies depicting the CO_2 exchange from *Typha*- and *Phragmites*-dominated sites, there is a concern that the uptake of CO_2 may not be able to overcome the CH_4 and CO_2 release to the atmosphere, especially following extraction. Therefore, further investigation is needed to explore whether the presence of these species directly translates into the peatland being a source of carbon to the atmosphere.

2.6 Measuring the Dominant Greenhouse Gases

Measurements of greenhouse gas emissions can be measured at multiple scales. This study will focus on fluxes obtained at the ecosystem level, using eddy covariance techniques, and at the plant community level, using static chambers. Both techniques have various advantages and disadvantages. Therefore, it has been suggested that measurements be obtained at both scales to obtain a clear understanding of the hydrology-carbon feedback in a peatland.

2.6.1 Ecosystem Scale Measurements

The eddy covariance (EC) technique is a widely used method of obtaining direct and continuous measurements of energy fluxes and trace gas exchange (e.g. CO_2, H_2O and CH_4) from peatlands at the ecosystem level, without disturbing the surface vegetation (Rinne et al. 2007).

The EC technique is described in detail by Burba and Anderson (2010) and can be summarized as follows. Horizontal air flow across an ecosystem is comprised of multiple eddies (rotating parcels of air) of varying sizes. The eddies transport the air parcels both vertically and horizontally via convection, which occurs due to a gradient in air density. Vertical convection depends on surface roughness and heating as the air flows across the surface. Therefore, EC tower instruments are placed above the roughness sublayer, and within the surface boundary layer, where turbulence is prevalent. Each air parcel has a given concentration of trace gas, temperature and humidity. If these characteristics and the vertical wind speed are known, then the flux is calculated as the covariance between the concentration of the trace gas and the vertical wind speed of the eddies (Burba and Anderson 2010). Under ideal conditions, turbulent flux can be calculated as

$$F = \overline{\rho_a} * \overline{w'c'} \ , \tag{2}$$

where ρ_a is the mean air density, w' is the departure from the mean vertical wind speed and c' is the departure from the mean mixing ratio of the trace gas of interest (Baldocchi 2003). The two major assumptions that need to be met for this equation to be used are that both air density fluctuations and mean vertical flow are assumed negligible for horizontal homogeneous terrain (Burba and Anderson 2010).

Turbulence exists over a large range of frequencies and therefore instrumentation needs to be able to capture instantaneous fluctuations in vertical wind speed and gas concentration at a very high speed, typically 10 times per second (10 Hz), and then averaged over 30-minutes periods to exclude events that occur at longer time scales (e.g. weather events) (Burba and Anderson 2010).

The EC method has many limitations and errors may arise due to the assumptions that are made, instrument error, physical phenomena and the characteristics of the terrain. To obtain the most accurate and reliable data, the instruments are installed facing the prevailing winds to avoid air movement through the structure and at an appropriate height above the surface to capture the upwind source area of interest. A rule of thumb is that the upwind distance to the edge of the feature of interest needs to be 100 times the height of the instrument that is measuring the trace gas and turbulence (Baldocchi et al. 2001).

The use of the EC method is restricted to flat terrain that is surrounded by a homogeneous landscape. Therefore, it cannot easily be used in mountainous regions or where there is a patterned landscape (e.g. where multiple pools or lakes are present). Furthermore, data

are removed during unfavorable weather conditions, where wind directions cause obstructed airflow, and when the air is too stable, such as at night, when turbulence is intermittent or absent (Baldocchi 2003).

EC technology has improved in the last decade and many of the errors and assumptions mentioned above can now be corrected for in post-processing, so that the total estimated uncertainty associated with eddy covariance measurements is around 7% for measurements obtained during the day and 12% for measurements obtained at night (Baldocchi 2003). Gap-filling procedures are then used to replace the removed or missing data.

2.6.2 Plant Community Scale Measurements

Ecosystem scale measurements provide a good understanding of the carbon dynamics of the peatland as a whole, but to better describe the processes that drive the variability in gas exchange, plant community scale measurements are required.

Plant community scale measurements can be obtained using chambers. These are implemented to measure gas exchange at smaller scales, and can provide great insight into the processes that drive the hydrology-carbon feedback by determining the contributions of trace gas exchange from the different vegetation types. They are also easy to use and are much less expensive than maintaining an EC tower (Clement et al. 1995).

However, there are many sources of error associated with chamber measurements. For example, when collars are inserted into the soil, the vegetation and/or rooting systems may be disturbed, which can lead to an under-or over-estimation in carbon fluxes. This issue is often addressed by waiting for a period of time after collar installation before commencing measurements (Davidson et al. 2002).

Other errors that arise with chamber measurements are due to a change in soil CO_2 concentration gradients which can cause an underestimation in fluxes of up to 15%, and pressure differences between the air inside and outside the chamber, causing either an under or over-pressurization. However, most of these errors can be addressed by installing properly sized chamber vents, and minimizing sampling times. As a general rule, the sampling period should be long enough to capture an increase in gas concentration, but short enough to not alter the temperature and pressure within the chamber (Davidson et al. 2002). There should also be a good seal in the grooves of the collars in which the chambers are placed. All samples obtained through a syringe, which is often the method used to calculate CH_4 fluxes, should be analysed within a few days (Strack et al. 2009).

Furthermore, the vegetation within the chamber may become stressed if air temperatures are too high, which may force the plants to close their stomata, under-estimating CO_2 fluxes, and may alter the plant mediated transport of CH_4 within the vascular plant species present (Davidson et al. 2002). Typically, CH_4 measurement runs are longer than CO_2 (if an IRGA is used), which increases the chance for headspace heating. Errors associated with heating inside the chamber can be addressed by using opaque chambers for CH_4 measurements, implementing a fan that circulates the air within the chamber, and attaching a cooling system to help regulate the temperature within the chamber for CO_2 measurements (Waddington et al. 2002; Günther et al. 2014).

Although there are many errors associated with the chamber technique, chamber measurements can be quite reliable and errors can be minimized if the chambers are properly designed and chamber techniques are properly implemented.

Uncertainty in chamber measurements may arise due to spatial and temporal variability. To address this issue, a large number of collars should be installed that accurately represent the vegetation coverage and there should be a higher frequency of sampling (Davidson et al. 2002).

Finally, systematic bias can occur when controlling for the quality of the chamber measurements to be used in analysis. This issue can be addressed by considering consistent criteria for the removal of any data points.

Chapter 3: Materials and Methods

3.1 Site Description

This study was conducted at a post-extracted, unrestored peatland in the Saint-Alexandre-de-Kamouraska region (47°44'0.35"N, 69°36'38.30"W), approximately 11 km west of Rivière-du-Loup, Quebec (Figure 3.1). For simplicity, the site will be referred to as SAK. The 30-year climate normals (1981-2010) for the St-Arsène weather station, near the peatland site, give a mean annual temperature of 3.5 °C, with mean January and July temperatures of -12.4 °C and 17.6 °C, respectively. The mean annual precipitation is 963.5mm, with 28% falling as snow (Environment Canada 2015).

Peat extraction started in the 1970's and was halted in 1999 when there was too much woody debris to continue extraction economically. No active restoration was done and the ditches were not blocked (Marie-Claire LeBlanc, l'Université Laval, Personal Communication). As a result, *Sphagnum* has not regenerated and the site consists mainly of bare peat fields with sparse *Eriophorum angustifolium* (Figure 3.2). *Phragmites australis* and *Typha latifolia* (commonly known as reed and cattail, respectively), both invasive species, have established themselves in the ditches. These species contain aerenchymous tissue which allows methane to bypass the distance between the water table and the surface where CH_4 would otherwise be oxidized into CO_2. Thus, these species can act as direct conduits for CH_4 release to the atmosphere.
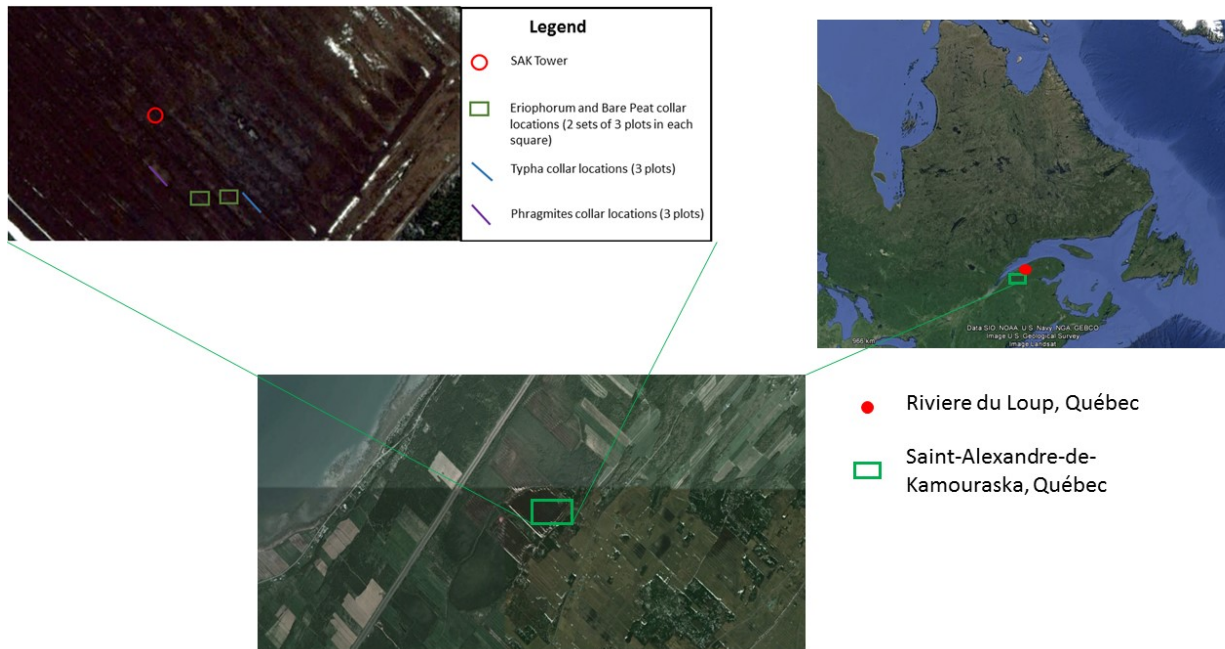
Figure 3.1. Site and collar locations.

Figure 3.2. Saint-Alexandre-de-Kamouraska (SAK): Post-extracted, unrestored peatland site.

3.2 Ecosystem Scale Flux Measurements

The EC technique (Baldocchi 2003) was used to directly and continuously measure the surface-atmosphere exchange of CO_2, H_2O, CH_4 and sensible heat fluxes at the ecosystem scale. The EC system consisted of a three-dimensional sonic anemometer (CSAT-3, Campbell Scientific, Edmonton, Canada), an open-path methane analyzer (LI-7700, LI-COR Biogeosciences, Lincoln, NE) and an open-path infrared gas analyzer (IRGA; LI-7500A, LI-COR, Lincoln, NE). All data were recorded at 10 Hz via an analyzer interface unit (LI-7550, LI-COR Biogeosciences, Lincoln, NE). The instruments were mounted ~1.5 meters above the peatland surface. CO_2 fluxes were measured year round, whereas CH_4 fluxes were only measured from March to November.

The storage flux (Fs), which is the rate of change in CO_2 concentration between the ground and the height of the tower, integrated from one 30-minute period to the next, was calculated following Morgenstern et al. (2004) as

$$F_s = h_m \, \overline{\rho_a} \, (\overline{\Delta S_c}/\Delta t) \ , \tag{3}$$

where h_m is the measurement height, $\overline{\rho_a}$ is the mean molar density of dry air, and $\overline{\Delta S_c}$ is the difference between S_c of the previous and following half hours, respresenting the mean molar mixing ratio of CO_2. NEE was then determined by

$$NEE = F_c + F_s \ , \tag{4}$$

where F_c represents the turbulent eddy flux of CO_2 calculated by the EC tower.

3.2.1 Data Handling

Data were processed using EddyPro software (v.5.2.0, LI-COR Biogeosciences, Lincoln, NE). Quality controls followed FluxNet protocols (e.g. Bergeron and Strachan 2012). A two-axis rotation and the WPL correction were applied. Data were rejected if they were greater than three standard deviations from the mean. Nighttime data were filtered for low turbulence using a friction velocity threshold (u*) value of 0.1 m/s, determined following the procedure in Mkhabela et al. (2009). Data were also rejected when the IRGA's path became dirty or was obscured by precipitation. Quality control, power loss, and precipitation events resulted in 49% and 53.5% of the data being rejected for the CO_2 and CH_4 fluxes respectively. Removing between 40% and 60% of the data is typical of EC operations (e.g. Humphreys et al. 2006; Strilesky and Humphreys 2012; Rinne et al. 2007).

3.2.2 Gap Filling

Small gaps (fewer than four half-hour periods) in the continuous 30-minute CO_2 flux data were filled by linear interpolation. Larger gaps were filled following Bonneville et al. (2008). Briefly, turbulent nighttime (PAR < 20 µmol m^{-2}s^{-1}) NEE was assumed to represent the ecosystem respiration (ER). A relationship was developed between ER and peat temperature at 20 cm depth to fill missing nighttime periods. Ecosystem level light response curves were constructed and used to model GEP. The resulting GEP was used with ER to fill the NEE missing data. CH_4 data were not gap filled as no standard technique is available.

3.2.3 Supporting Measurements

Air temperature and relative humidity (HMP3, Vaisala, Helsinki, Finland), wind speed and direction (RM Young, Traverse City, MI), four component radiation budget (CNR1, Kipp and Zonen, Delft, Netherlands), incoming and reflected PAR (LI-190Sa, LI-COR, Lincoln, NE),

precipitation using a tipping bucket rain gauge (TE525M, Texas Electronics, Dallas, TX), soil

temperature at 5, 10, 15, 20, 40 and 60 cm depths using Omega thermocouple wires, and soil

moisture (Campbell CS616, Campbell Scientific, Edmonton, AB) were averaged over 30

minutes and stored on a datalogger (CR5000, Campbell Scientific, Edmonton, AB).

