

Spatial variability of carbon emissions within a drained lake basin and its surrounding tundra,
Illisarvik, Northwest Territories.

by

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Abstract

This study investigates the spatial and temporal variations of carbon emissions and their controls at a 38-year-old drained lake basin on Richards Island, NT. Greenhouse gas fluxes were collected from plots with different vegetation throughout the basin and in surrounding tundra during the growing season. Wet Sedge sites were significant sources of methane (7 to 355 nmol m⁻² s⁻¹) while most other sites were sinks. Carbon dioxide fluxes varied from 0.5 to 13 μmol m⁻² s⁻¹ with highest fluxes outside the basin. Air temperature was positively correlated with carbon dioxide emissions at the majority of sites while soil moisture and vegetation type were the main controls on methane fluxes. Bulk age of the respired carbon dioxide was mostly modern, reflecting rapid cycling of recently sequestered carbon. Overall, carbon emissions were similar to those recorded at other tundra sites.

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1 Chapter: Overview and Introduction

1.1 Introduction

This thesis investigates emissions of carbon dioxide (CO₂) and methane (CH₄) as well as nitrous oxide (N₂O) from a drained lake basin in the Canadian Southern Arctic. These are important greenhouse gases that influence global climate. Permafrost environments store carbon (C) in a frozen state that slows the decomposition process by microorganisms (Davidson and Janssens, 2006). Over time, C is accumulated in the active layer if plant productivity exceeds decomposition. Since deglaciation, the Arctic has generally been a net sink for atmospheric carbon resulting in large, globally important, C stocks in these permafrost-affected soils (Hugelius et al., 2014). However, Arctic and subarctic landscapes are diverse and it is uncertain how different the C cycling through the soil-plant-atmosphere continuum is among these ecosystems.

Thermokarst landscapes are especially diverse and climate warming has the potential to increase the rate of drainage of thermokarst lakes (Marsh et al., 2009). To improve our knowledge of the post-drainage C budget of a thermokarst lake, a field study was conducted within the “Illisarvik” drained lake basin on Richards Island, NT. This lake was recently drained (< 50 years BP) (Mackay, 1997), offering temporally controlled settings for this research. The site has a very high spatial variability in terms of vegetation and soil moisture. Both factors may greatly influence the amount of C released to the atmosphere through soil and plant respiration (Davidson and Janssens, 2006; Adams et al., 1990). The field investigation involved making measurements of CO₂, CH₄ and N₂O gas fluxes and analyzing their relationships with soil and atmospheric weather variables as well as plant and soil characteristics to better understand C cycling at this site and its similarities and differences with sites around the Arctic.

1.2 Background

The northern circumpolar area contains a large proportion of the world's soil organic C. Hugelius et al. (2014) estimate that permafrost soils hold $\sim 1035 \pm 150$ Pg C within the top 0 – 3 m, which is twice as much as the atmosphere and equivalent to 50% of the world's below ground organic C pool. Arctic tundra is rich in soil organic C as a result of a small but long-term imbalance between plant productivity and microbial decomposition where the latter is limited by poorly drained and/or cold soil conditions (Belshe et al., 2013). In addition, processes such as burial (and incorporation into permafrost,) by cryoturbation, sedimentation, and peat accumulation lead to physical protection of this organic matter (ACGR, 1988; Bockheim, 2007).

The climate is warming at twice the rate in the Arctic as elsewhere on the planet (IPCC, 2013) and this warming may impact the C budget of this region in unpredictable ways. Warming may lead to greater plant productivity as a result of longer growing seasons and more favourable conditions for growth but it may also lead to greater rates of decomposition of both newly sequestered and old soil organic matter with emissions of carbon dioxide (CO₂) and methane (CH₄) offsetting any increases in CO₂ uptake by plants (Belshe et al., 2013; Schuur et al., 2009). To understand the role of the Arctic in the global C budget, it is important to have knowledge of the variability in CO₂ and CH₄ emissions due to plant respiration and microbial decomposition and the factors that influence this variability across Arctic landscapes.

CO₂ and CH₄ emissions have been found to be highly variable over time and space in the Arctic. For example, measured CH₄ fluxes vary from small to large emissions in polygonal terrain (0.155 to > 1397.6 mg CH₄ m⁻² day⁻¹; Martin et al., 2017). Other studies have found lower values across the Arctic with mean values of 29 mg CH₄ m⁻² day⁻¹ in an Alaskan wet meadow tundra (Fan et al., 1992) and 151.2 mg CH₄ m⁻² day⁻¹ in a fen located in an Arctic valley in Greenland

(Christensen et al., 2000). Microbial methane production and oxidation rates depend on the degree of soil saturation and thus emissions rates are controlled by environmental variables such as soil moisture and water table depth. Bartlett et al. (1992) found a relationship between soil moisture and CH₄ flux with emissions two orders higher in magnitude in a wet meadow tundra (15.5 to 426 mg CH₄ m⁻² day⁻¹) than in dry upland tundra (-2.1 to 18.1 mg CH₄ m⁻² day⁻¹). Other studies found that the presence of sedges enhance methane emissions due to their capacity to act as a gas conduit for CH₄ (Christensen et al., 2000) and that higher rates of emission comes from sites with sedges (Fan et al., 1992) or that once clipped, CH₄ emissions diminish significantly (Verville et al., 1998). CO₂ emissions due to respiration processes are also controlled by hydrological variables, vegetation, and temperature, among other variables (Welker et al., 2000). CO₂ emissions are much higher than CH₄ emissions ranging between 0.07 and 46.79 g CO₂ m⁻² day⁻¹ for polygonal terrain in Tuktoyaktuk Coastlands (Martin et al., 2017) or between -0.89 and 5.03 g CO₂ m⁻² day⁻¹ for thaw ponds located near Bylot Island and Nunavik in the eastern Canadian Arctic and Subarctic (Laurion et al., 2010).

The great variation in C fluxes in the Canadian Low Arctic is influenced by spatial variations in hydrology, vegetation and other variables that impact C cycling processes. This is particularly the case for ice-rich permafrost regions which include landscape features such as pingos, thermokarst lakes, and polygonal terrain (French, 2007). Carbon fluxes from thermokarst lakes and ponds have received scientific attention as these features can be important sources of CH₄ (Zimov et al., 1997; Walter et al., 2006; Karlsson et al., 2013). These water bodies have the potential to drain catastrophically with diversion of water through inter-connected ice wedges between lakes. This process is enhanced with higher temperatures and thawing of these ice wedges or by human disturbance such as construction of roads or pipelines (Mackay, 1988).

Drained lake basins occupy 47% of the Alaskan coastal plain (Zona et al., 2010). After a lake has drained, the lake bottom surface is exposed to the atmosphere and aggradation of permafrost within the talik (unfrozen ground below or within permafrost environment) occurs (Mackay, 1973; French, 2007; Mackay and Burn, 2002). The succession of vegetation over the years post drainage influences microclimate and soil characteristics such as organic matter and nutrient content and soil thermal and moisture characteristics. All of these factors may impact C cycle process including the rates of CO₂ and CH₄ emission to the atmosphere. A number of studies have investigated C cycling from drained lakes basin in the Arctic and have found that following drainage, productivity increases during the first few decades and eventually diminishes with time (Zona et al., 2010), and that organic layer thickness and the degree of decomposition of soil organic C tends to increase with time (Hinkel et al., 2003).

Nitrogen cycling is another important factor influencing C cycles. When there is an abundance of nitrogen, nitrate may be available to both plants and microbes and nitrous oxide (N₂O) may be produced through anaerobic decomposition. Thus, N₂O emissions are highest under saturated conditions and increase with available soil ammonium (Siciliano et al., 2009). However, this potent greenhouse gas is rarely emitted to any great extent from Arctic soils, even saturated ones (Chapin, 1996). For example, Ma et al. (2007) observed N₂O emissions between ~ 0.0022 to 0.0169 mg N₂O m⁻²day⁻¹ from a High Arctic lowland located on Devon Island, Nunavut. In contrast, Elberling et al. (2010) found unusually high rates of N₂O emission from wetland soils below the active layer after thawing and rewetting these permafrost soils in the laboratory, an equivalent of 34±3.2 mg N₂O m⁻²day⁻¹, while little N₂O was produced from samples collected within the active layer.

1.3 Research Objectives

Drained lake basins are common features near the western Canadian Arctic coast with one or two thermokarst lakes draining either completely or partially every year over the last few decades (Mackay and Burn, 2002). The study site, “Illisarvik” is located on Richards Island, NT. Illisarvik was drained for research purposes in 1978 (Mackay, 1982). Since then, the site has been monitored to study geomorphic processes associated with permafrost aggradation. The site has a high spatial variability in terms of vegetation, soil hydrology and active layer depths. These factors are expected to influence nutrient cycling and the amount of C stored below ground and released to the atmosphere through soil and plant respiration (Davidson and Janssens, 2006; Adams et al., 1990).

The goals of this research are to examine greenhouse gas (GHG) emissions from Illisarvik in order to i) quantify emission rates, ii) identify the factors that influence spatial and temporal variations in these fluxes, and iii) compare these fluxes with those in other Arctic ecosystems. Measurements of vegetation, soil characteristics, and CO₂, CH₄ and N₂O fluxes within the Illisarvik drained lake basin are used to address the following research hypotheses:

- 1) Spatial variations in CO₂, CH₄, and N₂O emissions are correlated with vegetation abundance, soil organic C content, air and soil temperature, thaw depth, and soil moisture.
- 2) CO₂ emissions largely reflect cycling of fresh (i.e., modern) plant residues.
- 3) Compared to other research sites in the Arctic, CO₂ and CH₄ emissions at Illisarvik are similar in magnitude but have greater spatial variability.

1.4 Thesis structure

This thesis is comprised of six chapters. Chapter 2 presents a review of our current understanding of C storage and cycling within Arctic ecosystems. Chapter 3 describes the study area and methods. Chapter 4 presents an analysis of the data collected during the 2016 field season at Ilisarvik. Chapter 5 is a discussion of the study findings and Chapter 6 provides a summary of the research and presents conclusions and suggestions for further research.

2 Chapter: Background

2.1 Introduction

An assessment of the current and future terrestrial Arctic C budget requires an understanding of (1) the unique characteristics of Arctic environments that influence C stocks and flows including thermokarst lakes and (2) the processes of C sequestration and C release from these ecosystems. It is also important to consider the methods which are used to quantify C stocks and C fluxes and how these influence our assessment of the Arctic C budget.

2.2 Arctic environments

The Canadian Arctic landscape extends from 55° N along Hudson Bay to 83° N at the northern part of Ellesmere Island (Barbour and Billings, 2000; Bliss, 1997). At these high latitudes, incoming radiation is low with an annual mean of 8.6 to 9.5 MJ m⁻² day⁻¹ (Ecosystem Classification Group, 2012). The Arctic is also characterized by low temperature and low precipitation (Hugelius et al., 2014). This cold and arid climate is well suited for tundra vegetation which can tolerate and survive through a short growing season and frozen or water-logged soils (Stow et al., 2004; Walker et al., 2005; Spadavecchia et al., 2008). Permafrost in the soils, ground at or below 0°C for two or more years, controls the volume of soil available for vegetation root growth, nutrient and moisture uptake during the growing season. Tundra vegetation includes tree species at the southern limits and in protected areas, shrubs, forbs and graminoids with a large variety of non-vascular moss and lichen species including those making up the cryptogamic crusts, the main vegetation found in the harshest climates (Walker et al., 2005). The July 10°C isotherm corresponds to the northern limit of spruce distribution (Halliday and Brown, 1943). Shrubs tend to favor a soil rich in nutrients where competition is high, while sedges tend to favor saturated soils (Spadavecchia et al., 2008).

Arctic vegetation traps snow during winter to depths varying from a few cm to over a meter deep, depending on vegetation height, density, slope and wind exposure (Mackay and Burn, 2002; Spadavecchia et al., 2008). A deep layer of snow protects vegetation from low air temperatures and wind damage during winter (Spadavecchia et al., 2008) while insulating the ground and limiting soil cooling (Sturm et al., 2001).

The Arctic can be divided into two regions, High and Low Arctic, according to Bliss (1997). The High Arctic possess a growing season length of 1.5 – 2.5 months (Bliss, 1997) and can be classified into dry tundra, mesic tundra and wet tundra (Welker et al., 2004). Dry tundra is characterized by a vegetation cover of less than 50%. Mesic tundra is characterized by higher soil moisture content during the growing season which allows for greater vegetation cover. Wet tundra is dominated by graminoids and other hydrophytic plants which can tolerate saturated soils during the growing season (Welker et al., 2004). The Low Arctic, mostly located south of 70° N has a growing season lasting 3-4 months (Bliss, 1997) and can be classified into a variety of tundra types including erect dwarf shrub and low shrub tundra (Walker et al., 2005). The erect dwarf shrub subdivision is composed largely of non-tussock sedges, dwarf shrubs (10-40 cm tall), moss and wetlands, with vegetation cover between 50-80%. The low shrub subdivision is composed of a surface moss layer, herbaceous, dwarf shrubs (20-50 cm tall) and low shrubs (< 80 cm tall) (Walker et al., 2005). Shrub expansion in the Arctic as a result of climate warming has been investigated by Sturm et al. (2001). There are many potential impacts of this vegetation change on ecosystems and soil-plant-atmosphere/climate interactions driven in part by changes in surface shading and snow trapping, for example (Sturm et al., 2001).

The coastlands of the western Canadian Arctic were covered by the Laurentide ice sheet during the late Wisconsinan between ~22 000 and 16 000 years BP (Murton et al., 2007), which

caused high pressure at the soil surface and permafrost degradation. During deglaciation, meltwater from the ice sheet infiltrated into the unfrozen ground and accumulated while permafrost aggradation occurred, causing ice rich permafrost in some areas (Rampton, 1988).

Topography influences the distribution of water resulting in drier uplands and wetter lowlands throughout the region. Depth of thaw, soil bulk density, texture, and organic matter content also influence the amount of water that can be stored in the soil column. The characteristics of Arctic soils are a result of parent material, topography, vegetation and a long period of cold climate (Bliss, 1997). Normally Arctic soils have low nutrient content and a pH of 5-6.5 in the Low Arctic and 6-8 in the High Arctic depending on parent material and soil moisture status (Bliss, 1997).

At landscape scale, permafrost distribution is controlled by air temperature to some extent and the Arctic includes regions with continuous, discontinuous, and sporadic permafrost coverage (French, 2007). Permafrost underlies about 50% of Canada with continuous permafrost occurring mainly in Nunavut, the Northwest Territories, and Yukon (regions where 80% or more of the area has permafrost) (Tarnocai et al., 2009). Permafrost is mainly controlled both by latitude and altitude, however on a local scale multiple factors can affect permafrost distribution such as: relief, aspect, soil properties, vegetation, snow cover, and water bodies (French, 2007).

Permafrost-affected soils have an active layer that thaws and freezes every year (French, 2007). During the growing season, the active layer thaws and if the soil is ice rich, the ice melt can enhance thermokarst processes. Soil subsidence is noticeable when excess ice melts upon thermal disturbance causing the surface to sink and causing depressions depending on the topography (Mackay, 1970).

2.2.1 Drained lakes

Thermokarst lakes cover about 30% of Tuktoyaktuk Peninsula and between 15 and 30% of the northern Mackenzie Delta (Rampton, 1988). In northern Alaska, thermokarst lakes and drained lake basins cover 46% of the Arctic Coastal Plain, an area which represents 34 570 km² (Hinkel et al., 2007). Rates of drainage of these lakes are 2.2 to 2.3 lakes per year in northern Alaska (Jones et al. 2011). Multiple studies have investigated thaw lake dynamics (Mackay, 1988; Marsh et al., 2009), vegetation within drained thaw lakes (Billings and Peterson, 1980; Oviden, 1986; Lantz, 2017), their soil properties (Mackay and Burn, 2002; O’Neil and Burn, 2012), and their current and possible future C budget (Hinkel et al., 2003; Bockheim et al., 2004; Zona et al., 2010). Other studies have monitored their extent and coverage over time (Plug et al., 2008; Jones et al., 2011; Lantz and Turner, 2015).