3.3 Plant Community Scale Flux Measurements

CO_2 and CH_4 exchange, at the plant community level, were measured using static

chambers during the warm seasons of two years (May to September, 2014 and 2015). Six 60x60

cm collars were placed to contain *Eriophorum* and six to contain bare peat. Locations were

chosen to capture the elevation gradient between two sets of ditches, thereby representing areas

of differing water table depths. Nine circular collars (28 cm diameter) were placed to measure

CH_4 fluxes from the ditches; three contained *Typha*, three contained *Phragmites*, and three

contained bare peat (Figure 2.1).

The chamber technique for plant community level CO_2 measurements is described in

detail in Pelletier et al. (2011). Briefly, for each measurement, a clear acrylic/plastic chamber

was placed over the collar, and water added to the grooves of the collar to act as a tight seal. CO_2

was sampled using a portable infrared gas analyzer (EGM4, PP Systems, Amesbury,

Massachusetts, USA). The chambers cycled cooled water through a copper coil to help regulate

the temperature within the chamber during measurements, and a fan within the chamber

circulated air. Four measurement runs were performed at each collar, representing four different

light levels. Each run lasted 105 seconds, with measurements of CO_2 concentrations and

Photosynthetically active radiation (PAR) recorded every fifteen seconds. The first run used a

clear chamber which transmits roughly 95% of the incoming PAR. The next two runs were done

using mesh shrouds, allowing the transmission of approximately two thirds and one third

respectively, of the incoming maximum PAR. The final run was done using an opaque tarp,

blocking the incoming light, with measurements representing dark respiration. Between each run,

the chamber was removed from the collar to allow CO_2 concentrations to reach ambient values.

Data from these four light conditions were used to create light response curves for *Eriophorum*

using

$$NEE_c = \frac{GP_{max} * \alpha * PAR}{\alpha * PAR + GP_{max}} + R \quad , \tag{5}$$

where the chamber-based Net CO_2 exchange (NEE_c) is modelled as a rectangular hyperbolic

relationship of PAR. The parameters α, GP_{max} and R represent the initial slope of the curve, the

maximum gross primary productivity and the dark respiration values, respectively. Light

response curves were created for *Eriophorum* for the two years of study using the "solver" tool

in Excel 2013 (Microsoft Office). All flux data follow the micrometeorological convention of

positive values indicating a release of CO_2 to the atmosphere and negative values indicating an

uptake of CO_2.

Methane gas samples from *Eriophorum* and bare peat were retrieved directly from an

opaque square chamber using a syringe (five samples taken over a twenty-minute time period at

0, 5, 10, 15, and 20 minutes) via a sampling tube and stopper. This method is similar to that

described by Whalen (2005) and Pelletier et al. (2007). *Typha* and *Phragmites*, which can grow

as tall as 2 meters (Kim et al. 1999), required the use of a cylindrical chamber (height of 125.5

cm and diameter of 27 cm), that was placed on the circular collars. Gas samples were collected

similarly to those using the square chamber. All gas samples were returned to the lab within 2-3

days and run through a gas chromatograph (Mini-2, SRI Instruments, California, USA) with 5 ml injections of each sample. N_2 was used as a carrier gas and five standards of CH_4 (5 parts per million) were used before each sample run.

At the time of sampling, water table positions were determined manually. A tube covered with a tape measure was placed into a PVC pipe, previously inserted in the soil. We measured the depth at which the water bubbles when air was forced into the tube. Air temperature was recorded using a thermocouple temperature profile and soil temperature measurements were obtained at depths of 2, 5, 10, 15 and 20 cm by inserting a temperature probe into the soil.

3.3.1 Quality Control of Chamber Measurements

CH_4 and CO_2 fluxes were calculated as the change in concentration over time. The best fit lines were then determined. Any samples resulting in a fit with an R^2 less than 0.8 were rejected as suggested by Pelletier et al. (2011). NEE_c measurements (at maximum light, using only the clear chamber), where the PAR was below 1000 $\mu mol\ m^{-2}s^{-1}$, were also removed, as suggested by Strack et al. (2016), since low PAR levels result in large fluctuations of CO_2 concentrations within the chamber.

Quality control measures resulted in the removal of 23% and 20% of the CO_2 and CH_4 chamber-based measurements, respectively in 2014. However, due to unfavourable climatic conditions in 2015 (e.g. cloudy days and increased rainfall events), 36% and 27% of the CO_2 and CH_4 chamber-based measurements were removed.

3.4 Vegetation Characteristics

A vegetation survey was conducted in August 2015 to create a spatial representation of the vegetation lying within the eddy covariance tower footprint. Using a handheld personal

navigation system (eTrexVista, Garmin Ltd., Olathe, KS, USA), patches of *Typha* and

Phragmites were located to within a 3-meter accuracy, and tufts of *Eriophorum* were tagged,

using a volumetric water content probe with embedded GPS software (H2, HydroSense II,

Campbell Scientific, Edmonton, AB, Canada), to within an accuracy of 5 meters.

3.5 Peat and Pore Water Chemistry

A peat core was obtained in August of 2015 to determine soil pH and bulk density with

depth. A PVC pipe was placed in the peat to obtain soil samples for the first 50 cm depth. To

obtain deeper peat samples (0.5-1.95 m depth), a Russian peat corer was used. Bulk density was

measured in depth increments of 10 cm and was calculated using the "Intact Core Method" as

described by McKenzie et al. (2002). Soil samples at the same depth increments were mixed

with deionized water using a 1:4 soil to water ratio, and the soil pH was measured using a pH

meter (PC 300, Oakton Instruments, Vernon Hills, IL USA) in the lab.

Pore water samples were obtained once in September 2015 when the water was below the

surface in both the peat fields and the ditches, and multiple times from April to mid June 2016

when the ditches were bank-full. The samples were collected from the same PVC tubes used for

the manual water table measurements, were filtered in the lab with 0.4 μm filter paper, and were

analysed for various nutrients such as calcium, magnesium, potassium, nitrate, ammonium, and

phosphate (Ca, Mg, K, NO_3, NH_4, PO_4). NO_3, NH_4 and PO_4 concentrations were all measured

colorimetrically at 520nm, 660nm, and 880nm respectively, using flow injection instruments.

The QuickChem Methods 10-107-06-2-C, 10-107-04-1 C, and 10-115-01-1-A were used to

calculate concentrations of ammonium, nitrate and phosphate respectively (Lachat Instruments,

Milwaukee, WI, USA). Ca, Mg and K were determined using flame atomic absorption

spectrophotometry (Perkin-Elmer, model 2380, Waltham, MA, USA). Finally, the pH of the pore

water samples was calculated using both pH paper strips, and a portable pH meter (PC 300,

Oakton Instruments, Vernon Hills, IL USA).

Chapter 4: Results

4.1 Environmental Variables

Air temperatures varied between the two study growing seasons (May-October) of 2014 and 2015. In 2014, air temperatures were within 1.0 °C of the corresponding normal for the months of May, August and September. June, July and October monthly air temperatures were 1.2, 1.6 and 1.4 °C warmer than normal, respectively. In 2015, the monthly air temperatures were +1.7, -1.2, -1.0, +1.6, +3.5, and -0.9 °C different from the corresponding normal for May, June, July, August, September and October respectively (Environment Canada 2015).

A sharp measured increase in air temperature in April corresponded with the start of the snowmelt period and was followed by a sharp decrease in shortwave albedo. The albedo increased slightly on day 107 in 2015 due to an overnight snowfall. The measured trend in albedo and snowfall is typical for wetlands (Lafleur 2008). However, in contrast to peatlands with active growth, the albedo did not increase over the course of the warm season because of the dominant cover of dark bare peat. The two years differed in the amount of snow that fell throughout the preceding winter. The snow pack reached a height of 70 cm at the end of March in 2014, whereas the snow pack only reached a height of approximately 30 cm in 2015 (Figure 4.1).

A thermocouple profile was not installed in the ditch until July 2014. However, measurements pre-snow melt in 2015 indicate that peat temperatures at 10 cm depth in the ditch were higher than peat temperatures at 10 cm depth in the bare peat fields, with the ditch temperatures staying at or around 0 °C and the peat temperatures below freezing. Furthermore,

peat temperatures were greater than 0 °C on day 144 at 40 cm depth in 2014, whereas the peat

only started to thaw on day 156 at 40 cm depth in 2015 (Figure 4.2).
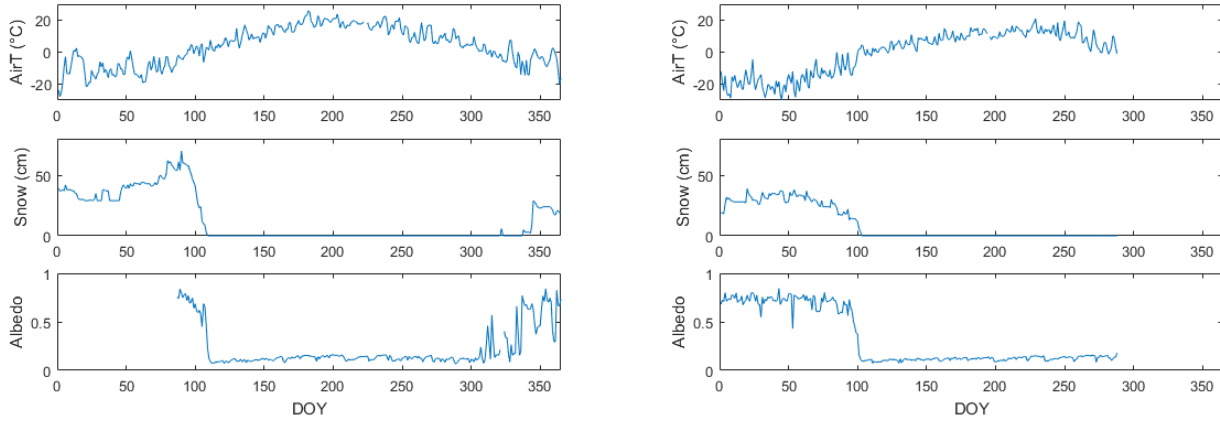
Figure 4.1. Air temperature (top), snow depth (middle), and albedo (bottom) for 2014 (left)

and 2015 (right).
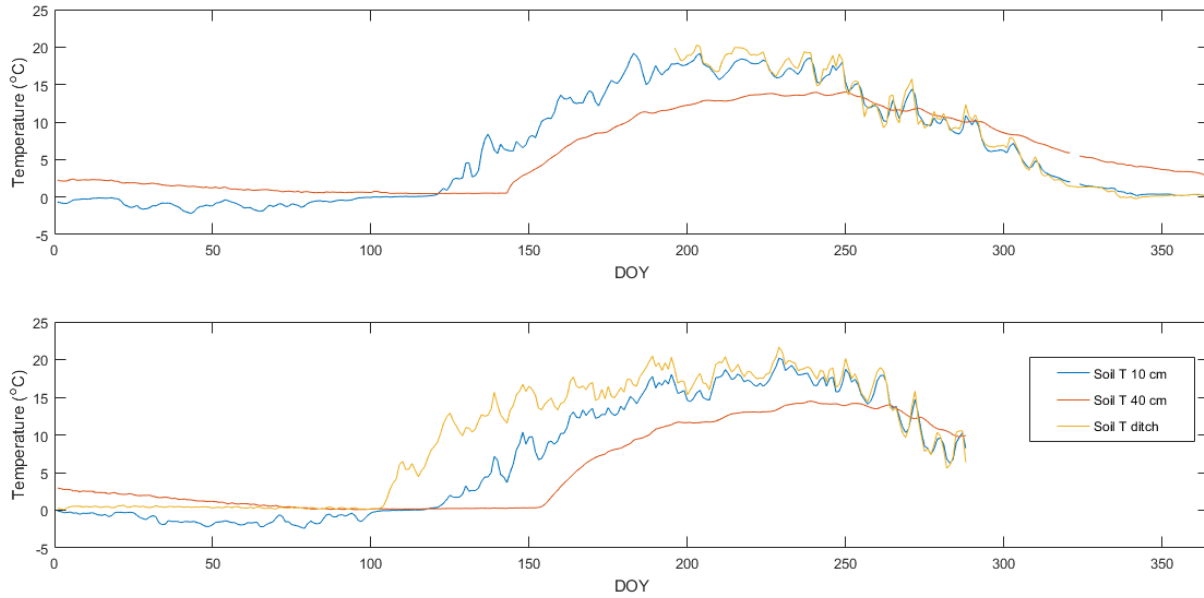
Figure 4.2. Soil temperatures at 10 cm depth in the peat field (blue) and in the ditch

(yellow) and soil temperatures at 40 cm depth in the peat field (red) for 2014 (top) and 2015

(bottom).

Precipitation measurements varied even more than air temperature both seasonally and inter-annually between the two growing seasons. In 2014, precipitation was more sporadic and at least 30 mm lower than the corresponding monthly normal, except for July and October, which were 3 mm and 65 mm higher than normal respectively. In 2015, monthly precipitation was +15 mm, +5 mm, +30 mm, -31 mm, -44 mm, and -35 mm different than normal for May, June, July, August, September and October, respectively.

Increases in soil moisture corresponded to rainfall events for the two study periods (Figure 4.3) and the resulting soil moisture values fell within the range of those reported at other extracted sites (Strack and Price 2009). Soil water content was slightly higher in 2015 than in 2014. In 2015, the maximum soil water content of 0.35 m^3/m^3 was reached around day 130 (Mid-May), while in 2014, maximum soil water content was 0.3 m^3/m^3 and was reached much later around day 275 (Mid-October).
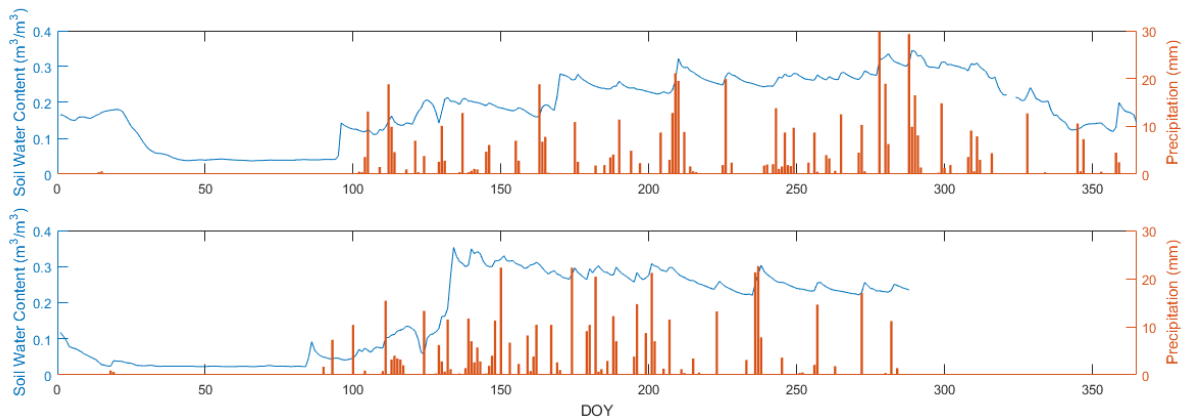
Figure 4.3. Soil water content and precipitation in 2014 (top) and 2015 (bottom).

Manual water table (WT) measurements showed a similar trend to soil water content (Figure 4.4). In 2014, the water table fell as low as 72 cm below the surface in September, and rose to 20 cm below the surface in November. In 2015, the WT followed a similar trend, but

remained closer to the surface throughout the growing season. Mid-season periods in 2015 where the WT rose above 30 cm below the surface correspond to precipitation events.
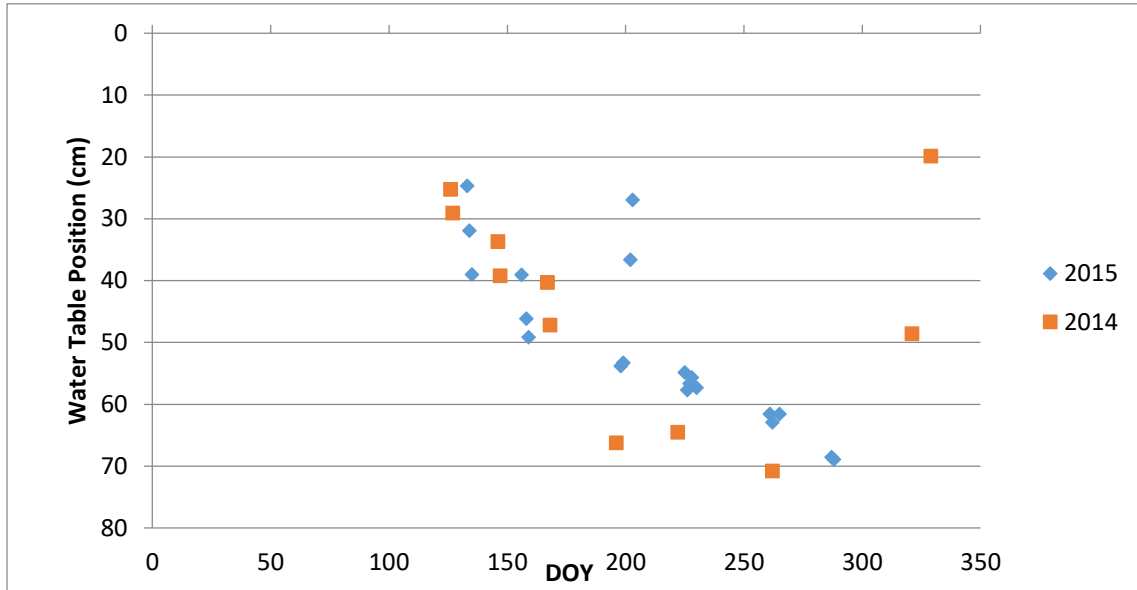
Figure 4.4. WT position from manual measurements in 2014 and 2015 (0 cm represents the surface).

Manual WT measurements were obtained across an elevation gradient representing a transect from a ditch to the peak of the adjacent berm. Surface maps were created to portray transects of WT positions through time for different elevation points within the peatland (Figure 4.5).
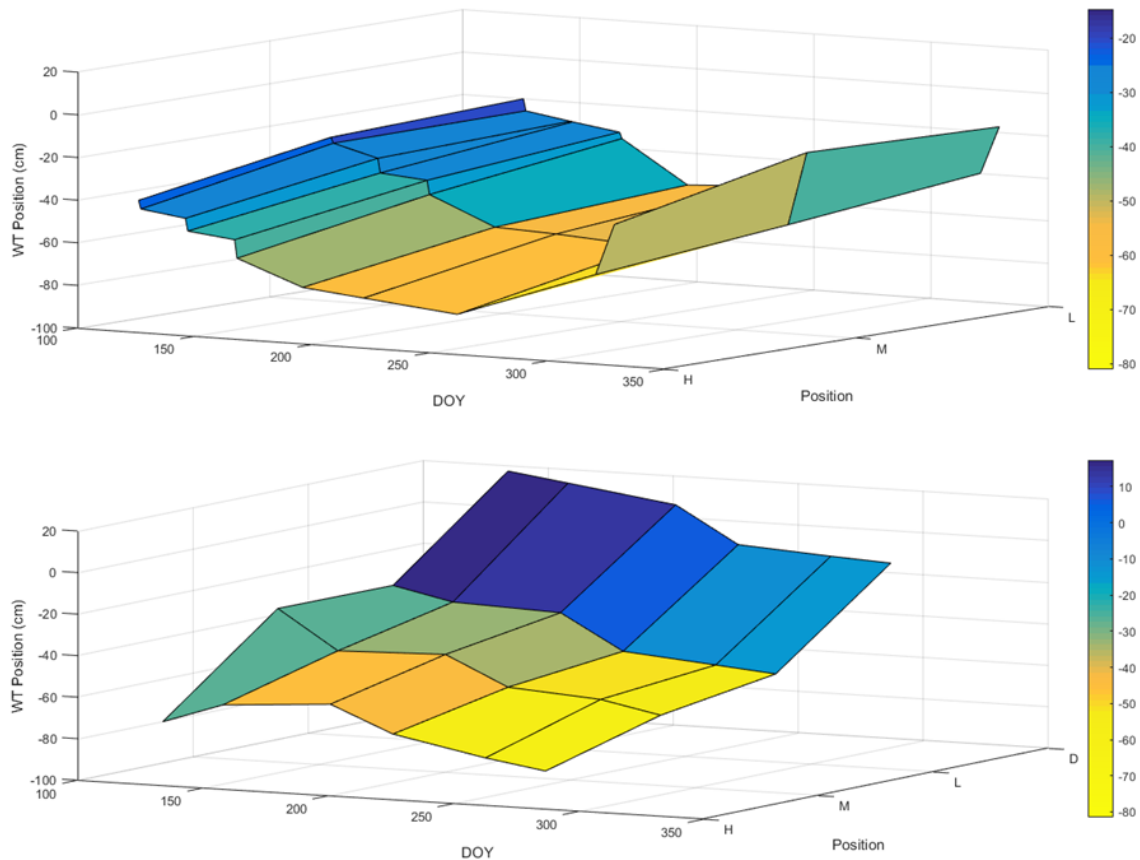
Figure 4.5. WT position for the 2014 (top) and 2015 (bottom) warm seasons. No ditch (D) measurements were available in 2014; L, M and H represent the low, medium and high positions of the transect across the berm.

The low (L) and medium (M) WT positions were similar throughout both years, whereas the high (H) WT positions were different (the water table was at a much greater depth). Although the time series trends were similar for all three positions, the WT changed slightly more rapidly in the M and H positions than the L position. In 2015, water remained above the surface of the ditch until the end of July; it then remained at a shallow depth throughout the rest of the measurement period.

4.2 Peat and Pore Water Chemistry

Bulk density ranged from 0.12 to 0.25 g/cm^3, values similar to those that have been reported for other northern peatlands (Waddington and Roulet 2000; Chambers et al. 2011). Peat pH ranged from 5.5 in the deepest layer to 3.5 closer to the surface, a pH profile typical of disturbed peatlands as well (Wind-Mulder et al. 1996).

The pH of the pore water was measured to be around 6 using both the pH paper strips and the portable pH meter. High ammonium (NH_4), calcium (Ca) and potassium (K) concentrations were measured from the pore water samples collected in September 2015, ranging between 2.6 to 6.3 mg N/L, 3.26 to 20.9 mg/L and 0.7 to 3.6 mg/L, respectively. Such ranges of pH and nutrient concentration values are usually associated with a rich fen (Mitsch and Gosselink 1993). Samples collected from the bank-full ditches between the end of April and mid-June 2016 show NH_4, Ca and K concentrations ranging between 0.1 to 0.7 mg N/L, 0.9 to 3.5 and 0.41 to 0.45 mg/L, respectively; values which are more associated with a bog (Mitsch & Gosselink 1993). PO_4, NO_3 and Mg concentrations did not change significantly between sampling times, ranging between 0.01 to 0.02 mg/L, 0.1 to 0.7 mg/L and 0.4 to 3.4 mg/L, respectively. Similar values of nutrient concentrations in disturbed peatlands have been reported in the literature (Wind-Mulder et al. 1996; Wind-Mulder and Vitt 2000).

4.3 CO$_2$ Measurements

4.3.1 Ecosystem Scale

Gap-filled ecosystem scale CO_2 carbon fluxes (g C m^{-2}d^{-1}) were derived from tower-based measurements using the eddy covariance approach. The reported fluxes follow the

atmospheric convention that positive values represent an atmospheric gain (release of CO_2 from the surface) and negative values represent an atmospheric loss (uptake by the peat surface).

Both growing seasons were overall sources of CO_2 to the atmosphere, where 2015 was a much higher source of CO_2 to the atmosphere than 2014, with annual cumulative NEE values of 153 g C m^{-2} in 2014 and 241 g C m^{-2} in 2015. Daily average NEE for 2014 showed maximum CO_2 release of approximately 1.5 g C m^{-2}d^{-1} at the end of June and the end of August, and maximum uptake around -1.0 g C m^{-2}d^{-1} in mid-May. For 2015, daily average NEE showed a maximum CO_2 release of approximately 1.7 g C m^{-2}d^{-1} at the beginning of August, and a maximum uptake around -0.3 g C m^{-2}d^{-1} in mid-May (Figure 4.6). Similar trends were found for daily average ER and daily average gross ecosystem productivity (GEP) between the two years as well, where greater values of GEP correspond to a higher uptake of CO_2, as suggested by Strack et al. (2006). 2015 had a greater annual cumulative ER (353 g C m^{-2}) than 2014 (330 g C m^{-2}), whereas 2014 had a greater annual cumulative GEP (176 g C m^{-2}) than 2015 (147 g C m^{-2}). Maximum daily average ER in 2014 was approximately 2.6 g C m^{-2}d^{-1} at the end of June, and maximum GEP in 2014 was approximately 2.0 g C m^{-2}d^{-1} in mid-May (Figure 4.6). Maximum daily average ER in 2015 was approximately 2.6 g C m^{-2}d^{-1} in mid-August, and maximum daily average GEP was approximately 1.6 g C m^{-2}d^{-1} in mid-July (Figure 4.6).
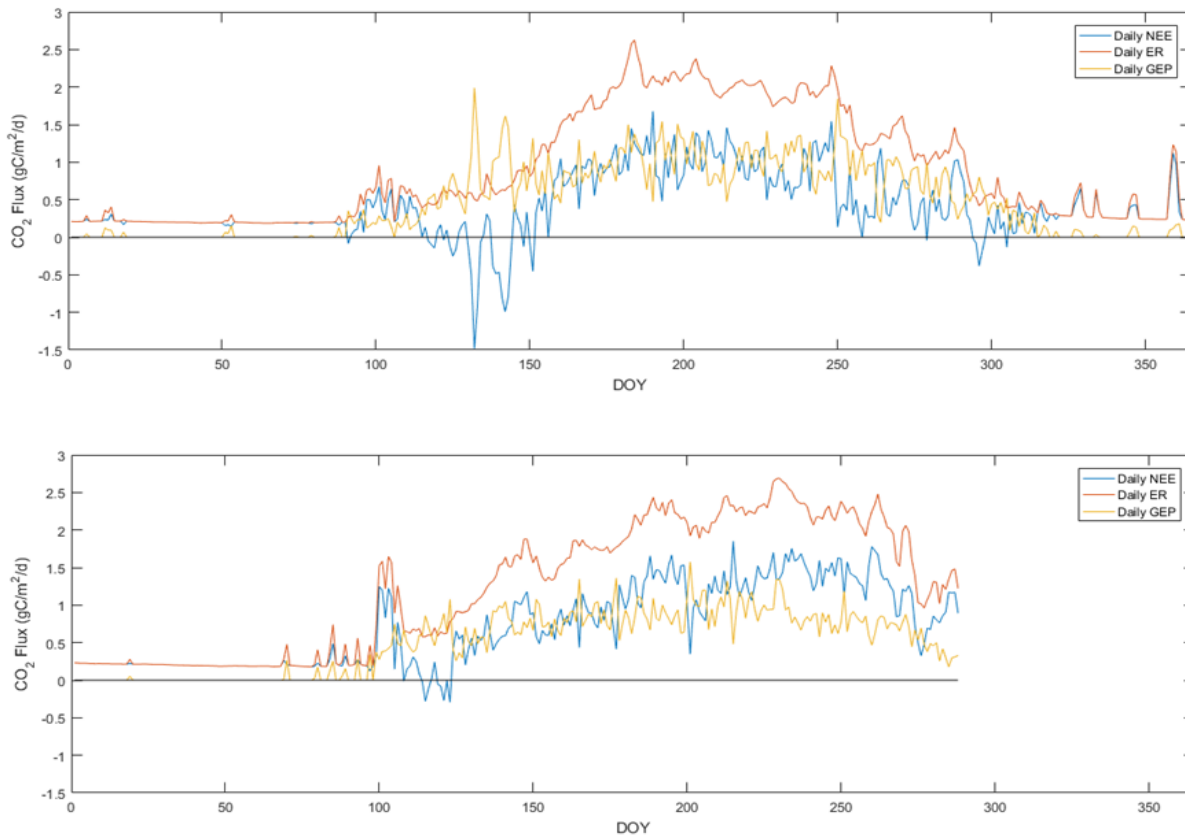
Figure 4.6. Daily net ecosystem exchange (NEE), ecosystem respiration (ER) and gross ecosystem productivity (GEP) for 2014 (top) and 2015 (bottom).