Thermokarst lakes are formed by a disturbance of the thermal equilibrium of the ground which results in thawing of ice-rich permafrost or melting of massive ice (Marsh et al., 2009; Jones et al., 2011). These Arctic features were very active during the Holocene and are a sign of local permafrost degradation under a warming climate (Rampton, 1988). According to Jones et al. (2011), partial drainage of larger lakes which are affected by shoreline erosional processes would increase runoff water based on topography and create or enlarge other thaw lakes downhill. Drainage of these lakes may occur through interconnecting ice-wedges and can be catastrophic, with lakes completely draining in less than 24 hours (Mackay, 1988; Marsh et al., 2009). This drainage via ice-wedge degradation can be triggered by natural processes such as rapid snowmelt in the spring (Mackay, 1988).

Thaw lakes need to commonly have a water depth greater than 1.5-2 m in order to stay unfrozen at the lake bottom all year round (Mackay and Burn, 2002). In such cases, a talik forms beneath the lake (French, 2007).

For a few years following drainage, the bottom of a drained lake is covered with either sandy peat or organic lake mud which are suitable for primary succession (Ovenden, 1986). With time, changes in surface conditions may occur, including vegetation succession, with a range of plant species colonizing the lakebed. Some species are more tolerant of saturated soils, such as *Carex aquatilis*, *Senecio congestus* and *Arctofila fulva* while other species are tolerant of drier conditions e.g., *Betula glandulosa*, *Alnus* spp and *Salix* spp (Billings and Peterson, 1980; Ovenden, 1986; MacDonald and Gajewski, 1992). Billings and Peterson (1980) found that within 4 years following drainage of two thaw lakes near the Arctic coast of Alaska, a well-developed network of flat top polygons covered with moss and scattered vascular plants formed. Sedges and grasses with both seeds and rhizome reproductive methods colonized the area faster than species with only one method. 30 or more years following drainage, the authors noted that the basins were covered with tundra graminoid vegetation, primarily *Dupontia fisheri*. Drained lake basins that have been studied in the Canadian Arctic or in Alaska typically become a sedge-moss peatland over time (Ovenden, 1986).

The common consensus is that soil organic C accumulates with time in the upper soil layers of drained lakes (Hinkel et al., 2003; Bockheim et al., 2004). Zona et al. (2010) found that ecosystem respiration tends to decrease with time since drainage on the Arctic coastal plain of northern Alaska; older sites (300- 2000 years BP) respired 4 g CO₂ m⁻² day⁻¹ in contrast to 5.2 g CO₂ m⁻² day⁻¹ for medium age sites (50-300 years BP) and 7.3 g CO₂ m⁻² day⁻¹ for young sites (< 50 years BP). Gross primary productivity also tends to decrease over time with younger sites

being more productive compared to medium and older sites. Thus, the net CO₂ exchange did not change much over time and there were no significant differences between medium aged and older sites.

2.3 Arctic C storage and fluxes

The accumulation of soil organic C stocks is a result of long-term C inputs that exceed C outputs. Initially, organic C accumulates in vegetation by photosynthesis. With senescence, plant material enters the soil and microbial decomposition results in loss of some portion of this organic C from the soil through gaseous fluxes (CO₂ and/or CH₄) or as aquatic C fluxes. CO₂ efflux from the soil surface is often referred to as “soil respiration” (Gaudinski et al., 2000) and is the result of respiration by roots (autotrophic respiration), microorganisms (heterotrophic respiration), and chemical oxidation of carbon-containing materials (Raich and Schlesinger, 1992). The mean rate of global soil respiration of tundra ecosystems is 60 g C m⁻² yr⁻¹ (or 220 g CO₂ m⁻² yr⁻¹) (Raich and Schlesinger, 1992). Spatial and temporal variations in these respiration processes through the Arctic and around the world are controlled by plant productivity, litter inputs, air and soil temperature, and precipitation (Raich and Nadelhoffer, 1989; Raich and Schlesinger, 1992).

Raich and Schlesinger (1992) found a positive relationship between net productivity and soil respiration ($r^2 = 0.87$) which can be expected since photosynthesis ultimately provides the substrate for respiration (Gifford, 2003). However, the turnover time of soil organic C is not directly linked to CO₂ efflux since there is transport and storage/protection of C within the soil profile (Trumbore, 1993). According to Raich and Schlesinger (1992) the soil C turnover time of tundra environments is 490 years, compared to 91 years in boreal forest or 10 years in tropical grasslands. This mean residence time was calculated by comparing the ratio of ¹⁴C/¹²C of newly

added vascular plant material to the ratio of $^{14}\text{C}/^{12}\text{C}$ in organic matter derived from vascular plants (Trumbore, 1993).

Fresh organic matter is input into the soil from above and through the root zone as plant detritus (Trumbore, 1993; Schlesinger and Andrews, 2000). In Arctic soils, cryoturbation is an important process that moves organic C downward or outward depending on topography. In summer this organic C reaches the bottom of the active layer where it can be incorporated into permafrost, 'protected' from decomposition, and accumulate. As freeze back occurs, this C freezes and cannot be decomposed by microorganisms. Bockheim (2007) explains that cryoturbation is a physical process that moves the soil as a result of frost action. Mackay (1980) proposed a theory of cryoturbation commonly called "cell circulation". Soils that promote cryoturbation are frost susceptible and silty with poor drainage (Bockheim, 2007). Within the top 0-3 m, Turbels (i.e. Turbic Cryosols) possess 42% of the total C in the northern circumpolar region (Kuhry et al., 2013). Within this type of soil, 70-86% of the total C pool is buried in the lower active layer, above the permafrost table (Kuhry et al., 2013). Orthels (i.e. Static Cryosols) contain less C because there is less cryoturbation with soil C content decreasing with depth (Kuhry et al., 2013).

In addition to cryoturbation, peat accumulation and sedimentation are two other means to bury organic C in permafrost-affected soils. Peat is formed of partially decomposed plants that accumulate where the rate of decomposition is less than the rate of net primary production (St-Hilaire et al., 2010). Low hydraulic conductivity and poor drainage may lead to anaerobic conditions which limit microbial decomposition resulting in a long-term sink of C (Moore and Dalva, 1997; St-Hilaire et al., 2010).

Abiotic factors can affect respiration rates and turnover times such as temperature, moisture and nutrient availability (Bond-Lamberty et al., 2004; Davidson and Janssens, 2006).

Decomposition is reduced when microbial activity is limited by low temperatures, by high or low water content, and through a variety of physical protection mechanisms, such as organic matter becoming trapped in soil aggregates where microorganisms have restricted access. Decomposition also tends to proceed slowly on more recalcitrant organic matter which is in a frozen state for thousands of years (Schuur et al., 2009). Temperature largely determines decomposition rates through changes in kinetic energy and the biogeochemical reactions that occur with decomposition (Davidson and Janssens, 2006). In very dry soils, a reduced amount of water limits diffusion of enzymes and soluble organic carbon in the soil column thereby reducing microbial activity and decomposition. However, in very wet soils, oxygen diffusion is slowed and the activity of microbes with aerobic metabolism is limited (Davidson and Janssens, 2006). Anaerobic environments (saturated soils) may be areas of CH₄ production by methanogens while methanotrophs oxidize CH₄ into CO₂ in aerobic environments. Multiple factors can affect methanotroph and methanogen activity such as nutrient availability, soil texture, soil temperature, pH and substrate availability (Le Mer and Roger, 2001).

An environment becomes a source of CH₄ when production exceeds consumption, otherwise the environment may be a small sink for atmospheric CH₄. Methane is released to the atmosphere through diffusion, plant mediated transport, or as bubbles escaping from below the water table in wetlands (Moore and Dalva, 1997; Le Mer and Roger, 2001). A study by Verville et al. (1998) examined the effect of vegetation on CH₄ efflux by removing sedges from plots in Alaska. Their results showed sedge removal decreased the emissions of CH₄ by 60%. Sedges have aerenchymatous tissues which allow gas transport for the aeration of roots below the water table. Without these tissues, CH₄ diffusing from below the water table to the atmosphere may become

oxidized within upper aerobic soil layers producing CO₂ and reducing total CH₄ emissions (Verville et al. 1998).

Another potent greenhouse gas, nitrous oxide (N₂O) may be emitted from soils through anaerobic decomposition and denitrification of nitrate where nitrate acts as an alternative electron acceptor when oxygen is depleted (Ma et al., 2007). However, in most Arctic soils, nitrogen is limiting to both plants and microbes (Chapin, 1996). As a result, nitrate is not available to denitrifying bacteria, limiting their activity and emissions of N₂O.

Multiple studies have examined CO₂ and CH₄ emissions and their controls in Arctic regions (Fan et al., 1992; Vourlitis and Oechel, 1999; Oberbauer et al., 2007; Merbold et al., 2009). Deeper thaw is expected to correlate positively with CO₂ emissions since there would be more substrate available for micro-organisms (Zona et al., 2010; Harden et al., 2012; Belshe et al., 2013; Kuhry et al., 2013; Schuur et al., 2015). Soil moisture is found to be an important control on CH₄ emissions, with higher emissions at wet or saturated sites such as wet meadow tundra, thaw ponds and polygonal terrain (Fan et al., 1992; Bartlett et al., 1992; Vourlitis et al., 1993; Merbold et al., 2009; Laurion et al., 2010; Martin et al., 2017). In contrast, CO₂ emissions were found to be lower at wet or saturated sites compared to drier tundra sites (Fan et al., 1992; Vourlitis and Oechel, 1999; Oberbauer et al., 2007; Merbold et al., 2009).

Belshe et al. (2013) conducted a meta-analysis of 54 C flux studies across 32 Arctic sites to examine the temporal trend of emission and uptake in tundra environments and found that there may be a shift in tundra C budgets from C sink to source in the future with higher temperatures, since despite an increase in CO₂ uptake by plants in the growing season, greater CO₂ efflux in the winter exceeds that uptake.

2.4 Carbon flux measurement methods

Two methods often used to measure CO₂ and CH₄ emissions in-situ are chamber and eddy-covariance techniques. These two possess different advantages and disadvantages. Eddy covariance is a technique that can measure CO₂ or CH₄ fluxes of an upwind area of ~ 10 ha as the covariance between CO₂ or CH₄ mixing ratio and vertical wind velocity (Baldocchi, 2003). This method is mostly effective on homogeneous landscapes. This technique is used on a long-term basis at sites around the world (Baldocchi, 2003). This instrumentation and power requirements for this technique are expensive. In addition, only Net Ecosystem Exchange (NEE) is measured and component fluxes, Gross Primary Production (GPP) and Ecosystem Respiration (ER), have to be estimated using a variety of techniques (Moffat et al., 2007).

Chamber flux techniques are used on a much smaller scale (around 1 m² or less) to characterize fluxes of different soil or vegetation characteristics within an ecosystem. Enclosure systems can capture any net emission, net uptake or change in concentration within that restricted sealed volume (Livingston and Hutchinson, 1995). Methods include steady state and non-steady state with static or flow through systems. Steady state flow through systems involves regulation of air with known gas concentrations to maintain rates of molecular diffusion at the soil-atmosphere interface (Livingston and Hutchinson, 1995) and require an in-situ gas analyzer (e.g. Wilson and Humphreys, 2010; Lai et al., 2012) while non-steady state static systems use vials to collect gas samples at intervals and analyse them with a gas analyzer usually off site (e.g. Crill et al., 1988; Bubier et al., 1995). This technique is cost effective since the materials and supplies are less expensive. All chamber techniques also have the advantage of being capable of measuring NEE and ER with the use of dark chambers which forces GPP to zero. When manual chamber measurements are made, human errors are possible, while disturbance to the plots is possible which

can impact soils and vegetation within and surrounding the chambers (by trampling, etc). Enclosure of the vegetation and soil will alter environmental conditions within the chamber. Also, for the vegetation to fit within the collar, cutting roots or stems during installation might be necessary. This might impact plant and soil C cycle processes and gas transport processes (Livingston and Hutchinson, 1995). Since collars are small and represent a certain vegetation type or ecosystem, a stratified sampling method is recommended in order to capture heterogeneous terrain. For global scale representation, these chamber measurements need to be extrapolated by factors between $10^{12} - 10^{14}$ making it difficult to apply these estimates globally (Livingston and Hutchinson, 1995). Also, since these non-steady state enclosure systems require manual sampling, temporal variability is an important factor to take into consideration. For example, it is important to sample on different days and at different times of the day to capture the diurnal and daily variability and avoid bias in the measurements (Livingston and Hutchinson, 1995).

3 Chapter: Study Area and Methods

This thesis examines the C emissions released by respiration processes and the temporal and spatial variability in these fluxes as they relate to different vegetation communities, soil moisture, temperature, organic matter content, available nutrients, and pH. Collection of field data took place at the Illisarvik drained lake basin, Richards Island, NT. This chapter describes the study area, the research site, the methods used for the field data collection, the laboratory analyses and data analyses.

3.1 Study area regional characteristics

The study area is located in the Tundra Plains Low Arctic north (LAn) ecoregion, which is part of the Southern Arctic ecozone (Figure 3.1) (Ecosystem Classification Group, 2012). The climate of this ecoregion is influenced by latitude, the proximity of the Beaufort Sea and vegetation. The proximity of the sea causes typically low clouds and fog induced by low pressure systems in the region. The average annual temperature of this ecoregion is between -12 and -9 °C with average annual precipitation of up to 200 mm (Ecosystem Classification Group, 2012). The period of sea ice cover in the Beaufort Sea extends from October until June (Hill et al., 1990).

The study area is north of the treeline and is within the Tununuk Low Hills physiographic subdivision, which describes the western part of Richards Island and is characterized by lakes of irregular size and poorly drained depressions (Rampton, 1988). This subdivision is underlain by material that varies from fine-grained till to glaciofluvial gravel (Burn, 2002). The landscape is relatively flat but irregular with maximum elevation at around 15 m (Mackay, 1963). The area is underlain by continuous permafrost to a depth of 600 m. The area was covered by ice during the last glaciation of Richards Island (22 000 to 16 000 years BP) (Rampton, 1988). The tundra

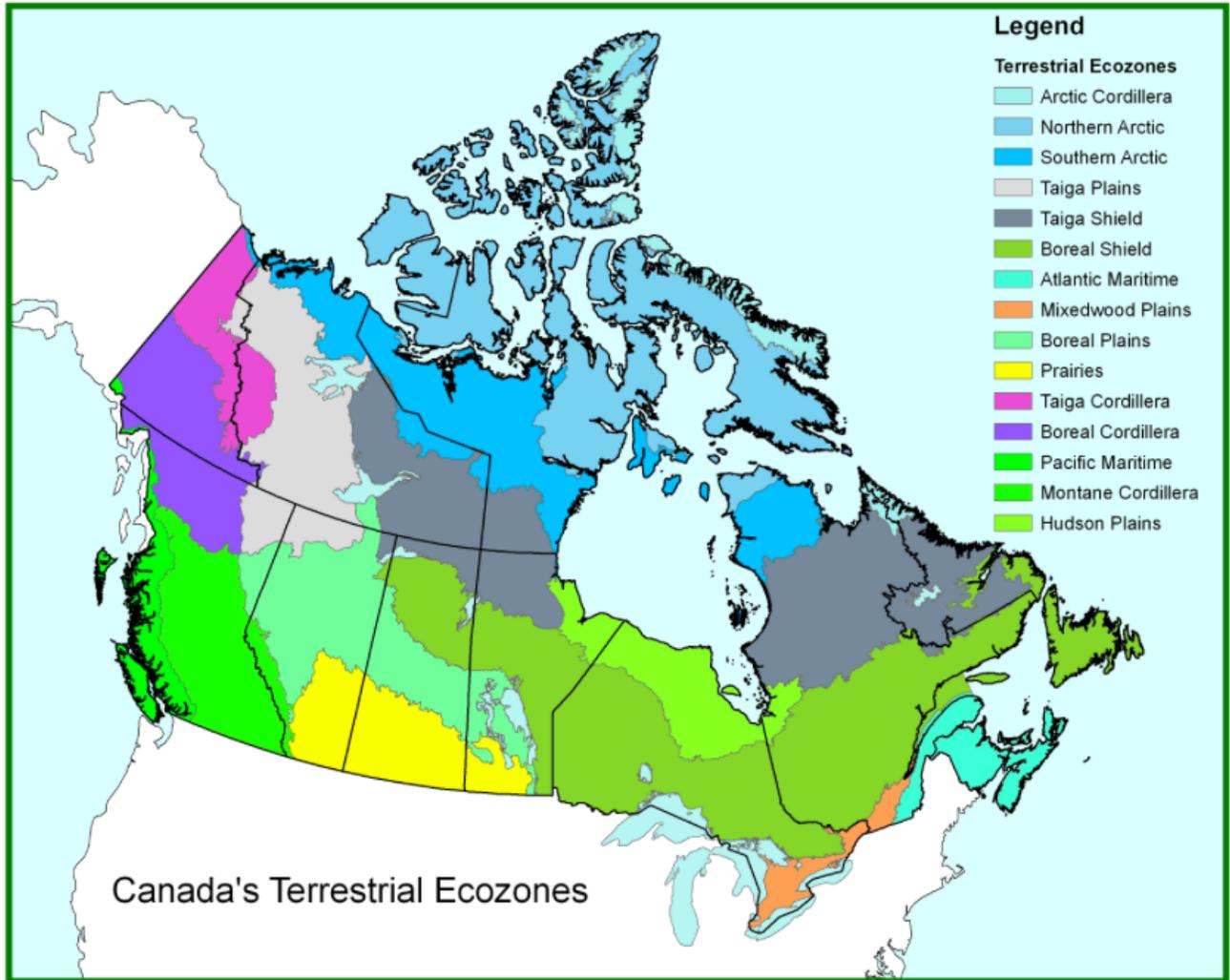


Figure 3.1. Canada’s Terrestrial Ecozones (Natural Resources Canada, 2017).

vegetation in the region includes dwarf shrubs; willows (*Salix* spp.), and birch (*Betula nana*), along with graminoid, sphagnum and brown mosses and lichens and sedges, usually (*Eriophorum scheuchzeri*) and (*Carex* spp.) in poorly drained areas (Mackay, 1963; Ecosystem Classification Group, 2012).

3.2 The Illisarvik drained lake basin

3.2.1 History of the site

The field study site is located at 69°28'5" N, 134°35'04" W (Figure 3.2). Dr. J.R. Mackay drained the Illisarvik thermokarst lake for research purposes by excavating an ice wedge on August 13th, 1978. Prior to drainage, the lake was around 2-3 m deep and underlain by a 32 m deep, bowl-shaped talik (Mackay, 1997). The initial purpose was to study permafrost growth post-drainage. Since then, a number of studies have been carried out on permafrost dynamics of the region (Burn and Kokelj, 2009), vegetation succession (Ovenden, 1986), active layer development (Mackay and Burn, 2002), segregated ice formation (O'Neill and Burn, 2012) and the history of the lake basin (Michel et al., 1989).

The lake formed by 9500 years BP then reached its maximum size at 6000 years BP to finally diminish to modern size around 2000 years BP (Michel et al., 1989). Following drainage, the lakebed sediment was exposed and permafrost aggradation began the following winter (Mackay and Burn, 2002). Two residual ponds remained. According to Ovenden (1986), by 1981, relatively high wind erosion in the central part of Illisarvik lakebed, resulted in a deposition of sediment (20-30 cm) along the basin margin. Vegetation succession occurred and is ongoing with predominately sedges (*Carex aquatilis*) around ponds and more diverse vegetation near the edges

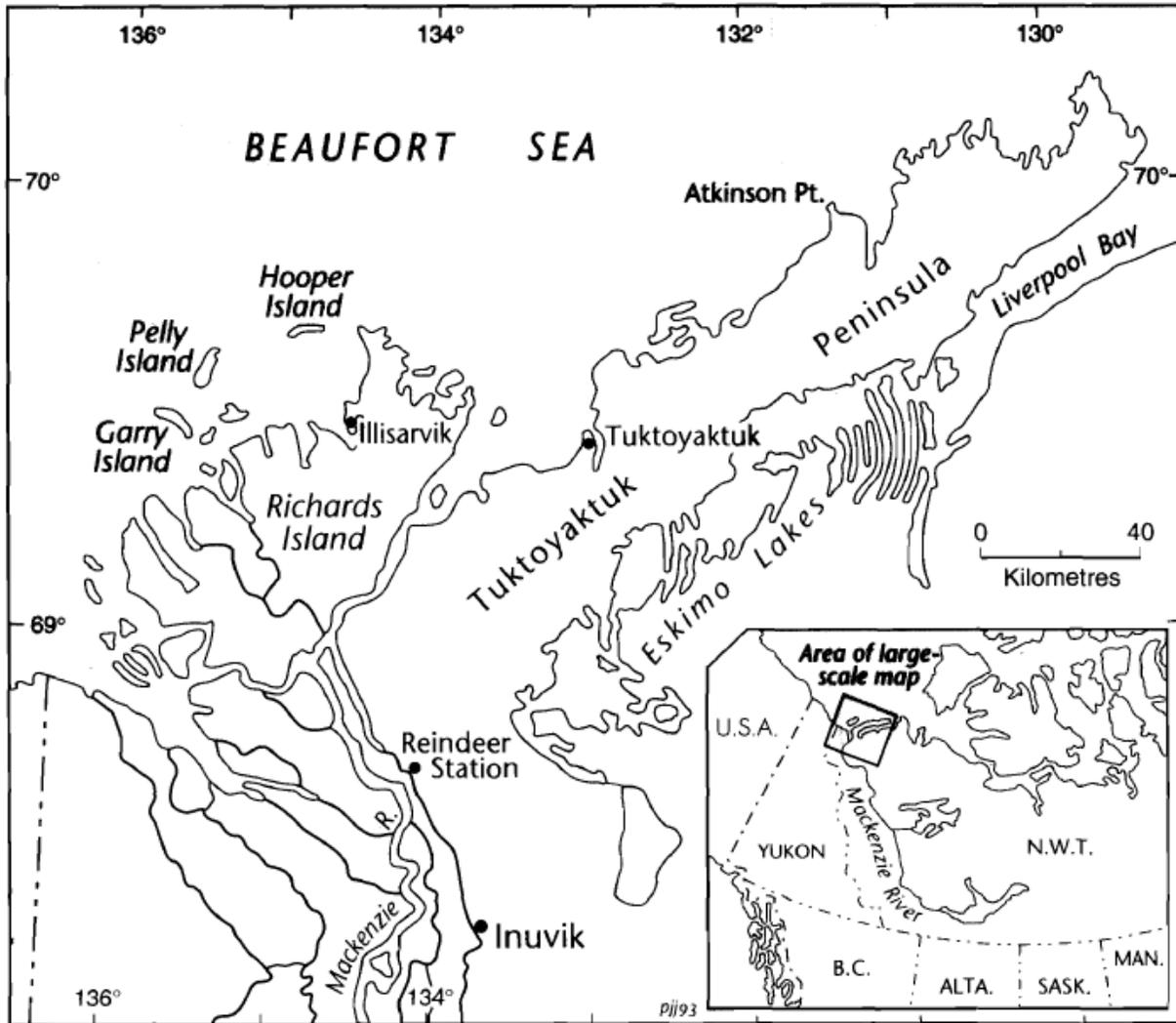


Figure 3.2 Location of Illisarvik on Richards Island, western Arctic coast (Figure 1 from ©Mackay, 1997. By permission from publisher).

of the basin varying from grasses (*Poa spp.*) to willows (*Salix Lanata*) (Ovenden, 1986). The Illisarvik basin is expected to become a wet sedge-moss peatland, if it follows the succession documented at other naturally drained thermokarst lake basins in the area (Ovenden, 1986).

3.2.2 Site characteristics

The drained lake basin is approximately 18 ha in area (600 m × 300 m) and approximately 50 m from the Arctic Ocean. The nearest community is Tuktoyaktuk, located approximately 60 km east where there is an Environment Canada weather station. During the 2016 field season (July-August), the average air temperature was 10.8°C and there was negligible precipitation (Environment Canada, 2017). At Illisarvik, average air temperature was 9.9°C, maximum daily temperature reached 26.3°C while minimum temperatures did not drop below 0.6°C (Figure 3.3). The 1971-2000 annual temperature normal for Tuktoyaktuk is -10.2 °C and annual precipitation is 139.3 mm (Environment Canada, 2017).

The vegetation within the Illisarvik drained lake basin is highly variable (Figure 3.4). The tundra vegetation outside the lake basin contrasts with the vegetation within the lake basin, which varies from bare ground to dense willows over 2 m tall (Figure 3.4). Outside the basin, the vegetation is typical of erect dwarf shrub tundra (Welker et al., 2005). Vegetation units were delineated by visual assessment of species type and soil conditions (saturated, dry, mineral, organic) and named based on the dominant species (see also Wilson, 2018).

3.3 Study design

Topography, drainage, and soil moisture are in part responsible for driving the spatial variability of the different vegetation units (Ovenden, 1986). Within the basin, ten sites were selected to best capture the vegetation and soil moisture variation within the study area (Figure

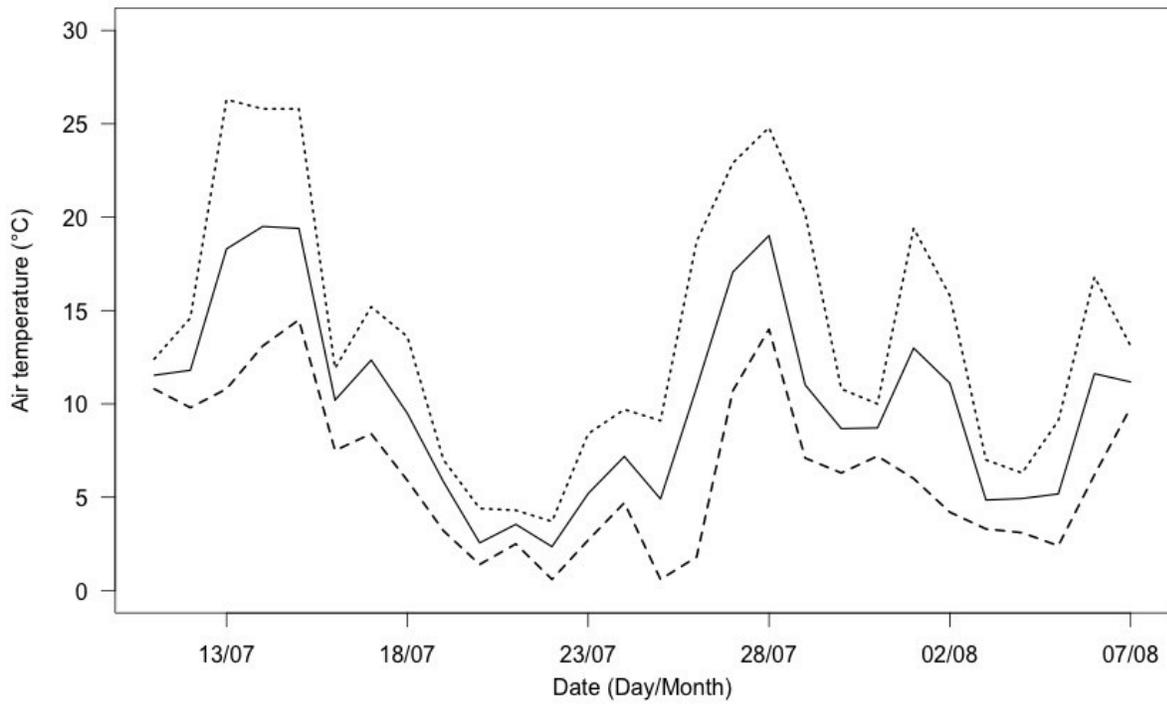


Figure 3.3 Illisarvik daily mean air temperature (solid line), maximum daily air temperature (dotted line) and minimum daily air temperature (dashed line). Air temperature measurements were made at a meteorological station on the southwest side of the drained lake basin.

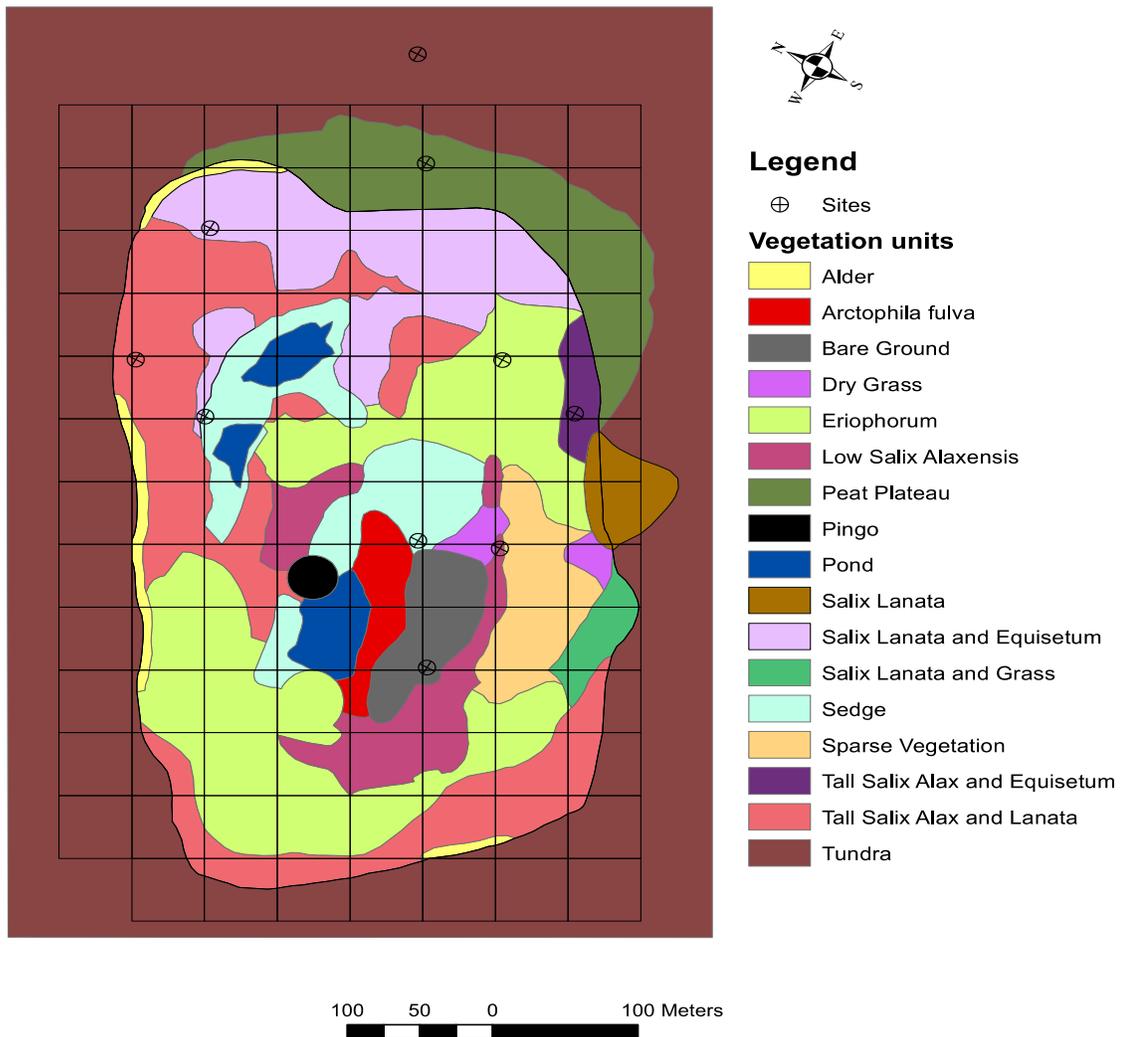


Figure 3.4 Vegetation units within the Ilisarvik drained lake basin, modified from Wilson (2018). The 10 study plots are shown with crossed circles.

3.4). Due to logistical limitations, plots were not replicated within vegetation units except at the Wet Sedge site.