Diurnal patterns of CO_2 are evident for both growing seasons. In 2014, the diurnal trends in 30-minute CO_2 fluxes averaged monthly showed the highest daytime uptake of CO_2 (through photosynthesis) in May, September and October and highest nighttime respiration in June, July and August. In 2015, the diurnal trends in 30-minute CO_2 fluxes averaged monthly showed the highest daytime uptake in May, June and October and highest nighttime respiration in July, August and September (Figure 4.7). The 30-minute diurnal CO_2 fluxes were generally higher in 2015 than 2014. These patterns in NEE indicate a strong diurnal relationship between CO_2

exchange and peat temperature. We analysed the relationship between monthly averaged half-hour NEE measurements and monthly averaged half-hour peat temperatures at 10 cm depth. There was a discernable hysteresis effect created by the difference between the daytime NEE-peat temperature and the nighttime NEE- peat temperature relationships (Figure 4.8).
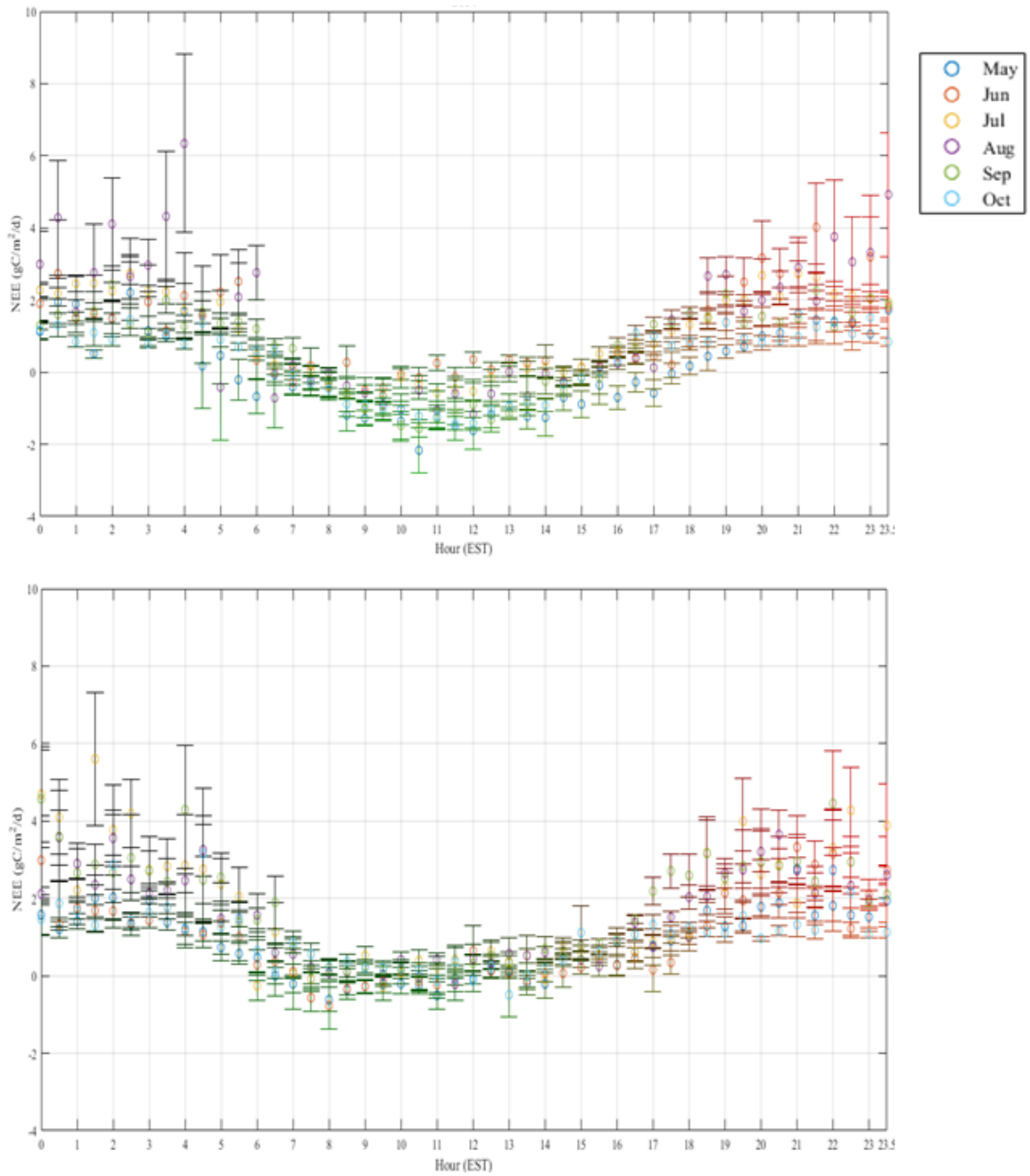
Figure 4.7. 30-minute CO₂ flux measurements averaged monthly for 2014 (top) and 2015 (bottom). Error bars represent one standard error from the mean.
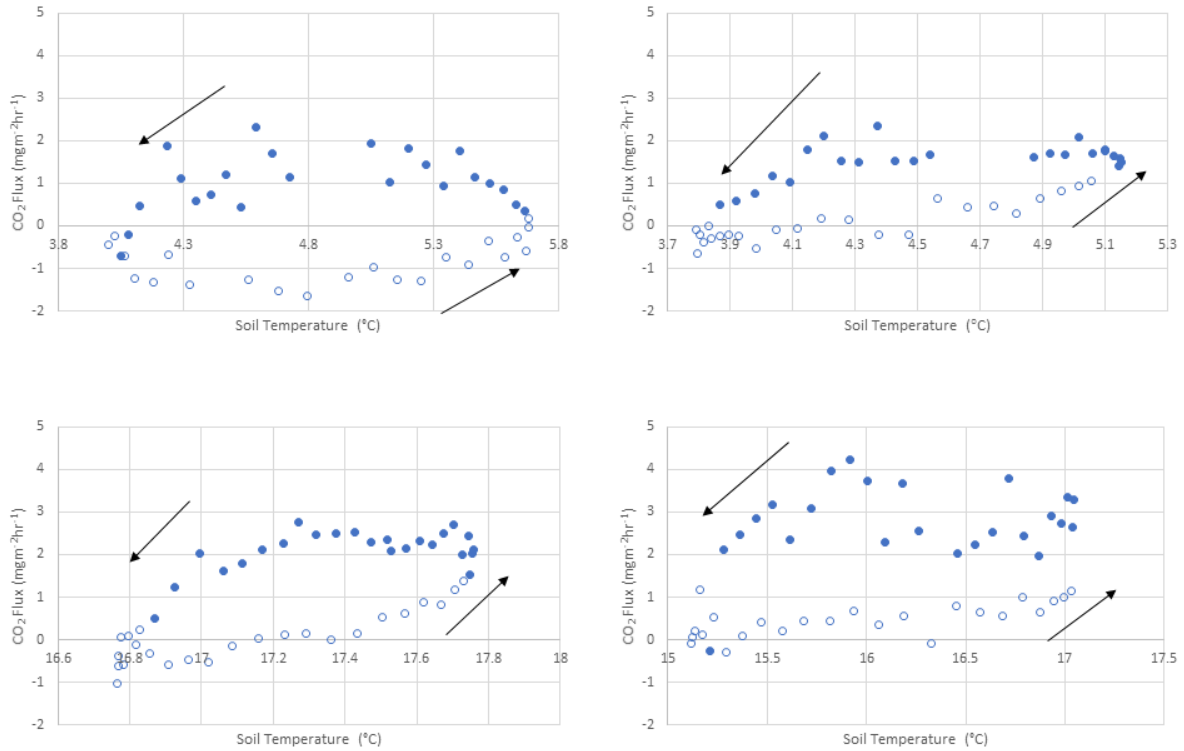
Figure 4.8. 30-minute net ecosystem exchange (NEE) averaged monthly against 30-minute soil temperature at 10 cm depth for May 2014 (top left), July 2014 (bottom left), May 2015 (top right) and July 2015 (bottom right). The open circles represent daytime fluxes (7AM – 7PM EST) and the closed circles represent nighttime fluxes. The other months are not shown here.

4.3.2 Plant Community Scale

Measurements of CO_2 exchange at the plant community scale show similar trends to those found at the ecosystem level. Average values of Net CO_2 exchange (NEE_c) (at maximum light level, using only the clear transparent chamber), where PAR was greater than 1000 µmol m^{-2}s^{-1}, show that *Eriophorum* took up CO_2 in both years with maximum uptake of -40 mg C m^{-2}hr^{-1} in both May 2014 and July 2015, whereas the bare peat released CO_2 in both years, with

maximum release of 38 mg C m^{-2}hr^{-1} in June 2014 and 21 mg C m^{-2}hr^{-1} in August 2015 (Figure 4.9). Average values of dark respiration (R) (with no light transmittance, using an opaque tarp) show a maximum release of CO_2 of 61 mg C m^{-2}hr^{-1} in August 2014 and 89 mg C m^{-2}hr^{-1} in August 2015 for *Eriophorum,* and a maximum of 38 mg C m^{-2}hr^{-1} in August 2014 and 31 mg C m^{-2}hr^{-1} in September 2015 for the bare peat (Figure 4.9). Chamber-based Gross CO_2 productivity (GEP$_c$) was calculated for *Eriophorum* as R – NEE$_c$, where maximum uptake was 74 mg C m^{-2}hr^{-1} in May 2014 and 91 mg C m^{-2}hr^{-1} in August 2015 (Figure 4.9). Since 92% of the surface area of the peatland site is dominated by bare peat cover, whereas *Eriophorum* only covers ~3% of the surface area of the peatland, dark respiration greatly exceeded the amount of CO_2 taken in by the *Eriophorum.* Thus, the plant community scale measurements also indicated that the site was an overall source of CO_2 to the atmosphere in both years.
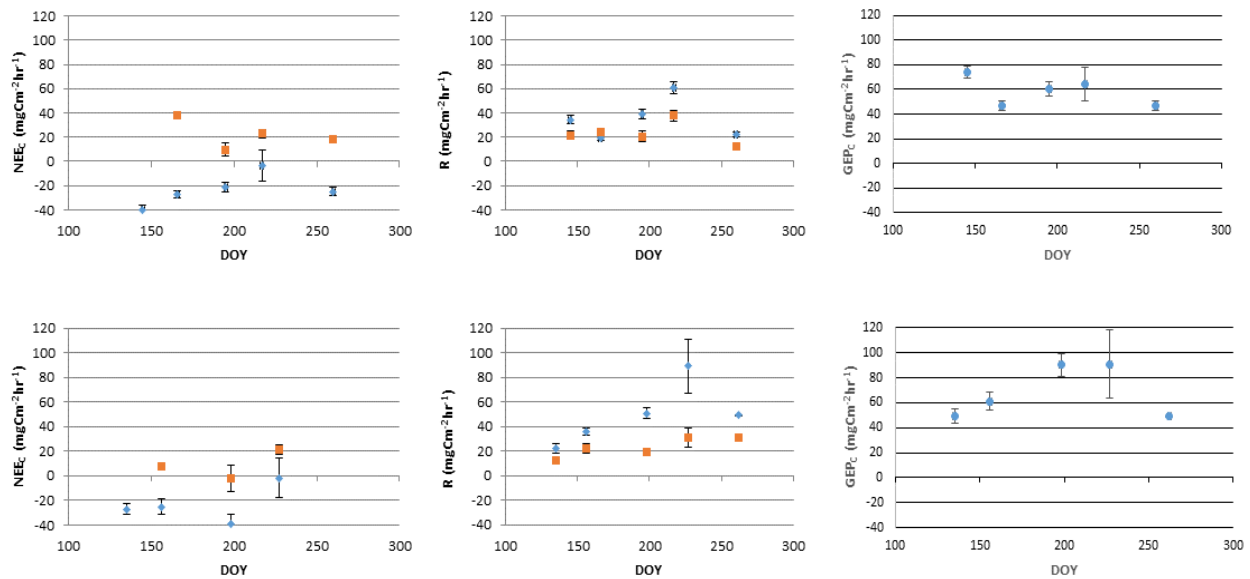
Figure 4.9. Plant community scale CO$_2$ exchange: NEE$_c$ (left), R (middle) and GEP$_c$ (right) for the *Eriophorum* (in blue) and bare peat collars (in orange), +/- standard error, in 2014 (top) and 2015 (bottom).