Among these ten sites, eight were located in the basin, one on the peat plateau and one on the surrounding tundra (Figure 3.4). The different vegetation communities were named for the dominant species present or using a general description. The “Wet Sedge” (Figure 3.5) community was represented at two sites; Wet Sedge 1 and Wet Sedge 2. For the purpose of analysis, five vegetation classifications were defined. The first was Tundra, followed by Peat Plateau. The third one was “Willows” which includes Dense Willow, Low Willow and Tall Willow. The fourth was “Wet Sedges” which includes the Wet Sedge 1 and 2 and Cotton Grass. The fifth was Grass and the last one was the Bare Ground.

3.4 Measurements

3.4.1 Flux measurements

Gas flux measurements were made on 11 dates between July 12 and August 5, 2016 using a static non-steady state chamber flux technique. At each site, a pair of collars was installed to capture the difference between below ground and total ecosystem respiration. First, plant functional group presence was assessed at each collar location and then the above ground vegetation was removed in one of the collars (see below for more details). Only the bare ground site had one collar since there was no vegetation present. The collars were installed at the beginning of the field season. The PVC collars had a height of 30 cm and a diameter of 24.3 cm and inserted to a depth of roughly 15 cm at each site. The distance between the collars was approximately 20 cm for each site. The chambers were 34 cm tall and made out of polycarbonate covered in black opaque tape to maintain dark conditions inside the chamber (Martin, 2015) (Figure 3.6). The

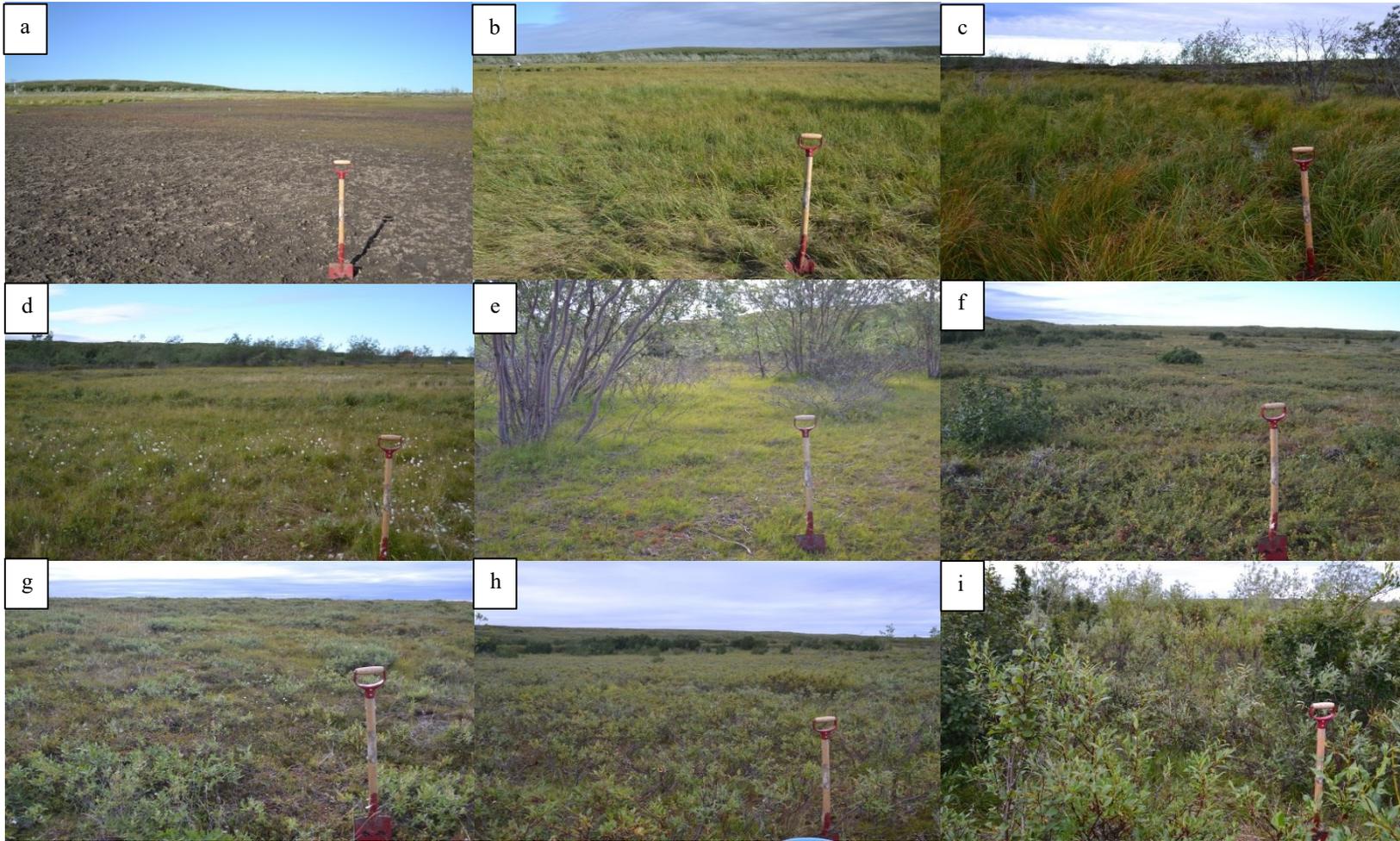


Figure 3.5 Different vegetation communities within the study site; (a) Bare Ground, (b) Grass, (c) Wet Sedges, (d) Cotton Grass, (e) Tall Willow, (f) Peat Plateau, (g) Tundra, (h) Low Willow and (i) Dense Willow. Wet Sedge 1 (c) represents both Wet Sedge sites since they are visually similar. Shovel is 1 m tall.



Figure 3.6 Top image: an example of a gas flux measurement. Bottom image: collars with (left) and without vegetation (right) at the Tundra site.

chambers contained a small vent (10 cm coiled $\frac{1}{8}$ diameter copper pipe) to ensure a constant pressure during measurements.

During gas flux measurements, the chambers were sealed to the collars with water and five air samples were collected into evacuated 12 cc vials sealed with doubled septa to ensure an air-tight seal. Each vial contained 2 mL of magnesium perchlorate to dry the air sample and 3 vials per box of 100 vials were filled with He before transport to the field in order to monitor the integrity of the vials through shipping, storage and analysis. Samples were collected at time 0, 5, 10, 15 and 20 minutes after the chambers were set on the collars. Air within the chamber was mixed with a 60 cc syringe attached to a three way stopcock, then air samples were collected with a 30 mL syringe and 24 mL was stored in the pre-evacuated vials as described by Wilson and Humphreys (2010). Measurements were made every two days during the field season between 9 am and 5 pm starting at a new site each day to randomize the sampling order to avoid bias. Calculation of CO_2 ($\mu\text{mol m}^{-2} \text{s}^{-1}$), CH_4 ($\text{nmol m}^{-2} \text{s}^{-1}$), and N_2O fluxes ($\text{nmol m}^{-2} \text{s}^{-1}$) (F) were carried out using the following generic equation:

$$(1) \quad F = V/A \times P/RT \times dx/dt$$

where the linear rate of change in the mixing ratio of the gas (dx/dt) is converted to the appropriate flux units using the chamber area (A, 0.0464 m^2), the chamber volume (V, between 0.0182 and 0.0242 m^3 adjusted for collar depth at each collar location) and the ideal gas constant (R, $8.314 \text{ J mol}^{-1} \text{ K}^{-1}$). Barometric pressure (P, Pa), air temperature (T, Kelvin) were measured using a LI-7700 open path CH_4 Analyzer (LI-COR Inc., Lincoln, NE, USA) and a HMP35C Temperature and RH sensor (Campbell Scientific, UT, USA), respectively at a nearby micrometeorological

tower. This tower was established as part of another study where the eddy covariance technique was used to assess net ecosystem exchange of CO₂ for the whole basin. Air temperature and pressure were measured using an HMP35C Temperature and RH sensor (Campbell Scientific, Logan UT, USA) and a pressure transducer on the LI-7700 Open-path CH₄ Analyzer (LI-COR Inc., Lincoln, NE, USA), respectively. Positive fluxes indicate emissions of gases to the atmosphere and negative fluxes indicate uptake by the surface.

Concentrations of CO₂, CH₄ and N₂O were determined at Carleton University, on a gas chromatograph (model CP 3800, Varian Inc., Pao Alto, CA, USA) using a He carrier gas at 30 mL min⁻¹ and equipped with a flame-ionization detector at 300°C and a methanizer at 350°C (Wilson and Humphreys, 2010). A Haysep N 80/100 pre-column (0.32 cm diam.x 50 cm length) and Poropak Qs 80/100 mesh analytical columns (0.32 cm diam. X 200 cm length) in a column maintained at 50°C was used for gas separation (Wilson and Humphreys, 2010). Samples were calibrated using three replicates of five standards varying from 383.1 to 15212.6 ppm CO₂, from 1.08 to 22.11 ppm CH₄ and 0.32 to 1.00 ppm N₂O. To maintain quality control, He blanks were included in every set of gas measurements to create a linear relationship between gas concentration and chromatogram area.

For CH₄ and CO₂ change over time, coefficients of determination (r^2) ranged from 0.002 to 0.997. After removal of point measurements (if more than 2 points were rejected, the entire flux measurement was rejected), that substantially reduced the linearity of the trend, r^2 ranged from 0.71 to 0.999. Overall, 72 measurements were rejected out of 1135 vials and no flux measurements was rejected out of 681. Only the bare ground site showed evidence of measurable N₂O emissions (evaluated with r^2 values exceeding 0.39).

During each flux measurement, soil temperature was collected with a simple meat thermometer next to each collar and volumetric water content was measured with a moisture probe (Hydrosense II, Campbell Scientific) in between the pair of collars at each site.

3.4.2 Soil Characteristics

A pit was dug at each site to characterize soil characteristics including C stocks. The depth of the soil profile depended on the position of the water or permafrost table. Two types of samples were collected at each 5 cm interval; samples with a known volume and “grab” samples without a known volume. One set of samples were made using a core (5 cm in depth and 6.2 cm diameter with a volume of 150.95 cm³). These core samples were used to determine bulk density. Grab samples were collected with a trowel and were analyzed for soil moisture content, organic matter content, %C, %N, and C:N, pH and electrical conductivity. The samples were stored frozen until soil analysis could be carried out.

Bulk density was determined from the mass of the oven-dried sample (at 105°C) to the total volume sample (g cm⁻³). Gravimetric water content was determined as the ratio of water mass to dry soil mass (g water g⁻¹ dry soil). The mass of water was determined as the difference between the mass of the initial sample (weighed in Inuvik) and the mass of the oven-dried sample. Volumetric water content was then calculated as the ratio of water volume to total soil volume by multiplying the gravimetric water content by bulk density and dividing by the density of water (m³ water m⁻³ soil).

The amount of organic matter content (g organic matter g⁻¹ dry soil) was quantified using the loss-on-ignition (LOI) method (Sheldrick, 1984). A small portion of soil (~ 2 g) from every sample depth was analyzed. Samples were dried in a pre-weighed crucible and in the oven at 105°C overnight. The samples were then weighed and heated at 550 °C for four hours to combust

all organic matter. The fraction of organic matter (ash-free) is the difference in these weights divided by the original dry weight. The fraction of carbonates ($\text{g carbonates g}^{-1}$ dry soil) was calculated as the difference in weight after combustion at $950\text{ }^{\circ}\text{C}$ for two hours and at 550°C for four hours. For %C, %N and C:N analysis, soil samples were oven dried at 60°C and analyzed using a LECO TruSpec CN Carbon/Nitrogen Determinator, at Queen's University. Total C on an area basis was calculated by multiplying the fraction of OM and bulk density for each layer and integrating with depth.

The texture of the mineral soil at each site was analyzed using sedimentation following Kalra and Maynard (1991) after treating the samples with hydrogen peroxide to remove organic matter and shaking with sodium hexametaphosphate to disperse aggregates. Oven dried ($105\text{ }^{\circ}\text{C}$) grab samples from a depth of 15-20 cm for each site were selected for this analysis.

pH and electrical conductivity were measured to help understand the chemical properties of the soil that may impact microbial activity, vegetation growth and C fluxes. Samples were pre-treated using water as the suspension medium (Kalra and Maynard, 1991) at a ratio of 1:4 for mineral soils (10 g of soil for 40 ml of H_2O) and 1:8 for organic soils (10 g of soil for 80 ml of H_2O). The pH was then measured with the B30PCI SympHonyTM Benchtop Meter (Carleton University) after calibration with reference pH 4 and pH 7 solutions. The electrical conductivity was determined using the same instrument on the same processed samples but with the conductivity probe. Probe calibration was made using $100\text{ }\mu\text{S}$, $1000\text{ }\mu\text{S}$, and $1413\text{ }\mu\text{S}$ solutions.

Plant Root Simulator (PRS, WesternAg, Saskatoon, SK, Canada) probes were used to estimate plant available anions and cations within the soil solution. A resin adsorbs these ions as water flows around the probes buried in the ground (Harrison and Maynard, 2014). One probe captures anions and the other captures cations so they are installed in pairs. Four pairs of probes

were installed at each site at a depth of 15 cm at the beginning of the field season and were removed at the end for a total burial time of 27 days. The probes were cleaned with distilled water at the Aurora Research Institute, Inuvik, NT according to manufacturer's instructions and sent for analysis to the WesternAg Laboratory, Saskatoon, SK.

3.4.3 Vegetation characteristics

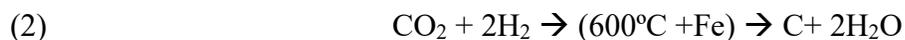
Aboveground vegetation and the LFH (litter, fermented and humic) soil horizon were collected in the collars for comparison within and among sites. The above ground vascular vegetation was collected from the unvegetated collar at the beginning of the season and was air-dried in the field. The different fluxes from the vegetated and unvegetated paired collars allowed for a comparison of respiration originating from the total ecosystem (with plant contributions) and from the full soil profile including any moss or lichens. The LFH horizon was then collected mid-field season in the unvegetated collar and also air dried. This manipulation allowed for a comparison of total ecosystem respiration with respiration from the soil only. At the end of the field season a day after the last flux measurement, the vegetated collars' vegetation and LFH horizon were also collected. At the laboratory, vegetation and LFH samples were oven dried at 60°C and weighed.

Leaf Area Index (LAI2000 Plant Canopy Analyzer, LI-COR) measurements were made on four days during the field season (July 12, 26, 27, and August 3) at each of the ten vegetation unit locations. Measurements are taken above and below canopy to calculate light interception at different angles. These measurements estimate the area of foliage and other plant parts per area of ground ($\text{m}^2 \text{ m}^{-2}$).

3.4.4 Carbon 14

To understand the source of the C released, gas samples were collected for ^{14}C analysis from each collar near the end of the field season. Since more C is needed for this analysis, chambers remained sealed to the collar for at least one hour. A 200 ml evacuated Wheaton glass bottle stoppered with a butyl rubber septa was filled with 300 ml of the evolved gas thereby over-filling the bottles by 100 ml or to a pressure of 1.5 atm. Atmospheric air was also sampled in the morning and at the end of the sampling day. For these samples three 200 ml bottles were filled with 300 ml of ambient air coming into the chamber from approximately 1.5 m above the ground and facing the wind direction.

Gas samples were processed at André Lalonde Accelerator Mass Spectrometer Laboratory, at the University of Ottawa. The samples were processed through the extraction line to extract and isolate CO_2 in Pyrex breakseals through valves cooled with liquid nitrogen. The samples were processed using graphitization methodology, where H_2 reduction is used over an iron catalyst (Vogel et al., 1984; St-jean et al., 2017).



The graphite samples were then pressed into targets (Crann et al., 2016). The targets were loaded in the sample wheel of the accelerator mass spectrometer and were then ionized. With an electrical and a magnetic field, these beams of ions are moved through an argon filled canal with a voltage of 3 000 000 volts to transform these electrons into positive ions. Then, the positive ion beams are projected at a few percent of the speed of light to the Analyzer Magnet that will bend the ion beams to 90° . Since ^{13}C and ^{12}C are lighter than ^{14}C , these ions will bend and be counted in the Vacuum Box for Faraday Cups, while ^{14}C will continue and go through a Particle Detector filled with isobutane (C_4H_{10}). The ^{14}C will collide at high speed with the C_4H_{10} and will create energy.