The 2014 light response curve for *Eriophorum* follows the general rectangular hyperbolic trend, with fit values of -0.11, -75.45 and 32.71 for the initial slope of the curve (α), the maximum gross primary productivity (GP_{max}) and the dark respiration (R), respectively. The light response curve in 2015 had a more negative GP_{max} and a less positive R in comparison to that of 2014, with fit parameters of -0.03, -137.29 and 29.04 for α, GP_{max} and R, respectively (Figure 4.10). The curve fit parameters from both years fall within the range of reported values for *Eriophorum* in other studies (Marinier et al. 2004; Frolking et al. 1998).
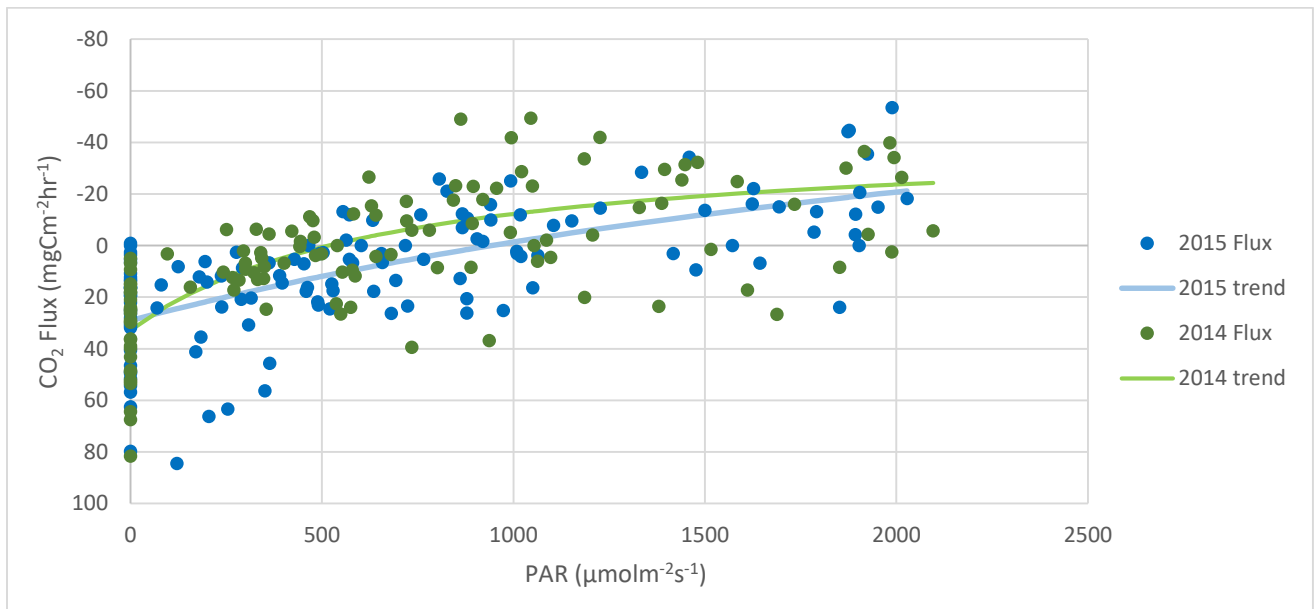
Figure 4.10. Light response curves for *Eriophorum* in 2014 (green) and 2015 (blue).

4.4 Environmental Controls on CO_2 Exchange

4.4.1 Ecosystem Scale

The dominant component of NEE is ER, which is a function of temperature. Therefore, it was not surprising that we found a strong linear relationship between NEE and temperature in

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both years, since NEE is also influenced by ER. Higher temperatures coincided with a higher net release of CO_2, especially for temperatures greater than 0 °C. In 2014, air temperature and soil temperature explained 41.5% and 44% of the variance in NEE, respectively and in 2015, air temperature and soil temperature explained 53.5% and 76% of the variance in NEE, respectively. As expected, there was a stronger linear relationship between ER and temperature in both years. In 2014, air temperature and soil temperature explained 77% and 91% of the variance in ER, respectively and in 2015, air and soil temperature explained 65% and 92% of the variance in ER, respectively (Table 4.1).

Table 4.1. Coefficient of determination (R^2) for linear regressions between ecosystem scale CO_2 exchange (NEE, ER and GEP) and environmental variables in 2014 and 2015. SWC is soil water content (%) at 30 cm depth, Precip. is rainfall (mm), WD is wind direction, Air T is air temperature measured at the flux tower, Soil T is soil temperature at 10 cm depth. Relationships are significant at * P<0.05, ** P<0.01 and *** P<0.001; all others are not significant (N.S.) at P>0.05.

Env. Var.	NEE (2014)	ER (2014)	GEP (2014)	NEE (2015)	ER (2015)	GEP (2015)
SWC	0.12***	0.35***	0.33***	0.55***	0.75***	0.71***
Precip.	0.09***	N.S.	N.S.	0.06***	N.S.	N.S.
WD_All	0.08***	N.S.	N.S.	0.05***	N.S.	N.S.
WD_May	0.20*	N.S.	0.22**	0.43***	0.35***	0.23**
Air T	0.42***	0.77***	0.34***	0.53***	0.65***	0.16***
Soil T	0.44***	0.91***	0.57***	0.76***	0.92***	0.24***

We did not expect there to be any significant linear relationship with wind direction, but wind direction explained 22% and 23% of the variance in GEP and 21% and 43% of the variance in NEE in 2014 and 2015 respectively, especially when only the month of May was considered. When any or all of the other months were included, wind direction did not have any effect on GEP nor on NEE.

There was a stronger linear relationship between NEE and soil water content and between ER and soil water content in 2015 than in 2014, where increased soil moisture led to higher respiration. Soil water content explained 12% and 55% of the variance in NEE and 39% and 75% of the variance in ER for 2014 and 2015, respectively. If an exponential fit is used, 50% and 81% of the variability in ER are explained for 2014 and 2015, respectively.

The linear relationship between NEE and precipitation was much weaker in 2015 than in 2014, where 9% and 6% of the variability are explained for 2014 and 2015, respectively. Soil water content explained more of the variance in GEP in 2015 (71%) than in 2014 (33%), whereas air and soil temperature explained more of the variance in GEP in 2014 (34% and 57%, respectively) than in 2015 (23% and 16%, respectively).

4.4.2 Plant Community Scale

Soil moisture seemed to play a large role in the CO_2 dynamics at the plant community scale as well, where we found WT position to be the main driver of bare peat respiration, explaining 77% of the variance in 2014 and 78% of the variance in 2015 (at the 90% confidence level). The relationships between peat respiration and air and peat temperatures were not significant. Relationships between both *Eriophorum* CO_2 exchange and dark respiration with soil moisture were not significant. The main driver of chamber-based CO_2 exchange for *Eriophorum* seemed to be peat temperature in 2014 ($R^2 = 0.79$) and air temperature in 2015 ($R^2 = 0.75$), and

the main driver of dark respiration for *Eriophorum* was air temperature in 2015 ($R^2 = 0.92$). There were no significant relationships between dark respiration and any of the environmental variables for *Eriophorum* in 2014.

4.4.3 Vegetation Survey and Footprint Analysis

Most of the site has very sparse vegetation. However, ditches were not blocked following extraction and invasive species have colonized large portions of the ditches. With this vegetative heterogeneity, wind direction may have played a role in the measured variability in CO_2 exchange at the ecosystem scale. We hypothesized that periods when the tower measurements indicated increased uptake of CO_2 corresponded to periods when the wind was coming from the direction containing a greater proportion of ditch in the upwind footprint. To test this, we compared the monthly average wind direction with the monthly average NEE exchange in May of both years (Figure 4.11). This was then coupled with a spatial representation of the vegetation lying within the tower footprint. Larger CO_2 uptake was measured in May of both years from the directions containing a greater abundance of *Eriophorum* in the ditches (Figures 4.11 and 4.12). As mentioned previously, *Eriophorum* take up roughly 3% of the peatland's surface cover.

Presently, the invasive species established at the study site cover a little under 5% of the peatland surface area and are mostly confined to the ditches. Thus, the current overall vegetation cover is about 8%. The remaining 92% of the peatland is dark bare peat. However, using Google Earth images from 2009 and 2011 and ArcGIS images from 2014 and 2015, it was estimated that the *Phragmites* had started to spread into the surrounding peat fields at a rate of approximately 244 m^2/yr.
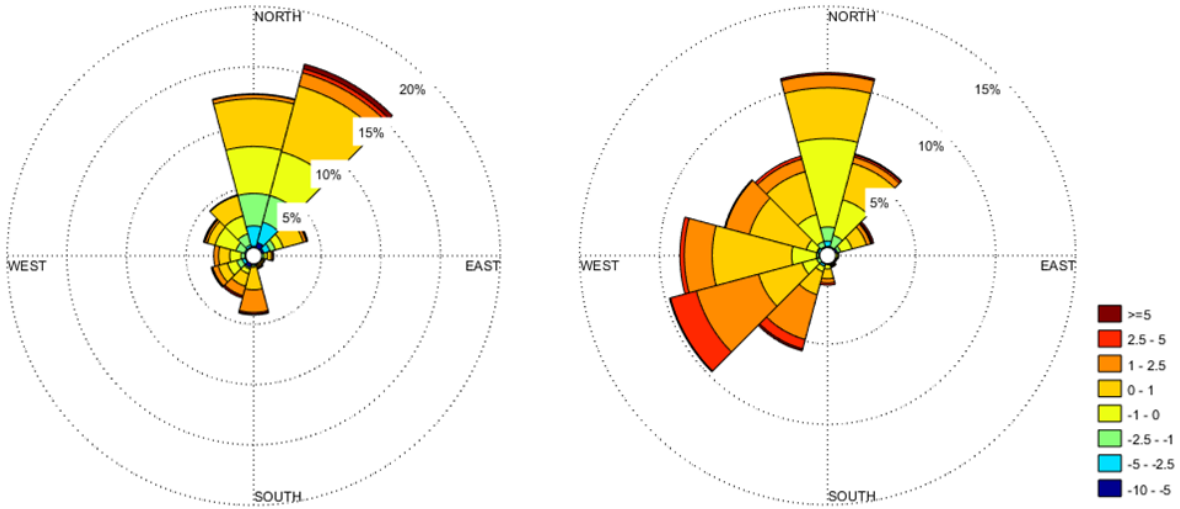
Figure 4.11. Wind rose for May 2014 (left) and May 2015 (right). The length of the wedge depicts the frequency of observations from the given wind direction, while the color represents the magnitude of the CO_2 flux. Negative values indicate an uptake of CO_2.
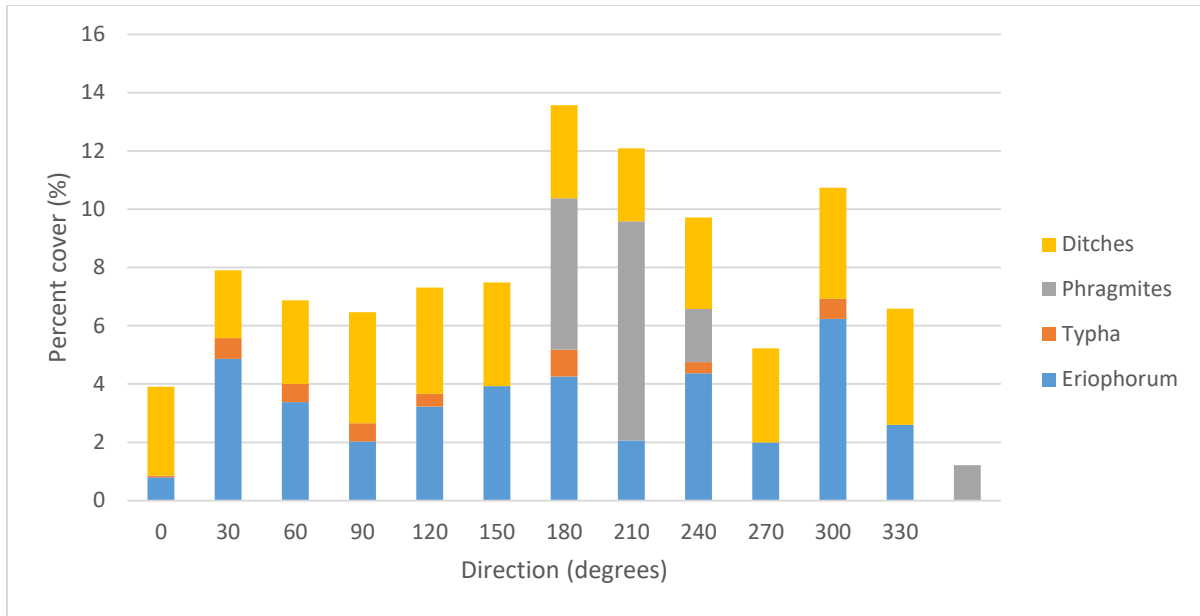
Figure 4.12. Surface percent cover by major plant species and wind direction based on the vegetation survey taken in August 2015.

A footprint analysis of the tower flux data indicated that during May of both years, the fluxes originated from distances extending to between 75 and 95 meters from the tower (based on a 70% probability) and 130 and 180 meters from the tower (90% probability). The vegetation survey used a 150-meter radius from the tower and is thus representative of the distribution of vegetation at the site from a tower-flux perspective.

4.5 CH$_4$ Measurements

4.5.1 Ecosystem Scale

After analyzing the 2014 tower-flux CH$_4$ dataset, the quality of the data was deemed questionable; inconsistencies could not be explained nor corrected. The 2015 dataset showed

none of the problems of the 2014 data. Therefore, to err on the side of caution, only the data for 2015 will be presented here.

Daily average methane fluxes indicate that 2015 was a slight overall source of CH_4 to the atmosphere. Assuming no flux during the cold periods, the site released approximately 0.8 g C-CH_4 m^2 annually. This falls within the range measured at other unrestored peatlands (Strack and Waddington 2012; Waddington and Day 2007). Most of the CH_4 fluxes in 2015 were not statistically different from zero. However, there were short periods of time, in March, June, August and September, where significant spikes in methane fluxes were discernable (Figure 4.13).
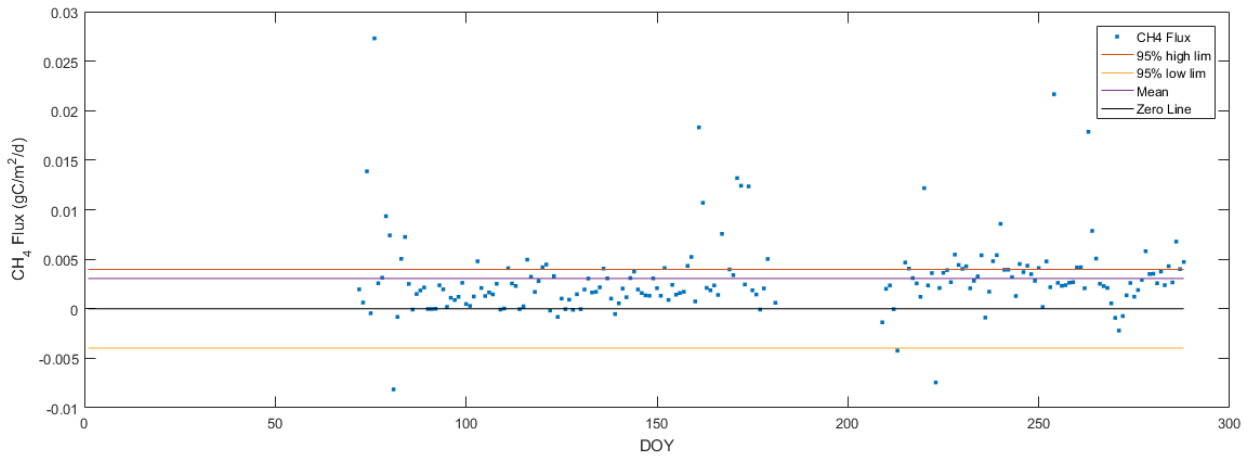
Figure 4.13. Average daily CH$_4$ fluxes for 2015 (blue dots). The mean annual flux is depicted by the purple line, zero flux is depicted by the black line, and the 95% confidence interval is depicted by the yellow and red lines.

4.5.2 Plant Community Scale

Chamber-based measurements of methane were available in both years of the study and results indicate that 2014 was a higher source than was 2015, with most of the emissions being

released by the invasive species (Figure 4.14). *Typha* reached a maximum emission of 4.6 mg C m^{-2}hr^{-1} in September 2014 and 1.4 mg C m^{-2}hr^{-1} in July 2015 and *Phragmites* reached a maximum of 4.6 mg C m^{-2}hr^{-1} in June 2014 and 0.9 mg C m^{-2}hr^{-1} in September 2015. In contrast, *Eriophorum* only reached a maximum emission of 0.1 mg C m^{-2}hr^{-1} in May 2014 and June 2015 (Figure 4.15).
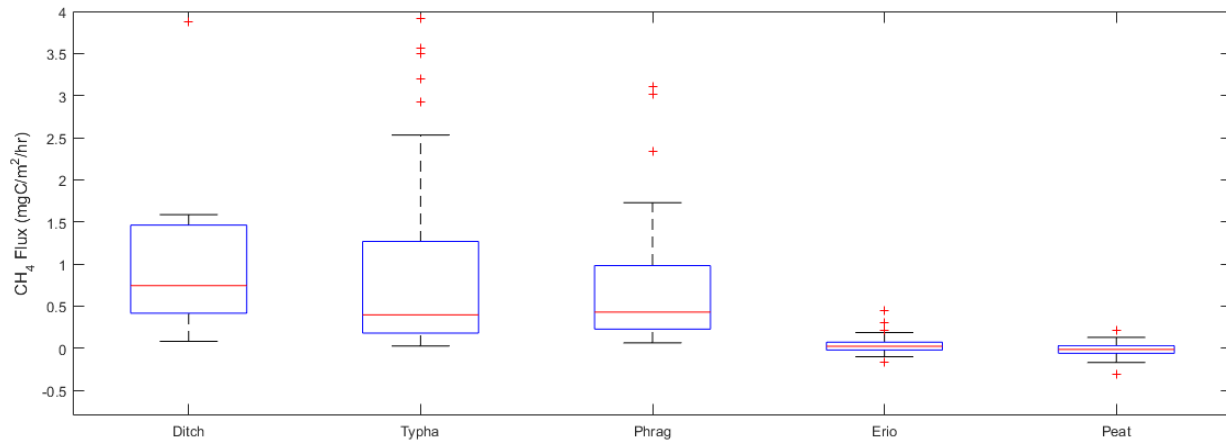
Figure 4.14. Comparisons of plant community scale CH$_4$ fluxes between the bare ditch, *Typha, Phragmites, Eriophorum* and bare peat field plots (2014 and 2015 combined).
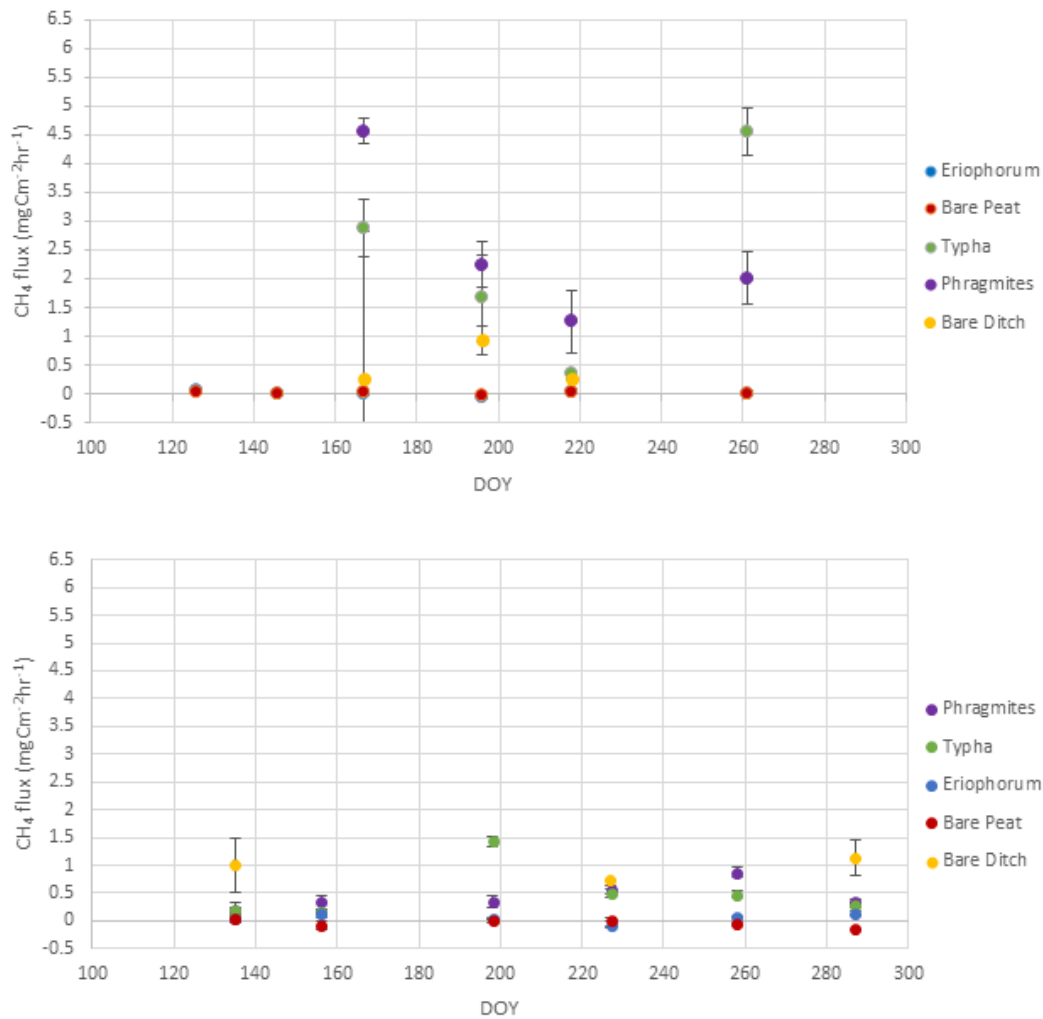
Figure 4.15. Plant community scale CH₄ measurements for 2014 (top) and 2015 (bottom).

4.6 Environmental Controls on CH₄ Exchange

4.6.1 Ecosystem Scale

We performed linear regression between the CH₄ fluxes and the environmental variables by considering one, two and three week periods prior to and including each of the spikes we

observed in CH_4 release. We assumed that the environmental conditions were similar throughout these periods of time. However, no significant linear relationships between the CH_4 fluxes and the environmental variables were observed for any of the periods taken into account. Multiple regression did not enhance any of the relationships. However, the methane burst in March *did* correspond with a brief period where the snow had melted slightly during the spring. According to Bubier et al. (2002), bursts of methane occur in late winter or spring when snow melts, where methane trapped under the accumulated snow or ice is released into the atmosphere.

4.6.2 Plant Community Scale

Relationships between CH_4 fluxes and the environmental variables at the plant community scale were determined separately for each of the three plant species. There were only significant linear relationships between the CH_4 fluxes from the reeds and soil moisture. Soil water content explained 97% of the variance in CH_4 fluxes in 2014 ($P = 0.01$) and water table position explained 80% ($P = 0.10$) and 73% ($P = 0.07$) of the variance in CH_4 fluxes from the reeds in 2014 and 2015, respectively.

We did not test for non-linear relationships of CH_4 fluxes with environmental variables at the plant community level, because determining whether the fit is significant is very difficult when using small sample sizes.

Chapter 5: Discussion

5.1 Spring CO_2 Exchange

Both years showed similar trends in daily average NEE, but with some distinct differences. There was an initial release of CO_2 at the beginning of April in both years (around day 90). This was associated with the snow melt period (Figure 5.1), where the CO_2 accumulated under the snow or ice throughout the preceding winter was released to the atmosphere (Bubier et al. 2002). Both the depth of snow and the shortwave albedo are seen to decrease at the same time that this CO_2 release was measured by the tower. In comparing the two years, the depth of snow on the ground in April 2014 was greater than in 2015 (Environment Canada 2015), but there was a slightly more rapid snowmelt in 2015. The shorter snowmelt period may explain why there was a greater maximum CO_2 release in 2015, however, the CO_2 release period lasted longer in 2014 and the cumulative CO_2 release over the two snowmelt periods was similar (both $\sim 8gCm^{-2}$). Furthermore, the period with the greatest snow melt occurred roughly 10 days earlier in 2014 than in 2015. We believe that the difference in snow depth coupled with the difference in length and timing of the snowmelt period may explain why the trend in CO_2 exchange was different between the two years despite the similar total cumulative CO_2 release. Studies have found bursts of CO_2 coincident with the timing of snow and ice melt, and since the snowmelt period acts as a prolonged wet period, NEE tends to be high during this time, and then decreases rapidly directly after snowmelt (Bubier et al. 2002; Lafleur 2009). Aurela et al. (2004) found that the timing of the snowmelt can greatly affect the annual CO_2 balance in a subarctic fen, where a later start to the growing season resulted in a smaller annual uptake and a larger annual release in CO_2. Humphreys et al. (2014) also found that NEP became negative earlier at an ombrotrophic bog in Eastern Ontario when an earlier snowmelt period occurred.
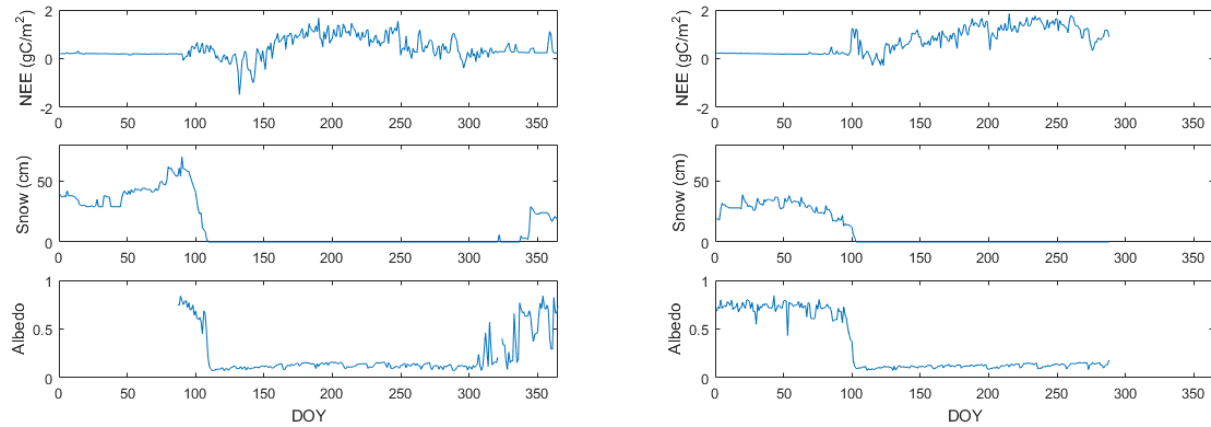
Figure 5.1. Net ecosystem exchange of CO_2 (NEE) (top), snow depth (middle) and shortwave albedo (bottom) in 2014 (left) and 2015 (right).