Then, every time there is a collision it is recorded and compared to the amount of ^{12}C counted. Results include $\Delta^{14}\text{C}$, which is the depletion of $^{14}\text{C}\%$ based on the reference year (1950), where greater negative values represent older samples and less negative values represent modern samples and age BP using a decay model based on Stuiver and Polach (1977) where age is calculated as $-8033 \ln(F^{14}\text{C})$ and where $F^{14}\text{C}$ is the fraction of modern C relative to the reference year (Stenström et al., 2011)

3.5 Data Analyses

The temperature sensitivity of CO_2 emissions was analyzed using Equation (3) describing the change in flux for a temperature change of 10°C (Q_{10}) and a reference flux at 10°C (R_{10}). Equation 3 was fit to the data using the nonlinear least squares method in R (nls, R statistical package).

$$(3) \quad \text{Respiration} = R_{10} Q_{10}^{((T - 10)/10)}$$

where Respiration represents the CO_2 flux expressed in $\mu\text{mol m}^{-2} \text{s}^{-1}$ and T represents air temperature.

CH_4 fluxes were log transformed for presentation. Pearson's correlation coefficient (r) or Spearman's rank correlation (ρ), when data distributions were skewed (even after transformation), were used to examine potential relationships between fluxes and environmental variables for both temporal and spatial scales. The non-parametric Mann Whitney U (W) test was used to compare distributions and Wilcoxon Signed Rank (S) test was used to compare matched pairs. Results were deemed statistically significant when $p < 0.05$.

4 Chapter: Results

This chapter examines spatial and temporal variations in CO₂, CH₄ and N₂O fluxes and the factors that influenced these trends. A complete portrait of each site is presented including soil and vegetation characteristics. An analysis of the age of emitted CO₂ is presented to help characterize C cycling processes.

4.1 Site characteristics

4.1.1 Vegetation

The diversity of vascular plant species found within the collars (Table 4.1) varied among the different vegetation units. Sites around the ponds (see Figure 3.3 for locations of Wet Sedge 1 and 2, Grass and Bare ground plots) had two plant species or less while collars near the edge of the old lakebed and onto the surrounding tundra were more diverse in terms of species number (see Table 4.1).

The difference in vegetation biomass between the two collars at each site was around 3 g m⁻² (Table 4.2). Although vegetation harvest was 26 days apart, this suggests that the vegetation within the collars was near full biomass at the beginning of the measurement period. The Bare Ground site only had one collar, thus only one biomass value (near zero). Leaf Area Index was higher in the Wet Sedge sites with values above 4 m² m⁻² (Table 4.2). The lowest LAI was at the Tall Willow collars with 0.39 m² m⁻². Note that LAI was not measured directly under the willow shrubs and would be expected to be much greater there. The Bare Ground site had some *Poa arctica* within the collar which was removed at the beginning of the study (Table 4.1) while the area around the collar had an LAI of 0.44 m² m⁻² associated with a sparse distribution of this grass.

Table 4.1 Plant species within both collars combined for every site.

Sites	Plant species
Tundra	<i>Cassiope tetragona</i> , <i>Vaccinium uliginosum</i> , <i>Lupinus arcticus</i> , <i>Betula glandulosa</i> , <i>Dryas integrifolia</i> , <i>Vaccinium uliginosum</i> , <i>Bistorta vivipara</i> and <i>Sphagnum</i> spp.
Peat Plateau	<i>Vaccinium uliginosum</i> , <i>Cassiope tetragona</i> , <i>Andromeda polifolia</i> , <i>Ledum descumbens</i> , <i>Pyrola grandiflora</i> , <i>Betula glandulosa</i> , <i>Poaceae</i> spp. and <i>Vaccinium vitis-idaea</i>
Cotton Grass	<i>Eriophorum scheuzeri</i> , <i>Juncus balticus</i> , <i>Equisetum arvense</i> , <i>Deschampsia capitosa</i> , <i>Carex</i> spp., and <i>Poaceae</i> spp.
Tall Willow	<i>Equisetum arvense</i> , <i>Poaceae</i> spp., <i>Salix pulchra</i> , <i>Poa arctica</i> and <i>Salix alasensis</i> .
Dense Willow	<i>Salix glauca.</i> , <i>Equisetum arvense</i> , <i>Arctograstis latifolia</i> and <i>Betula nana</i> .
Low Willow	<i>Salix glauca</i> , <i>Equisetum arvense</i> , <i>Poaceae</i> spp. and <i>Sphagnum</i> spp.
Grass	<i>Carex aquatilis</i> and <i>Calamagrostis stricta</i> .
Wet Sedge 2	<i>Carex aquatilis</i> and <i>Calamagrostis stricta</i> .
Wet Sedge 1	<i>Carex aquatilis</i> .
Bare Ground	<i>Poa arctica</i> .

Table 4.2 Summary of vegetation and soil characteristics for all the sites. Biomass values represent vegetation within collars. Values for leaf area index (LAI) (3 measurements/site), soil bulk density (2 samples/site), soil volumetric water content (11 measurements/site) and soil temperature (24 measurements/site, only Bare ground had 11 measurements) are expressed as mean and standard error in parentheses.

	Tundra	Peat Plateau	Dense Willow	Low Willow	Tall Willow	Cotton Grass	Wet Sedge 1	Wet Sedge 2	Grass	Bare Ground
<i>Vegetation characteristics</i>										
Vegetated collar biomass (g m ⁻²)	12.79	8.07	1.40	4.68	1.03	7.17	4.07	5.19	4.60	0.66
Unvegetated collar biomass (g m ⁻²)	9.76	9.96	0.70	4.54	1.34	6.06	3.52	5.19	4.46	N/A
Leaf Area Index (LAI)	1.78 (0.40)	1.95 (0.4)	3.40 (0.47)	1.60 (0.14)	0.39 (0.21)	1.70 (0.99)	4.22 (0.80)	4.18 (0.88)	3.94 (0.13)	0.44 (0.31)
<i>Soil characteristics</i>										
Thaw depth (cm) – July 11	50	23	37	65	90	65	40	50	36	81
Thaw depth (cm) – August 6	63	26	>100	>100	>100	80	49	74	52	>100
Bulk density (g cm ⁻³) 0-15 cm	1.14 (0.36)	0.14 (0.04)	0.34 (0.08)	0.54 (0.10)	0.55 (0.09)	0.49 (0.08)	0.25 (0.07)	0.53 (0.17)	0.62 (0.02)	0.91 (0.04)
Volumetric water content (%) (0-20 cm)	42.7 (0.5)	53.0 (0.2)	54.3 (6.4)	56.5 (0.5)	45.3 (0.4)	54.2 (0.5)	71.5 (1.0)	67.2 (0.9)	59.9 (0.3)	52.7 (0.9)
Soil temperature (°C) at 10 cm	5.9 (0.3)	3.6 (0.2)	6.7 (0.3)	7.3 (0.3)	8.1 (0.3)	7.2 (0.3)	7.2 (0.4)	6.5 (0.4)	5.2 (0.3)	10.4 (1.2)
pH (15-20 cm)	6.37	5.38	5.19	4.80	6.25	5.02	5.46	5.34	5.85	3.31
Electrical conductivity (µS) (15-20 cm)	142.1	1145	321	807	236	5290	966	1006	2330	3580
Mineral soil texture (15-20 cm)	Silt loam	Organic	Loam	Sandy loam	Sand	Loamy sand	Silt loam	Silt loam	Loam	Sandy loam

4.1.2 Soil characteristics

Thaw depth for all sites increased over the field season (Table 4.2). The increase ranged from 3 cm (Peat Plateau) to more than 63 cm at sites with willows (Dense Willow, Low Willow, Tall Willow). The Bare Ground site also had substantial thaw with an increase of over 29 cm between July 11 and August 6 and had the deepest thaw depth of all the sites, over 100 cm. The site with least thaw (26 cm) was in the organic soil at the Peat Plateau site (Table 4.2). Soil temperature at 10 cm below the surface was highest at the Bare Ground site with an average of 10.4°C (Table 4.2). Soils were coolest at the Peat Plateau with average values of 3.6°C (Table 4.2). As expected, average soil temperature was positively correlated with maximum thaw depth ($r = 0.66$, $p = 0.002$). Generally, the sites with sand textured mineral soils (which includes the Willow sites and the Bare Ground) had the highest soil temperatures and thaw depths.

The sites ranged from moist to wet throughout the basin (Table 4.2). Volumetric water content was greatest (on average 72 and 67%) for the Wet Sedge sites. The driest site was the Tundra with 42.7% volumetric water content. Given the near surface bulk density at that site, this was still a high soil water content representing about 75% saturation.

Electrical conductivity varied across the basin with very high values for the Bare Ground and Cotton Grass sites and lower values for Willow sites and Tundra (Table 4.2). Soils were most acidic at the Bare Ground site ($\text{pH} = 3.31$) and were only slightly acidic at the Tundra site ($\text{pH} = 6.37$). For the rest of the sites, pH conditions varied from 4.80 to 6.25. pH did not significantly correlate with any of the soil variables listed in Tables 4.2 or 4.3.

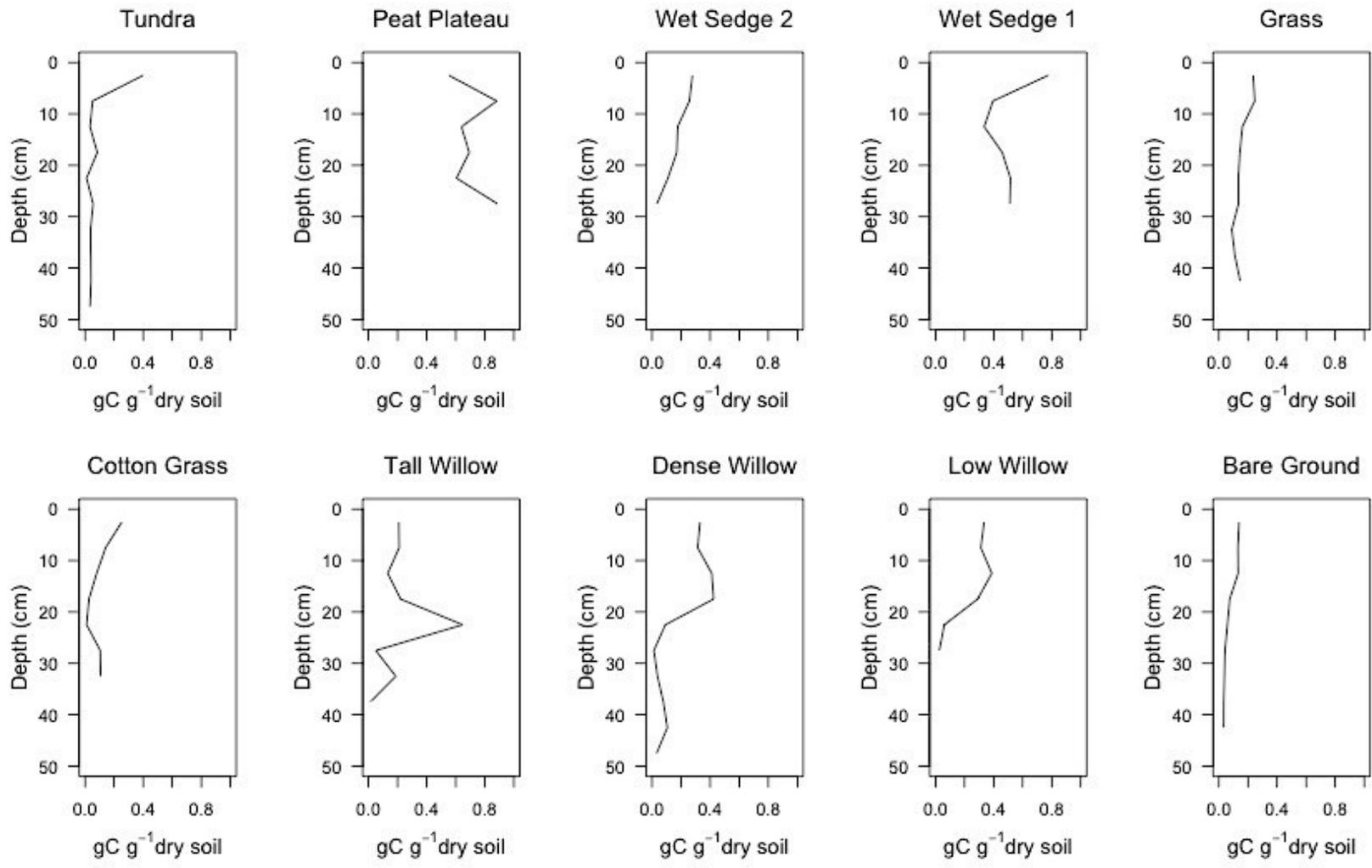
For the top 15 cm of the soil profile, the bulk density was greater than 1 g cm^{-3} only at the Tundra site. The lowest bulk density was in the organic soil at the Peat Plateau site (0.14 g cm^{-3}). As expected, bulk density was negatively correlated with %OM (0-30 cm) ($\rho = 0.59$, $p = 0.009$).

Lowest %OM was at the Bare Ground site and highest was at the Peat Plateau site (Table 4.3). The %OM and aboveground biomass ($\rho = -0.16$, $p = 0.509$) and LAI ($\rho = 0.29$, $p = 0.407$) did not significantly correlate. The ratio of %C to %OM was on average 0.48 ± 0.02 . The only sites with %N above 1% was the Peat Plateau and Wet Sedge 1 sites. The C:N was highest for Peat Plateau with a ratio of 22.5 and the lowest was at the Bare Ground site where the ratio was 10.7 where the %C was only 10% of that at the Peat Plateau site.

Over the soil profile to the depth of the frost table or water table, the majority of sites' %OM decreased with depth (Figure 4.1). The only exception was at the Tall Willow site which seemed to have buried layers of organic material, which were confirmed visually. This site was located near the edge of the basin where erosion from the bank during the lake phase may have resulted in OM layering (Ovenden, 1986). Total C to a depth of 15 cm on an area basis was greatest at Low Willow site with $14.03 \text{ kg C m}^{-2}$ followed by Grass and Dense Willow sites, both with about $\sim 10.5 \text{ kg C m}^{-2}$ (Table 4.3). Integrating between 0 and 30 cm, Peat Plateau had the highest greatest total C with 9.76 kg C m^{-2} followed by Grass with 8.08 kg C m^{-2} . The lowest 0-30 cm values of total C were at the Tall Willow, Wet Sedge 2 and Bare Ground sites with values $< 5.0 \text{ kg C m}^{-2}$. Total N to a depth of 15 cm and to a depth of 30 cm positively correlated with total C ($r = 0.88$, $p = < 0.001$ and $r = 0.89$, $p = < 0.001$, respectively) (Table 4.3).

Table 4.3 Mean values with standard error for organic matter, nitrogen and carbon content in (%), the average carbon to nitrogen ratio for the top 30 cm and the kg m⁻² of total C and N for the 0-15 cm and 15-30 cm soil layers of each site. Deepest soil measured was: a; 10 cm b; 20 cm c; 25 cm. Soil characteristics to 30 cm estimated to be the same as this deepest sampled depth.