The initial release of CO_2 in April was followed by a brief period of net uptake in mid-May of both years, which may be explained by many factors. The uptake in CO_2 corresponds to periods when peat temperatures at 10 cm depth started to increase in both years. This increase was more rapid in 2014 than in 2015. There was also a greater depth of snow, which acted as an insulator, explaining the higher peat temperatures observed in 2014. There was a large increase in soil water content around day 125 in 2015, which was associated with increasing peat temperatures at 10 cm depth. However, there was more snow available to melt in 2014, so the peatland was wetter in May 2014. Furthermore, measurements pre-snow melt in 2015 indicate that peat temperatures in the ditch were higher than peat temperatures in the bare peat fields. The plants present in the ditches, which are highly productive during the growing season, would have benefitted from the higher peat temperatures and higher soil moisture in 2014, promoting a faster initial growth (hence explaining the higher uptake in May of 2014).

Wind direction may also have played a role in the increased uptake we observed in May of both years, with tower measurements capturing CO_2 fluxes that showed increased uptake from

the productive species during these periods. Based on the chamber measurements, the *Eriophorum* showed its highest uptake in May of 2014 and had also begun taking up CO_2 in May 2015. The light response curve parameters for *Eriophorum* in 2015 showed a less negative GP_{max} and a stronger R compared to the light response curve parameters in 2014, also indicating that the *Eriophorum* were more productive in 2014, where there was more CO_2 uptake for the same light intensity. We didn't have the ability to measure the CO_2 exchange of the *Typha* and *Phragmites* directly at the site. However, visual inspection indicated that *Typha* and *Phragmites* had not yet begun to grow until after this period. Bonneville et al. (2008) found that a temperate cattail marsh (*Typha latofolia*) in eastern Ontario only began showing increased uptake in June. In contrast, Zhou et al. (2009) found that a reed wetland (*Phragmites australis*) in the northeastern part of China showed a rapid increase in CO_2 uptake in May. However, this peatland lies within a warm temperate climate region, which experiences warmer annual temperatures and a greater number of frost-free days than eastern Quebec, indicating that the growing season for *Phragmites* would have started earlier in that study. We did not see increased uptake in the direction of the *Phragmites*. Instead, we found increased CO_2 release from the deeper ditches containing *Phragmites* that have not yet started to grow in May. This compares with Waddington et al. (2010) who found that the average ditch respiration was generally higher than the average peat respiration along a ditch to ditch transect in an unrestored, cutover peatland.

At the ecosystem scale, the measurements indicating larger uptake were captured from the directions containing a greater abundance of *Eriophorum* in the ditches in both years, which lie within wind directions of 30° and 60°. In contrast, the measurements indicating larger CO_2 release were captured from the directions containing the deeper ditches in 2015, which lie within

wind directions of 180° and 330°. This would explain why results showed a linear relationship between NEE and wind direction. The relationship was stronger in 2015 because both the directions showing increased uptake and increased release were captured within the month of May. This may have led to the lower uptake in CO_2 we found in May of 2015 as well. However, since all the vegetation combined only covers about 8% of the peatland, it is difficult to attribute the higher uptake in May to directional bias in tower measurements from areas of higher vegetation cover. Therefore, although there is some evidence that wind direction may explain some of the variability in NEE and GEP, other variables should be taken into account, such as peat temperatures and soil moisture.

5.2 Warm Season CO_2 Exchange

5.2.1 Ecosystem Scale

The inter-annual variability in NEE exchange between the two years, from June to October, may be explained by differences in air and peat temperature. June and July of 2014 experienced much warmer air temperatures than in June and July of 2015 (Environment Canada 2015). Although peat temperatures in 2014 were comparable to those of 2015 overall, peat temperatures started to thaw earlier (day 144) at 40 cm depth in 2014 than in 2015 (day 156). Therefore, we believe that the higher air and peat temperatures in June and July of 2014 and the difference in timing of the soil thawing may have both been responsible for the higher CO_2 release in June and July of 2014. In contrast, highest CO_2 release in 2015 was in the fall periods of September and October, when air temperatures were the highest.

5.2.2 Plant Community Scale

Chamber-based CO_2 measurements for *Typha* and *Phragmites* were obtained from the literature. Reported values indicate that both species can be very productive, especially during the warm season months (Brix et al. 2001). A study performed at the plant community scale (using static chambers) in a restored cutaway peatland in Ireland, showed that *Typha* took up CO_2 during both warm seasons of the study. Values ranged between 0.1 and 0.5 $gCm^{-2}hr^{-1}$ (Wilson et al. 2007). Strachan et al. (2015) also found that a *Typha* marsh in Eastern Ontario was an overall sink of carbon, showing a net uptake of 97 (+/- 57) $g\ C\ m^{-2}yr^{-1}$. In contrast, Rocha and Goulden (2008) found that a freshwater *Typha* marsh in California flipped between a sink (-251 $g\ C\ m^{-2}$) and a source (515 $g\ C\ m^{-2}$) of CO_2. However, that site is located in a Mediterranean climate region, which experiences much lower annual precipitation amounts and much higher annual temperatures than eastern Quebec, indicating that the site was seasonally dry and may explain why the site flipped between a sink and a source of CO_2.

The CO_2 exchange for *Phragmites* is not well documented. To the best of our knowledge, no studies have found an annual release of CO_2 from peatlands dominated by reeds. Two studies, located in Northeast and Central China, have analyzed the carbon dioxide exchange from reed wetlands, and both report an annual CO_2 uptake of -65 $g\ C\ m^{-2}yr^{-1}$ (Zhou et al. 2009) and -261 $g\ C\ m^{-2}yr^{-1}$ (Han et al. 2012), respectively.

Since our site has a very low overall vegetation cover of < 8%, the overall release in CO_2 to the atmosphere, especially from the vast bare peat cover, exceeds the uptake of CO_2 in both years. The site is thus an overall source of CO_2.

5.3 Environmental Controls on CO_2 exchange

The trends in daily average NEE follow the general corresponding trends in daily average ER for the majority of the time in both years, whereas NEE follows the general corresponding trend in daily average GEP (in both years) *only* during the brief periods when the site experiences increased uptake in May. Considering that ER is modelled based on empirical relationships with peat temperature, the variability in NEE throughout and across the two years is primarily temperature driven. This is especially important in April when the initial release in CO_2 release occurs with the snowmelt period. However, since GEP is the residual between the measured NEE and the modelled ER, the brief period of increased uptake in May of both years is driven by the vegetation present or, as mentioned previously, by a combination of environmental variables, such as wind direction, soil moisture and temperature.

Strong relationships between NEE and ER and air and soil temperature have been reported in multiple studies (Nilsson et al. 2008; Strack 2008). For example, Bubier et al. (2003) found that ecosystem respiration was strongly correlated with air and peat temperature at 5 cm, where higher temperatures led to increased CO_2 release, especially in the wetter year of the study. Results from our study show the same; air and peat temperature were the most significant drivers of both NEE and ER and 2015 was a higher source of CO_2 to the atmosphere due to the higher temperatures overall. There was also a discernable hysteresis effect created by the difference between the daytime NEE-peat temperature and the nighttime NEE- peat temperature relationships. At night, where longwave radiation is dominant, CO_2 respiration is highest and depends primarily on peat temperatures, whereas during the day, where shortwave radiation is dominant, the amount of incoming solar radiation (measured as PAR) drives the amount of CO_2 uptake by the vegetation, and peat temperatures drive only the amount of respiration from the

bare peat surface. In previous studies, such diurnal responses in CO_2 exchange with peat temperature have been attributed to various phenomena. Phillips et al. (2011) found that a hysteretic temperature response was attributed to a lag in heat transport processes between the surface and the peat at a certain depth, whereas Updegraff et al. (1998) found that diurnal temperature responses to CO_2 exchange were better associated with a change in microbial response. In this study, it seems more likely that the hysteresis effect would be associated with the lag time in soil heating processes due to the difference between daytime and nighttime peat temperature changes. However, soil moisture may also play a role. Niu et al. (2011) found that soil water content contributed to a hysteresis response in CO_2 exchange, but only at some sites and on a seasonal scale rather than a diurnal scale. The difference in soil moisture throughout the year at our site may explain the difference in the hysteretic temperature responses we observed between the different months of the growing season.

Pelletier et al. (2011) found that the relationship between ER and temperature differed with topography. They report a stronger relationship with air temperature on the high and low hummocks and a stronger relationship with peat temperature at 40 cm on the lawns. However, in our study, topography was not as important. There was no significant difference in CO_2 exchange in either year of the study between the three topography positions on the berm (§ 4.1).

Water table position and soil moisture, both intricately related, have been shown to drive a large portion of the variability in CO_2 exchange (Strack & Price 2009), but the strength of the relationship may differ across sites. Roulet et al. (2007) found that a deeper water table at an ombrotrophic bog in Eastern Ontario led to less uptake of CO_2, whereas Waddington and Roulet (1996) found lower CO_2 uptake in wetter areas in a boreal peatland in northern Sweden. A deeper water table has also been associated with greater ER (Pelletier et al. 2011). However,

Bubier et al. (2003) only found a strong relationship between water table position and ER in the drier summer of their study.

Manual water table positions and continuous measures of volumetric soil moisture and rainfall events have proved to act as good proxies for understanding the link between changes in the hydrology and carbon dynamics at the site. According to Strack and Price (2009), soil moisture can significantly affect the C balance in peatlands because it controls both the photosynthesis and respiration components. Although soil moisture on its own did not seem to be a prominent driver of CO_2 exchange in this study, soil moisture did have an impact on the CO_2 exchange when coupled with other environmental variables. For example, the strength of the relationship with NEE increased when precipitation was coupled with both air and soil temperature, explaining 52% of the variance in NEE when multiple regression was used ($P <$ 0.001). This makes sense since the soil moisture was less constant in 2014 and the precipitation events were more sporadic. Multiple regression did not enhance any of the relationships between NEE and the environmental variables in 2015, but when soil and air temperature were coupled with soil water content, the strength of the relationship with ER *was* enhanced, explaining 92% of the variability ($P < 0.001$).

5.4 Seasonal Variability in CH_4 Fluxes

The extraction of peat normally decreases methane release to the atmosphere as the oxidation zone is increased in the bare peat fields (Mahmood and Strack 2011). Using the continuous measurements of CH_4 in 2015, we found that this was the case for the majority of the time, where mean CH_4 emissions hovered near zero. However, there were short periods of time where significant spikes in daily average CH_4 fluxes occurred (Figure 4.13).

There may have been a spike in CH_4 flux in July 2015 as well, but due to low instrument signal strength, the data for July were rejected. We scaled up the CH_4 emissions obtained at the plant community scale for each month of the growing season in 2015 (May to October) to the ecosystem level, providing a means in which the missing data in July can be inferred. Results at the plant community scale suggest that not only did a spike in CH_4 flux occur in July, but that July may have contained the maximum CH_4 release in 2015. We can also infer from our chamber-based measurements that June contained the maximum CH_4 release in 2014. It is important to note that due to the low vegetation cover at our site, the magnitude of the methane released is small compared to the overall CH_4 released from a natural peatland (Strack et al. 2016). Nevertheless, the periods of increased CH_4 release at the site are significant, and can be attributed to the presence of the invasive species.

5.5 Environmental Controls on CH_4 Exchange

Multiple studies suggest WT position to be the main driver of CH_4 exchange (Bubier 1995; Roulet et al. 1992), especially during the growing season months. On the other hand, Mahmood and Strack (2011) found that water table depth and soil temperature act as interacting controls on CH_4 fluxes when certain ranges of water table depth were considered. Goodrich et al. (2015) found that the relationship between CH_4 fluxes and peat temperature was strong when the water table was within 10 cm of the surface, but when the water table dropped dramatically, the CH_4 fluxes were no longer dependent on peat temperature. In our study, the ditches provided the invasive species with favourable environmental conditions in which to establish themselves. Consequently, the CH_4 fluxes from the ditches, and from the invasive species within, were significantly higher than the CH_4 fluxes from the bare peat and the *Eriophorum* on the adjacent

berms ($P < 0.01$). However, we found no significant difference in CH_4 exchange between the three topography differences along the berm transect (all $P > 0.65$), nor any interacting effects with the environmental variables at the plant community scale.

5.6 Implications for Restoration

5.6.1 Trace Gas Exchange

Results at the plant community scale indicated that the invasive species (*Typha* and *Phragmites*) emitted more CH_4 than the early colonizing *Eriophorum* and the bare peat across both growing seasons. The ditches (without vegetation) were also a higher source of CH_4 than the bare peat and *Eriophorum* as well. This may be attributed to the lack of restoration efforts at the site. For example, it has been suggested that should drainage ditches remain unblocked in unrestored peatlands, the peatland will be a higher source of CH_4 than if the drainage ditches were no longer present, due to higher soil moisture conditions and shallow water table positions (Roulet and Moore 1995; Waddington and Day 2007). Furthermore, it has been suggested that the establishment of invasive species is more likely to occur in the drainage ditches left behind after extraction, and may exacerbate CH_4 emissions to the atmosphere due to the facilitated transport of methane gas through the aerenchyma of the vascular plant species (Hudon et al. 2005; Zhao et al. 2013). Thus, CH_4 emissions may increase should the invasive species continue to spread into the surrounding peat fields. Therefore, using an upscaling approach with the chamber-based measurements, we developed a simple model to estimate the CH_4 emissions in various scenarios of *Phragmites* colonization. The site is currently an annual source of approximately 0.8 g C m^{-2}. However, it is likely that the CH_4 release in 2014 was larger than in 2015 based on the chamber-based measurements. If the invasive species fill the length of every

ditch, the CH_4 emissions would be 1.55 times larger than present. If the *Phragmites* spread out from the ditches to colonize the surrounding banks to a similar extent that they have from the one ditch presently in the study, the methane emissions are estimated to be 6 times larger than present. CH_4 emission would be highest in the most unlikely case that the invasive species spread and cover the entire peatland whereby, the emissions would be 20 times larger than present. These numbers are rudimentary at best but give a general indication of how much influence the invasive species have on the methane emissions at the site. Since we were unable to obtain direct measurements of CO_2 fluxes at the plant community scale for the invasive species, we estimated how the CO_2 dynamics would change based on the measurements obtained from the literature (Bonneville et al. 2008; Rocha and Goulden 2008; Zhou et al. 2009). We found that with the present cover, the invasive species would contribute to an annual uptake of about 4.6 g C-CO_2 m^{-2}, which is larger than the current annual CH_4 emissions. We suggest that future studies measure fluxes of both greenhouse gases with several chamber measurement periods across multiple growing seasons to confirm these findings. It seems possible though that the amount of carbon taken in through photosynthesis may be enough to offset the amount of CH_4 released should the invasive species spread out to colonize the surrounding banks. However, even though the site may be an overall sink of carbon in that case, a site colonized by *Phragmites* and/or *Typha* is not the result that managers are attempting to achieve from a restoration management perspective (Mahmood and Strack 2011). Presently, the site still remains a large source of CO_2, mainly due to large rates of respiration from the vast bare peat cover and is thus still an overall source of carbon.