Sites	%OM	%N	%C	C/N	Total C kg m ⁻² 0-15 cm	Total C kg m ⁻² 0-30 cm	Total N kg m ⁻² 0-15 cm	Total N kg m ⁻² 0-30 cm
Tundra	10.6 (6.0)	0.24 (0.06)	3.68 (1.06)	15.0 (0.52)	4.88	6.20	0.31	0.44
Peat Plateau	71.0 (5.8)	1.84 (0.07)	40.83 (2.21)	22.52 (1.51)	7.66	9.76 ^c	0.35	0.46 ^c
Low Willow	23.5 (6.2)	0.67 (0.16)	11.09 (2.51)	16.27 (1.42)	14.03	5.04 ^c	0.91	0.27 ^c
Dense Willow	26.4 (7.0)	0.85 (0.16)	14.62 (2.65)	19.29 (2.42)	10.46	6.91	0.62	0.36
Tall Willow	28.4 (9.2)	0.69 (0.09)	11.34 (1.77)	15.82 (0.49)	4.48	3.43 ^b	0.30	0.19 ^b
Cotton Grass	10.3 (4.4)	0.41 (0.09)	6.20 (1.56)	12.86 (1.22)	6.91	1.02 ^c	0.45	0.11 ^c
Wet Sedge 1	49.2 (9.8)	1.46 (0.12)	23.04 (1.99)	15.81 (0.06)	7.77	N/A	0.49	N/A
Wet Sedge 2	19.8 (3.1)	0.74 (0.04)	9.91 (0.52)	13.49 (0.08)	4.72 ^a	3.37 ^b	0.35 ^a	0.25 ^b
Grass	18.4 (2.1)	0.74 (0.11)	9.30 (1.33)	13.06 (0.40)	10.52	8.08	0.83	0.64
Bare Ground	9.6 (1.8)	0.40 (0.06)	4.25 (0.62)	10.69 (0.12)	8.52	4.28	0.80	0.39



41 Figure 4.1 Organic matter (g C g⁻¹ of dry soil) with depth for each site.

4.2 Carbon fluxes

4.2.1 Temporal variation

CO₂ fluxes varied throughout the field season with lower emissions generally corresponding to a decrease in daily mean air temperature (Figure 4.2). Median fluxes were greatest at the beginning of the field season (12-14-15 July) and on the 28 July when a 2nd warm period occurred. Fluxes were lowest at the beginning of August when air temperature was lowest.

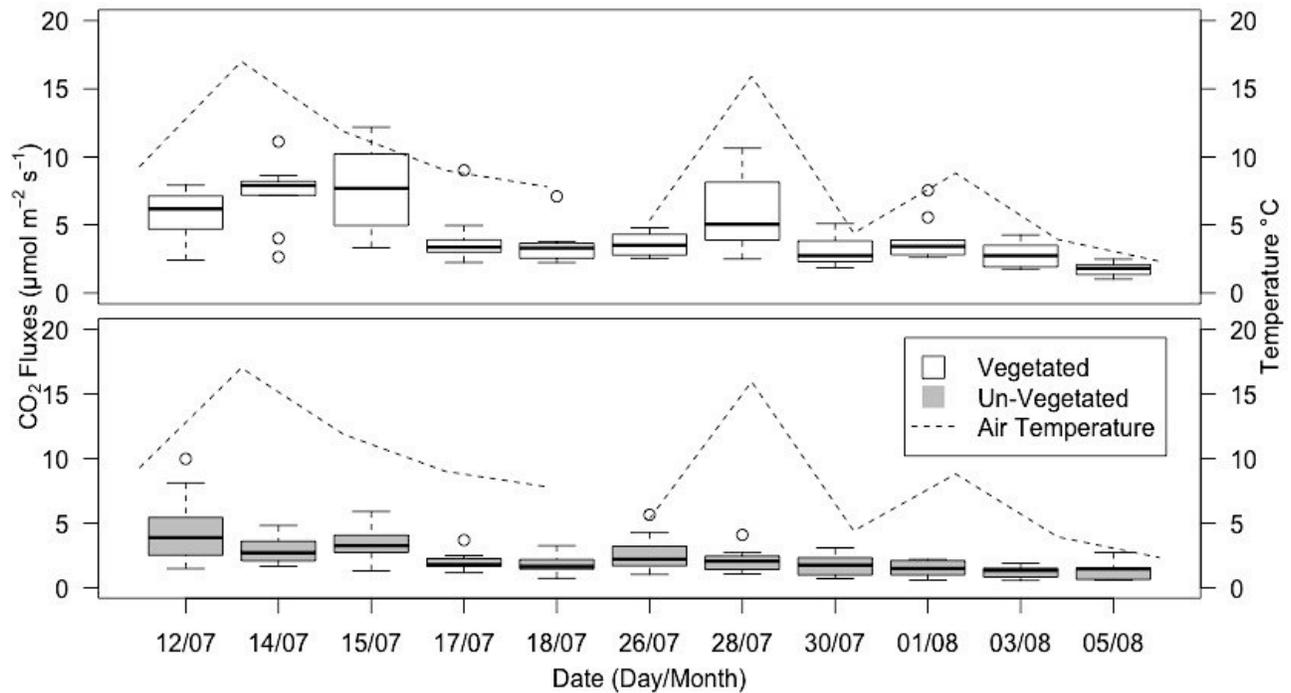


Figure 4.2 Temporal variations of CO₂ fluxes for all sites with both unvegetated and vegetated collars. Note that dates on the x-axis are spaced evenly. Box plots show the interquartile range with the vertical size of the box, the median flux is the line in the center, the whiskers represent maximum and minimum values excluding outliers and the circles represent outliers which are defined as 1.5 times outside of the interquartile range above the upper quartile and below the lower quartile.

Methane fluxes also varied throughout the field season but appeared less influenced by temporal variations in temperature. There was greater temporal variation in CH₄ flux in vegetated collars vs unvegetated collars (CV = 42.6 and 35.5%, respectively) (Figure 4.3). Only the Wet Sedge sites had relatively high CH₄ emissions and these fluxes are presented as outliers in Figure 4.3.

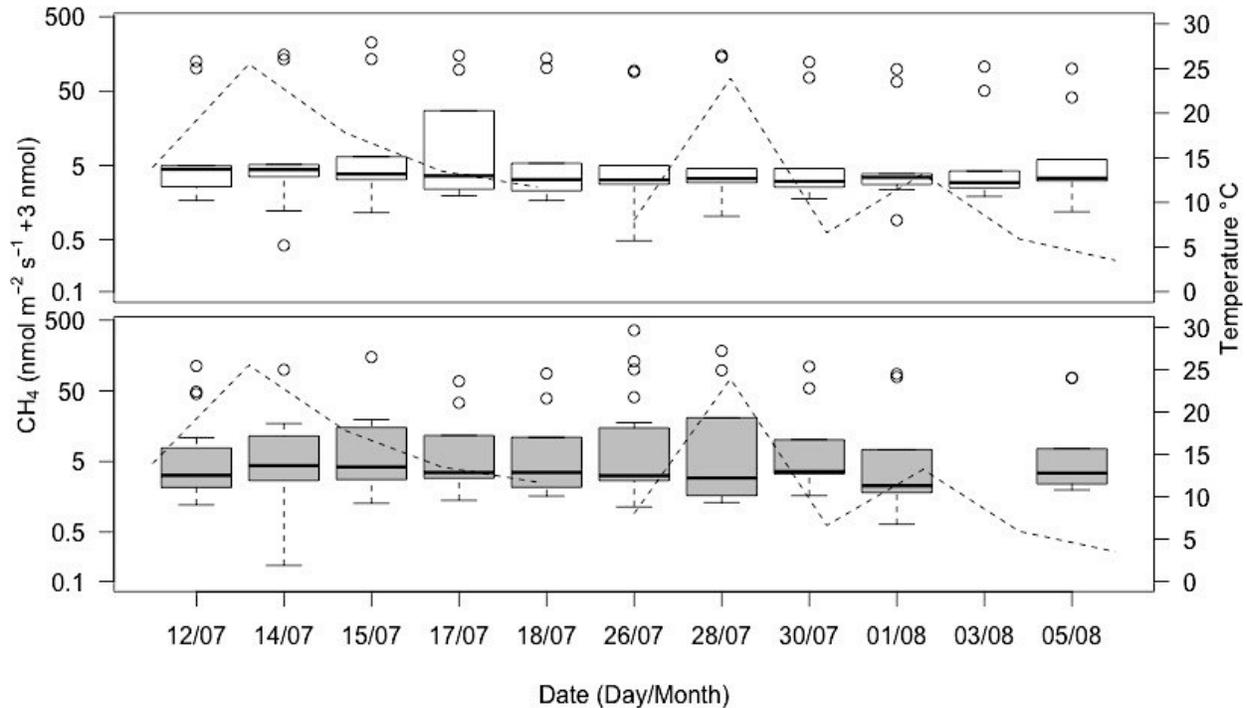


Figure 4.3 Temporal variations of CH₄ flux for all sites with vegetated collars (top) and unvegetated collars (bottom). The dashed line represents mean air temperature. 3 nmol m⁻² s⁻¹ have been added to each measurement to offset negative fluxes on a log scale. Box plots are described in Figure 4.2.

Nitrous oxide emissions were non-negligible only at the Bare Ground site (Figure 4.4) and did not vary with soil temperature ($r = 0.14$, $p = 0.68$) or soil moisture ($r = -0.26$, $p = 0.56$).

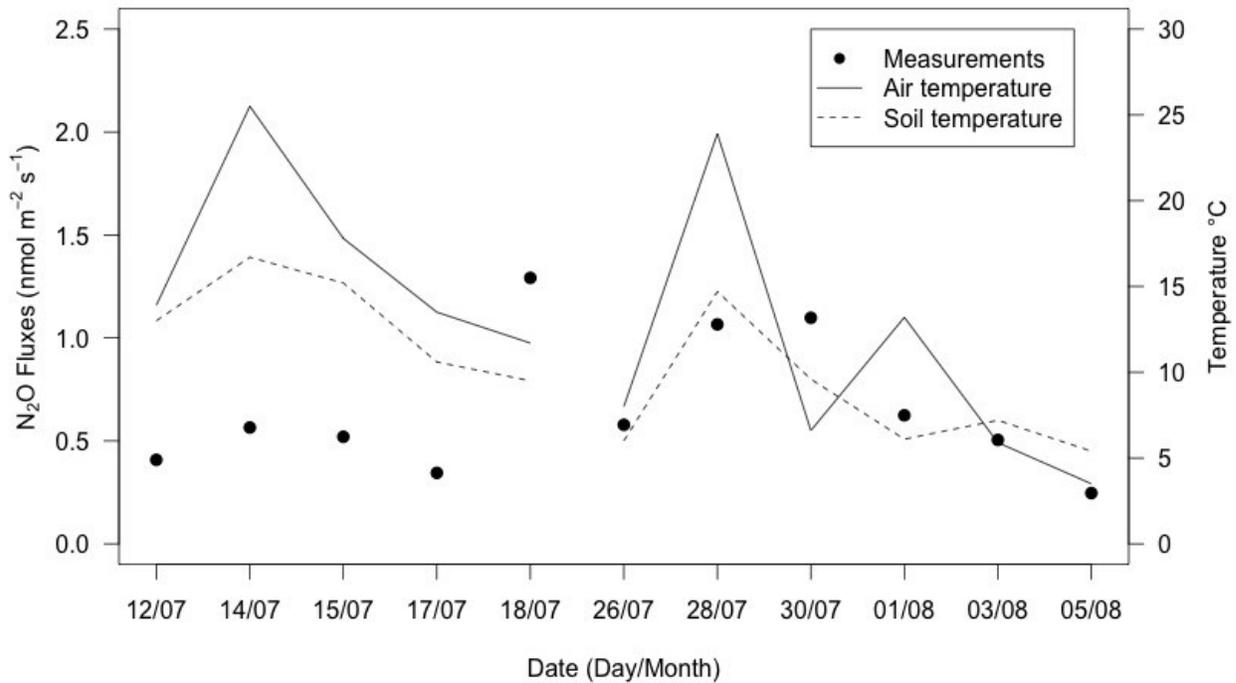


Figure 4.4 Temporal variations of N₂O flux for Bare Ground site with only the unvegetated collar (circles) and temporal variations in soil temperature (dashed line) measured beside the collar and daytime average air temperature on the measurement dates.

4.2.2 Spatial variation

Carbon dioxide emissions were significantly greater from the vegetated collars than from the unvegetated collars within a given site except for at the Dense Willow ($W = 91$, $p = 0.138$) and the Grass ($W = 99$, $p = 0.059$) sites (Figure 4.5). Overall, the vegetated collars had significantly greater fluxes compared to the unvegetated collars (with LFH intact: $W = 3941$, $p = <0.001$ and with LFH removed: $W = 5969$, $p = <0.001$), reflecting the important contribution of autotrophic respiration. The mean air temperature before removal of the LFH layer was 16.6°C and 9.9°C after removal of the layer. Despite lower temperatures, the stronger difference between the paired

collars after removal of the LFH suggests that heterotrophic respiration within the LFH layer with its “fresh” plant residues is also an important contributor to total respiration at these sites.

Median CO₂ fluxes were greatest from the vegetated collars at the Tundra and Cotton Grass sites while the sites with greatest 95% ranges (temporal variability) were Tundra and Wet Sedge sites. The lowest fluxes were from the Bare Ground site where there was the least variability as well.

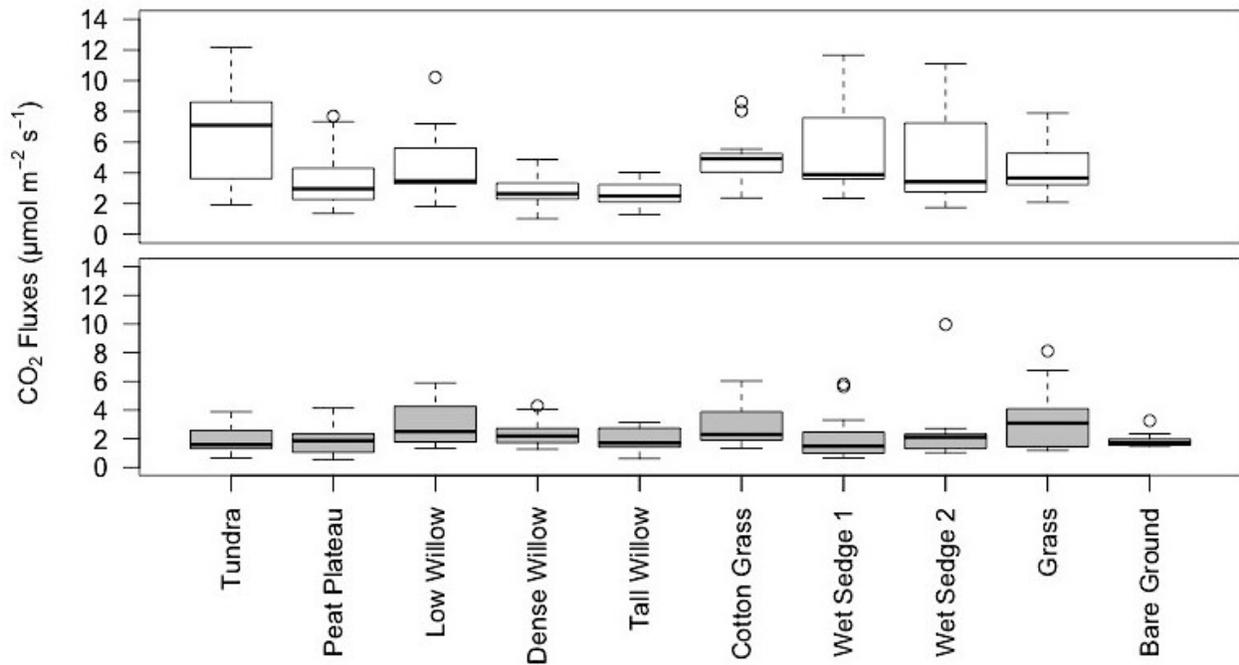


Figure 4.5 Carbon dioxide fluxes from vegetated collars (above) and unvegetated collars (below) at 10 sites. Box plots are described in Figure 4.2.

Wet Sedge sites emitted CH₄ at one to two orders of magnitude greater than the other sites. Methane emissions from the Wet Sedge sites’ vegetated collars were significantly greater than emissions from the unvegetated collars (Figure 4.6). Low Willow site had significantly more CH₄

emitted from the unvegetated collar. However, at this site, median CH₄ emissions remained less than 8.03 nmol m⁻² s⁻¹. The Tundra site had a significant difference between collars with less CH₄ uptake at the unvegetated collar ($W = 35, p = 0.017$). Emissions did not differ significantly between collars at all the other sites. Median fluxes were negative, indicating CH₄ uptake, at Peat Plateau and Bare Ground sites regardless of collar type (Figure 4.6). Median fluxes were positive but near zero at Grass, Tall Willow, Dense Willow (unvegetated) and Cotton Grass (vegetated) sites with values less than 0.8 nmol m⁻² s⁻¹.

4.2.3 Controlling factors

One of the important factors controlling temporal variations in CO₂ emissions was temperature. CO₂ fluxes at some sites increased with increasing air temperature (Figure 4.7). When temperatures were low, fluxes from many of the sites were low. At higher temperature, there was a greater spread in the CO₂ fluxes.

The Q₁₀ exponential relationship with air temperature described the variation in CO₂ emissions from vegetated collars well (r^2 ranged from 0.47 to 0.89) while in unvegetated collars, r^2 values ranged from <0.001 to 0.49 (Table 4.4). The R₁₀ parameter, an estimate of respiration normalized to 10°C, for each site and collar type did not significantly correlate with %C or %N (Figure 4.8 and 4.9) nor with any of the environmental variables (thaw depth, pH, organic matter content and volumetric water content). The vegetated collars had higher R₁₀ values, compared to the unvegetated collar for every site. The Q₁₀ was higher at the vegetated collars for every site except Tall Willow. The lowest value for Q₁₀ at 0.95 was at the Bare Ground site, which means that with an increase in temperature, there was negligible change in flux (Table 4.4).

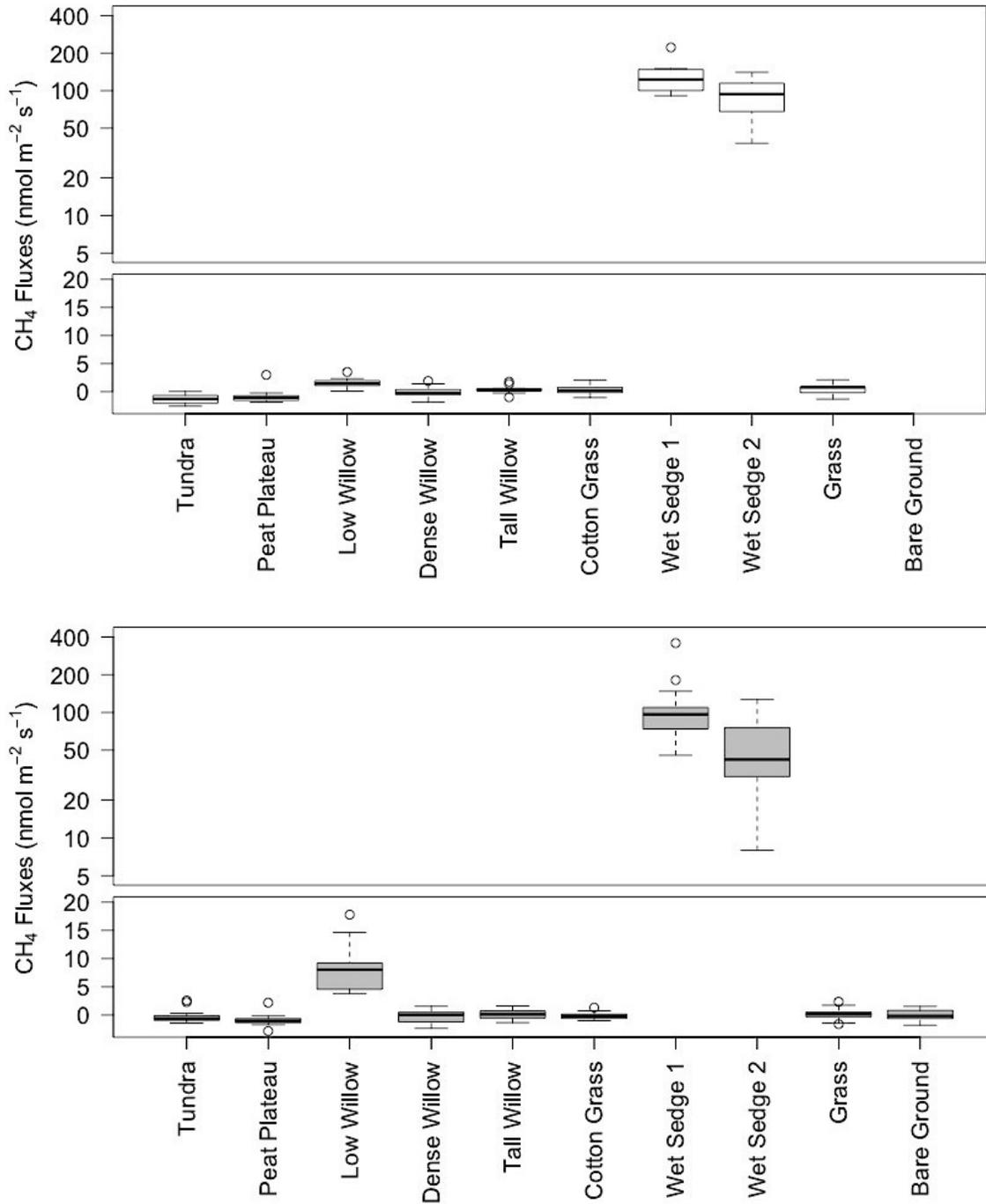


Figure 4.6 Methane fluxes from vegetated collars (above) and unvegetated collars (below). Note that the top portion of each panel is on a Log scale and overlaps in range with bottom portions. The box plots are described in Figure 4.2.

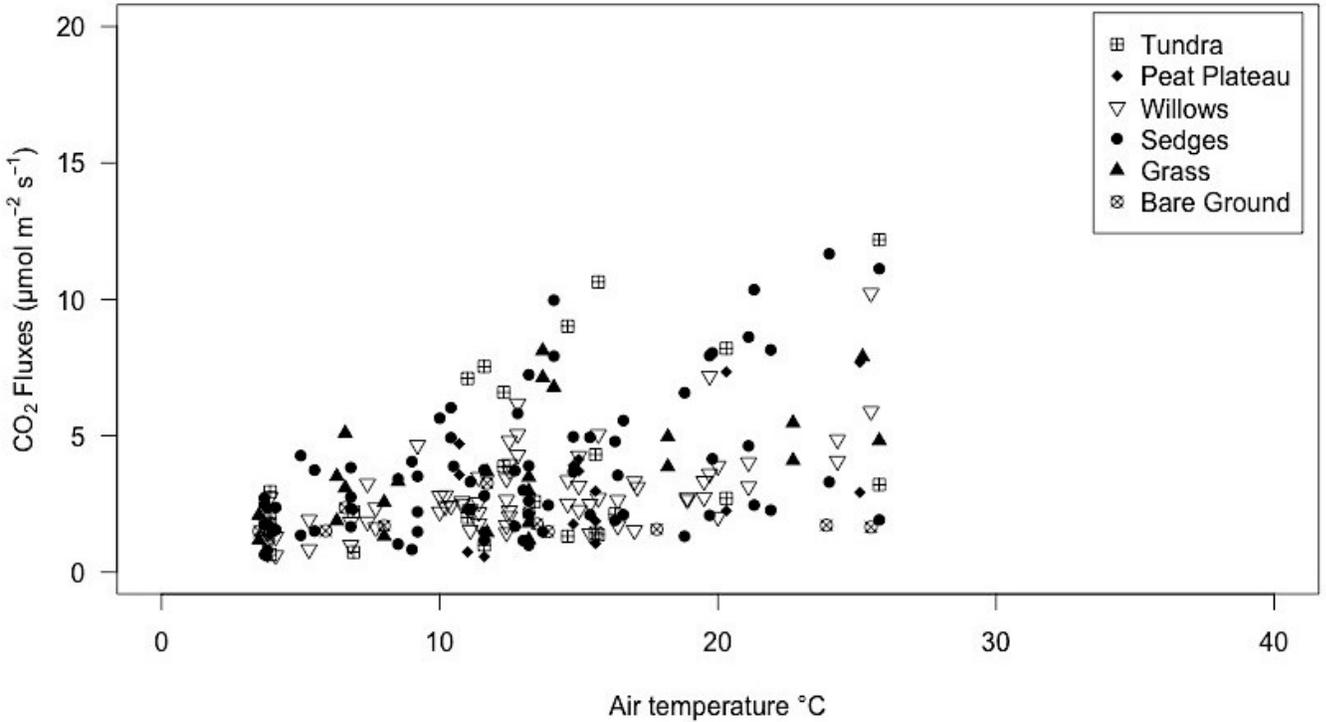


Figure 4.7 CO₂ fluxes classified by vegetation group vs. air temperature for vegetated and unvegetated collars.

In general, CH₄ emissions increased with soil volumetric water content (Figure 4.10). This trend was largely driven by the Wet Sedge sites' CH₄ fluxes as high as 300 nmol m⁻² s⁻¹ and a mean volumetric water content above 65% (Table 4.2). Median CH₄ fluxes from each collar correlated significantly with VWC, iron (Fe), manganese (Mn) and boron (B). Only the median CH₄ fluxes for vegetated collars correlated significantly with total adsorbed N (Table 4.5). In addition, the relationship between CO₂ and CH₄ was small but significant ($r = 0.21, p = 0.001$).

Median CO₂ fluxes and R₁₀ from all the vegetated collars did not correlate with any environmental variables (Table 4.5). However, both median CO₂ fluxes and R₁₀ for the unvegetated collars significantly correlated with magnesium (Mg) adsorption rates while R₁₀ correlated with VWC and calcium (Ca) and sulfur (S) adsorption rates (Table 4.5).

Table 4.4 Coefficient of temperature sensitivity for a change of 10 °C (Q_{10}) and respiration rate at 10°C (R_{10}) for each site and collars with standard error in parentheses.

Site	Vegetated collars		Unvegetated collars	
	Q_{10}	R_{10} ($\mu\text{mol m}^{-2} \text{s}^{-1}$)	Q_{10}	R_{10} ($\mu\text{mol m}^{-2} \text{s}^{-1}$)
Tundra	1.72 (0.23)	5.35 (0.74)	1.56 (0.30)	1.58 (0.27)
Peat Plateau	2.04 (0.22)	2.78 (0.32)	1.35 (0.33)	1.79 (0.34)
Low Willow	1.93 (0.17)	3.64 (0.33)	1.74 (0.27)	2.46 (0.35)
Dense Willow	1.57 (0.16)	2.40 (0.21)	1.42 (0.22)	2.21 (0.26)
Tall Willow	1.67 (0.11)	2.16 (0.10)	1.78 (0.36)	1.55 (0.23)
Cotton Grass	1.86 (0.20)	3.97 (0.33)	1.45 (0.37)	2.62 (0.46)
Wet Sedge 1	2.23 (0.19)	3.85 (0.35)	1.38 (0.48)	1.98 (0.54)
Wet Sedge 2	2.04 (0.23)	3.56 (0.46)	1.05 (0.47)	2.36 (0.74)
Grass	1.53 (0.21)	3.68 (0.50)	1.45 (0.34)	2.97 (0.63)
Bare Ground	N/A	N/A	0.95 (0.12)	1.90 (0.18)

The temporal variations in CO_2 fluxes correlated with environmental variables differently depending on whether there was vegetation in the collar or not, likely reflecting the relative importance of autotrophic vs. heterotrophic respiration (Figure 4.11). Fluxes correlated positively and significantly with air temperature at all vegetated collars but only at 3 sites with unvegetated collars. Fluxes correlated positively and significantly with soil temperature at all vegetated collars except for one site but not at any site for unvegetated collars. CO_2 fluxes at all but 3 sites were significantly correlated to VWC in unvegetated collars but not at any sites in vegetated collars.

Compared to CO_2 fluxes, temporal variations in CH_4 fluxes were less consistently correlated with environmental variables. At the Wet Sedge sites where CH_4 emissions were relatively high, the temporal variations in these fluxes were significantly and positively correlated to soil and air temperature only in the vegetated collars (Figure 4.12). Emissions from the unvegetated collars at these sites were significantly and negatively correlated to VWC and in one

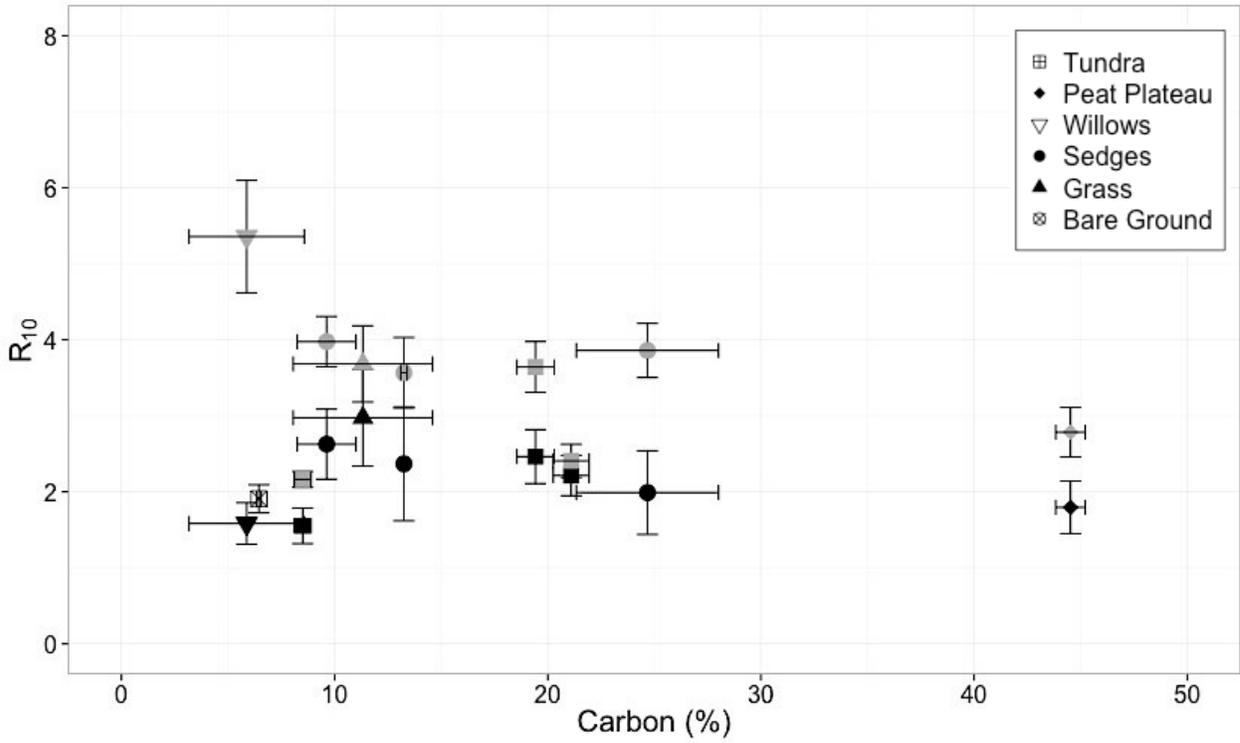


Figure 4.8 Mean respiration rate at 10 °C (R_{10}) with mean %C of the total soil profile (to 30 cm).

Black symbols represent R_{10} for unvegetated collars and gray symbols represent vegetated collars.

Error bars represent ± 1 SE.