Values of NEE have been reported for peatlands that have undergone various stages of restoration (Table 5.1). Peatlands in their natural state represent the largest sink of CO_2, whereas

peatlands that have been extracted and where no restoration efforts have been conducted, represent the largest source of CO_2. Restored peatlands have the ability to gain back their C storing capacity, but it may take years for progress to be seen. For example, Bois-des bels (BDB), a peatland that was restored 16 years ago in the same area as SAK, was an overall source of CO_2 (148 g C $m^{-2}yr^{-1}$) ten years post restoration (Strack and Zuback 2013). However, the contemporary cumulative NEE for BDB is an average uptake of CO_2 around -95.0 g C $m^{-2}yr^{-1}$ (Personal Communication, Kelly Nugent, McGill University). As a comparison, a well-documented natural ombrotrophic bog in Eastern Ontario shows similar cumulative NEE values (Humphreys et al. 2014) to BDB, confirming that BDB has regained its C-storage ability 16 years post-restoration.

It has been suggested that CO_2 release from extracted, unrestored peatlands can decrease over time since the halting of extraction (Schothorst 1977). However, this is in sharp contrast to the results found in this study and others (Waddington et al. 2002; Wilson et al. 2015). SAK had remained an overall persistent source of CO_2 16 years after extraction had ended, showing an increase in CO_2 emissions across the study period with annual cumulative NEE values of 153 g C m^{-2} in 2014 and 241 g C m^{-2} in 2015.

Table 5.1. A comparison of NEE between natural, restored and unrestored peatlands

Peatland Type (location)	NEE ($gCm^{-2}yr^{-1}$)	Restoration Type (time since restoration)	Reference
Raised bog (Canada)	-2 to -112	Natural	(Roulet et al. 2007)
Raised bog (Sweden)	-2 to -10	Natural	(Waddington and Roulet 2000)
Treed bog (Canada)	1 to -0.3	Cutover portion	(Waddington et al. 2010)
	-0.5 to -2	Restored portion (1-3 years)	
Poor fen (Sweden)	-13 to -15	Natural	(Nilsson et al. 2008)
Treed mire (Finland)	3	Clear felled and drained	(Mäkiranta et al. 2010)
Raised bog (Southern Germany)	127	Restored (6 years)	(Drosler 2005)
	236 to 401	Unrestored	
	-71	Natural	
Raised bog (Finland)	-87	Restored (10 years)	(Kivimäki et al. 2008)
	41	Bare Plots	
Raised bogs (UK, Ireland)	90 to 304	Unrestored (industrial peat extraction), ditches remain	(Wilson et al. 2015)
Bog-poor fen complex (Quebec, Canada)	88 to 399	Post-cutover	(Waddington et al. 2002)
Raised bog/Minerotrophic fen? (Canada)	152 to 241	Unrestored	This Study

Values of CH_4 emissions have also been reported for peatlands that have undergone

various stages of restoration (Table 5.2). Natural peatlands emit the largest amount of methane to

the atmosphere, whereas extracted, unrestored peatlands emit the smallest amount of methane to

the atmosphere. However, extracted sites that still contain ditches and invasive species emit slightly more than unrestored peatlands where the ditches have been blocked (Waddington et al. 2010).

Table 5.2. A comparison of CH$_4$ emissions between natural, restored and unrestored peatlands

Peatland Type (location)	CH$_4$ (gCm^{-2}yr^{-1})	Restoration Type (time since restoration)	Reference
Raised bog (Finland)	0.41 to 0.95	Restored by rewetting (3 years)	(Tuittila et al. 2000)
subarctic fen (Schefferville, Quebec)	0.1 to 0.5	Natural	(Moore and Knowles, 1987)
Rich fen (Canada)	0.6 to 3.5	Naturally revegetated peat field	(Mahmood and Strack 2011)
	1.3 to 53	Remnant ditch	
Treed bog (Canada)	0 to 1.3	Restored peat field	(Waddington and Day 2007)
	1 to 40	Restored remnant ditch (2 years)	
	-0.03 to 0.1	Unrestored peat field	
	1.2 to 13	Unrestored ditch	
Boreal oligotrophic fen (Southern Finland)	12.6	Natural	(Rinne et al. 2007)
Raised bog (Sweden)	3	Natural	(Waddington and Roulet 2000)
Raised bog/Minerotrophic fen? (Canada)	0.8	Unrestored	This study

5.6.2 *Typha* and *Phragmites*

Relative to other extracted sites, SAK is considered a dry site due to the ditches not being blocked. Typical mosses associated with peatlands like *Sphagnum* do not thrive well in such dry conditions, but competitive plant species that have expansive root structures like *Typha*, *Phragmites* and *Eriophorum* can more easily access a water supply that may be well below the surface (Hudon et al. 2005; Seabloom et al. 2001). The presence of the ditches at SAK provided slightly wetter areas where the invasive species were able to preferentially establish themselves. The subsequent fluctuations in water table depths following precipitation events allowed the invasive species to continue to thrive over the other species. Squires and Van der Valk (1992) found that *Typha* is slightly more restrictive, and tend to not be present where water tables drop more than 30 cm below the surface. This may explain why the *Typha* have remained confined to the ditches, whereas the *Phragmites* have started to spread outside of the ditches and into the peat fields. However, this may also limit the extent of the colonization. Zhao et al. (2013) found that *Phragmites* can thrive where the water table falls as low as 60 cm below the surface. This study found water table depths as low as 72 cm below the surface. If the peat fields are too dry, even the *Phragmites* explansion will slow down.

5.6.3 Peat and Pore Water Chemistry

We believe that the initial establishment of the reeds and cattail occurred due to the shift in nutrient dynamics at the site post extraction. Following extraction, the peatland now sits at a lower elevation and groundwater is able to flow into the peatland. The resulting peatland may have more fen-like properties rather than the ombrotrophic bog that was originally present.

Peat and pore water can take on the characteristics of the underlying glacial deposits (Siegel et al. 1995). Toward the surface of the peat core, the concentration of hydrogen ions was

high and the pH of the peat was in the range of a bog, but at greater depths, the concentration of hydrogen ions decreased, resulting in a higher pH. Furthermore, there was a distinct difference in nutrient concentrations in the pore water between the two sampling periods. When the pore water was sampled in September, where water had ample time to come in contact with the underlying soil layer, the nutrient concentrations and the pH of the pore water were similar to that of a moderate-rich fen (Chee and Vitt 1989; Vitt et al. 1995). When the pore water was sampled from the bank-full ditches in April and May, the nutrient concentrations were similar to that of a bog because the water during that period was primarily from precipitation and it had not had time to come in contact with the underlying soil layers. Other studies have found the same result in extracted peatlands (Wind-mulder and Vitt 2000).

It is also possible that the extraction process can lead to permanent chemical damage. If there was a sufficient amount of peat removed during the extraction process so as to reach the mineral peat below, which may be the case in this study, the re-establishment of desired species, like *Sphagnum*, and hence the restoration of the site, may not even be possible (Wind-mulder et al. 1996). The shift in nutrient concentrations may have facilitated the establishment of the invasive species as well (Engloner 2009).

Chapter 6: Summary and Conclusion

Northern peatlands are significant long-term stores of C. However, anthropogenic disturbances (e.g. peat extraction) can drastically alter the C dynamics of a peatland; consequently, they may change from a sink of C to a source (Strack and Zuback 2013). Previous studies have analyzed the benefits and advantages of various restoration management practices from extracted peatlands (Girard et al. 2002; Lavoie et al. 2003; Waddington et al. 2010), but the carbon exchange that results from unrestored peatlands is not well understood. Therefore, the objectives of this study were: (1) to continuously measure carbon dioxide and methane fluxes at the ecosystem scale over two years; and, (2) to determine which plant communities contribute to CO_2 and CH_4 exchange, from a post-extracted, unrestored peatland in eastern Quebec.

Using eddy covariance techniques, we determined that although CH_4 emissions decreased following extraction (contributing less than 1% to the total C released), SAK remained a persistent source of C to the atmosphere (from an ecosystem scale perspective). The site was extracted in 1999 and no restoration efforts were attempted. Consequently, the site consisted of 94% bare peat cover, with remnant ditches that had not been blocked, both of which released a large amount of CO_2 to the atmosphere. The small amount of vegetation cover did not take in enough CO_2 through photosynthesis to compensate for the large amount of CO_2 respiration from the ditches and the bare peat fields. The site was thus an overall source of C, releasing an annual total of 154 g C m^{-2} and 242 g C m^{-2} (CH_4 and CO_2 emissions combined) in 2014 and 2015, respectively.

2015 was a higher source of CO_2 to the atmosphere than 2014. This was attributed to many factors. There was a higher period of uptake in May 2014 following the snowmelt period,

which corresponded with higher peat temperatures and higher soil moisture that would have benefitted the highly productive vegetation in the ditches, promoting a faster initial growth in 2014. Peat temperatures at 10 cm and 40 cm depth started to thaw earlier in 2014, which also contributed to an earlier start of the growing season. Throughout the rest of the growing seasons (June through October), air temperatures were generally higher in 2015 which corresponded with greater values of ER than in 2014, and since the trends in NEE followed the trends in ER very closely in both years, the higher temperatures corresponded with a greater annual release of CO_2 (NEE) in 2015. Furthermore, rainfall events were more sporadic in 2014 and hence soil moisture was generally lower throughout the growing season in 2014. Overall, the warmer and slightly wetter growing season in 2015 resulted in a greater release of CO_2 to the atmosphere. This compares with other studies who have also found that cutover, unrestored peatlands are persistent sources of CO_2 (Waddington et al. 2002; Wilson et al. 2015). The results of this study were compared to a restored peatland of the same age in the same area. Contrary to SAK, the restored peatland acted as an annual sink of carbon, showing that restoration efforts can allow extracted peatlands to gain back their carbon storing ability, even if it may take years before restoration efforts show some progress.

The plant community scale measurements also showed that SAK was a large source of C to the atmosphere in both years of the study. Although the *Eriophorum* proved to be quite productive throughout both growing seasons, the vast cover of bare peat led to respiration values greatly exceeding the CO_2 taken in by the vegetation in both years. The remnant ditches provided a slightly wetter area where the invasive species had preferentially established. Both the presence of the invasive species and the remnant ditches increased the CH_4 emissions at SAK, with 2014 being a larger source of CH_4 (from a plant community scale perspective). By developing a

simple model, we determined that if the invasive species should colonize a greater surface area of the peatland, the CH_4 emissions will be further exacerbated.

6.1 Scope for Future Research

We found that the net CO_2 and CH_4 exchanges at the site were affected by various environmental variables such as soil moisture and peat temperatures. Although soil water content and rainfall events proved as good proxies for explaining the effects of soil moisture on the net exchanges at the site, a continuous measure of water table depth would have provided further confirmation of their effects, especially when describing inter-annual differences. Therefore, future studies should continuously monitor water table positions to determine its effects on the net C exchange in other restored and unrestored peatlands.

The chamber based measurements also helped confirm the prominent drivers of net C exchange at the site, but some relationships were not statistically significant. This was possibly due to the small sample size of chamber-based measurements that were obtained. Therefore, future studies should sample both greenhouse gases frequently for all plant types present, while also considering the effects of topography and hydrological differences within the peatland site.

Roulet et al. (2007) suggest that when evaluating a peatland's carbon sink/source potential, it is important to include the contribution of dissolved organic carbon (DOC) efflux into the total carbon budget. Waddington et al. (2008) report DOC export rates between 4 and 10 g C $m^{-2}yr^{-1}$ from a cutover site, which falls within the range of reported values for other northern peatlands (Blodau 2002). We were unable to obtain measures of DOC efflux from SAK because the site was very dry and there was not enough flow through the installed weir to determine an

efflux rate. However, we believe that the contribution of DOC efflux would have been negligible compared to the magnitude of CO_2 respiration. Strack and Zuback (2013) found the same; the contribution of CO_2 release was much greater than the contribution of DOC efflux in the unrestored section of BDB, but the DOC efflux was greater than the contribution of CH_4 emissions. It would have been interesting to compare the contribution of DOC export at SAK with the contribution of CH_4 flux, especially if the various scenarios of *Phragmites* colonization were taken into account. We believe that should the invasive species continue to spread, the contribution of CH_4 emissions may also exceed the contribution of DOC efflux at SAK.

6.2 Conclusion

This study provided a baseline case with which to compare the results from restored peatlands, thereby depicting the true net benefit (from a carbon uptake perspective) of implementing restoration practices. While the tower fluxes provided an ecosystem-scale measurement of the net exchanges, plant community scale measurements helped determine the processes and mechanisms controlling carbon uptake and methane release, and helped determine how shifts in vegetation and peatland function affected the hydrology-carbon feedback.

It is anticipated that the results from this study will provide managers with a better understanding of the need to balance the economic benefits of peat extraction while minimizing the amount of greenhouse gases released to the atmosphere, and how post-extraction peatland restoration is the key to achieving this balance.

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