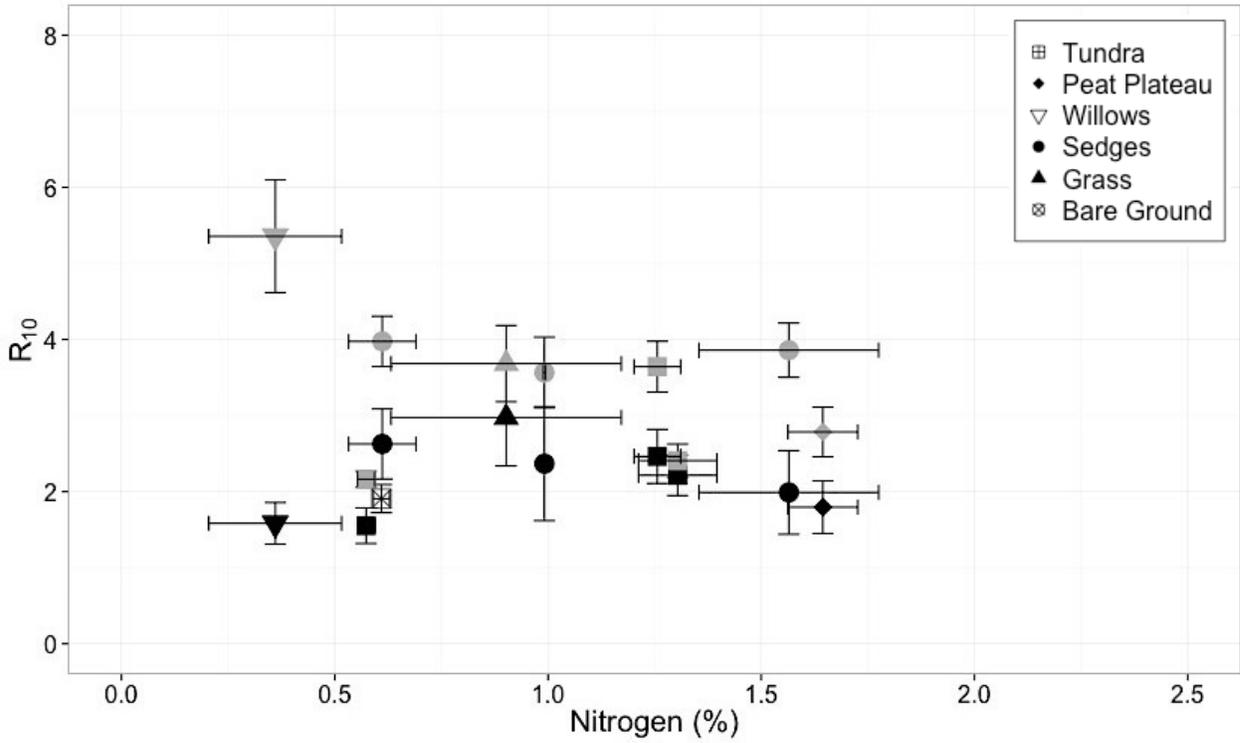


Figure 4.9 Mean respiration rate at 10 °C with mean %N of the total soil profile to a depth of 30 cm. Black symbols represent R₁₀ for unvegetated and gray symbols represent vegetated collars.

Error bars represent ± 1 SE.

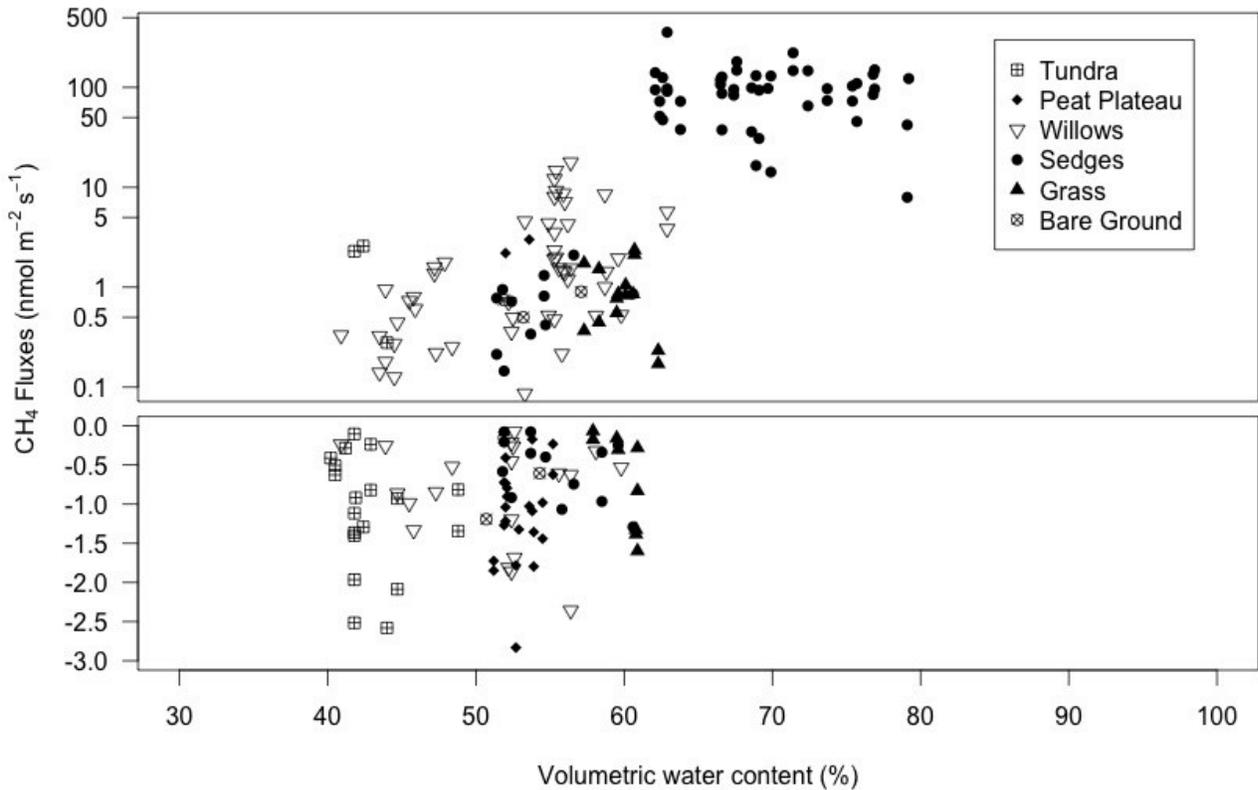


Figure 4.10 Methane fluxes from vegetated and unvegetated collars vs. soil volumetric water content for each of the vegetation types. The negative fluxes in the bottom graph indicate methane consumption.

collar, also to soil temperature. The Low Willow site with the next highest CH₄ emissions correlated significantly with air temperature in both the vegetated and unvegetated collars. At the Peat Plateau site where small rates of CH₄ uptake occurred, air temperature was significantly and negatively correlated to temporal variations in these fluxes (Figure 4.12).

4.2.4 Radiocarbon signatures

Overall the vegetated collars had significantly more positive $\Delta^{14}\text{C}$ values suggesting emissions releasing more recently sequestered C ($S = 18.5$, $p = 0.027$) although the $\Delta^{14}\text{C}$ from vegetated and unvegetated collars were significantly and positively correlated ($\rho = 0.75$, $p = 0.02$)

(Figure 4.13). The bare ground site has the most negative $\Delta^{14}\text{C}$. Based on the age model, this value represented 2300 yr BP (see Appendix 2 for values). This would be the integrated age of the carbon emitted from the ground mixed with atmospheric C within the collar. Correlations between $\Delta^{14}\text{C}$ and the variables listed in Tables 4.2-4.5 were generally not significant for either vegetated or unvegetated collars. The only exception was $\Delta^{14}\text{C}$ from unvegetated collars which were negatively correlated with bulk density ($\rho = -0.77$, $p = 0.009$) and positively correlated with %N ($\rho = 0.644$, $p = 0.044$).

Table 4.5 Spearman's correlation coefficients showing significant relationships between environmental variables and median CO₂ and CH₄ fluxes for both vegetated and unvegetated collars, in bold font when correlations are significant (p values < 0.05).

Variables	median CO ₂ Veg	median CO ₂ unveg	median CH ₄ Veg	median CH ₄ unveg	R ₁₀ Veg	R ₁₀ Unveg
Thaw	-0.55	0.15	0.12	0.19	-0.51	-0.10
Soil Temp.	-0.24	-0.18	0.41	0.27	-0.18	-0.14
VWC	0.1	0.32	0.87	0.79	0.07	0.67
OM	-0.58	-0.08	0.3	0.28	-0.53	-0.12
Biomass	0.6	0.08	-0.45	-0.39	0.58	0.04
LAI	0.27	0.01	0.42	0.43	0.17	0.36
pH	0.15	-0.42	-0.23	-0.03	0.10	-0.42
eS	0.23	0.42	0.23	-0.1	0.20	0.55
Δ¹⁴C	-0.02	-0.29	0.45	0.45	0.03	-0.09
Ca	0.1	0.59	0.68	0.59	0.13	0.78
Mg	0.08	0.76	0.32	0.13	0.1	0.78
K	-0.28	-0.26	-0.5	-0.31	-0.25	-0.57
P	-0.22	0.14	0.43	0.64	-0.18	0.22
Fe	-0.27	0.32	0.77	0.82	-0.22	0.48
Mn	-0.07	0.42	0.83	0.72	-0.08	0.59
B	-0.3	0.22	0.82	0.86	-0.27	0.41
S	0.18	0.48	0.43	0.2	0.13	0.68
Total N	-0.16	-0.38	0.80	0.55	-0.22	-0.19

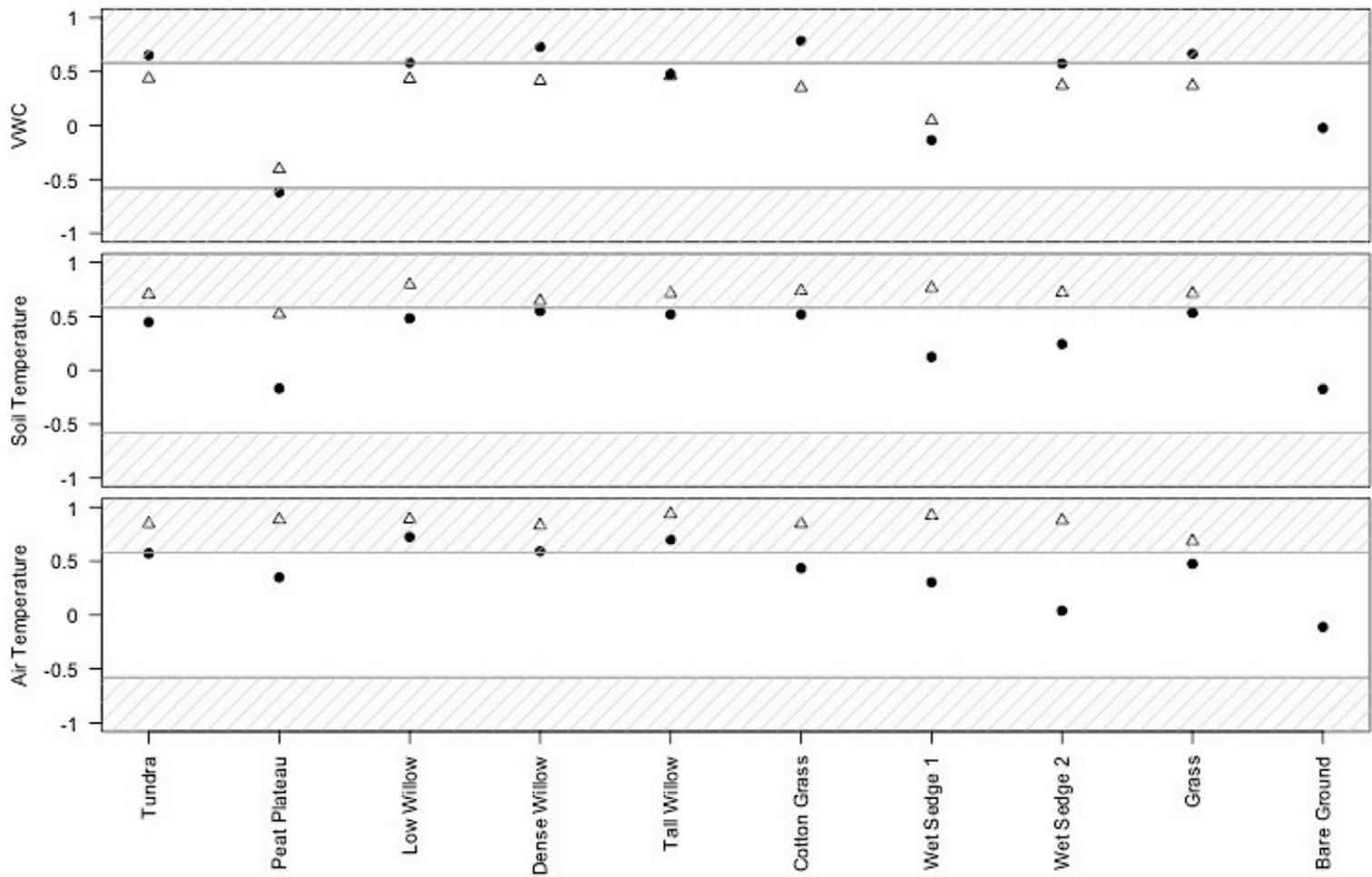


Figure 4.11 Pearson's correlation coefficients for CO₂ flux and environmental variables per sites. Triangles represent vegetated collars and black circles represent un-vegetated collars. Significant values ($p < 0.05$) are located in the gray zone

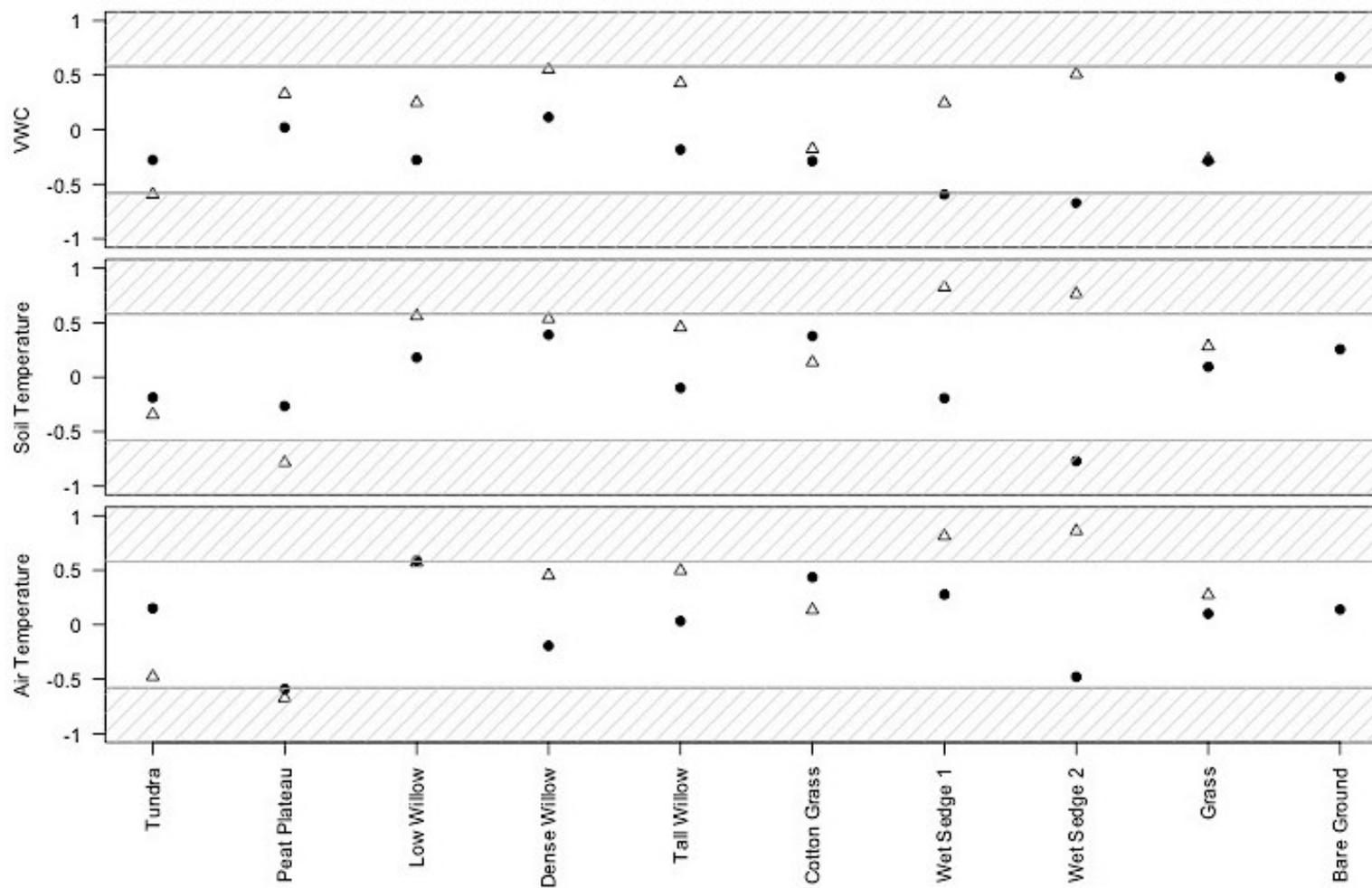


Figure 4.12 Pearson's correlation coefficients for (+3 nmol m⁻² s⁻¹) log transformed CH₄ fluxes and environmental variables. Triangles represent vegetated collars and black circles represent un-vegetated collars. Significant values (p < 0.05) are located in the gray zone.

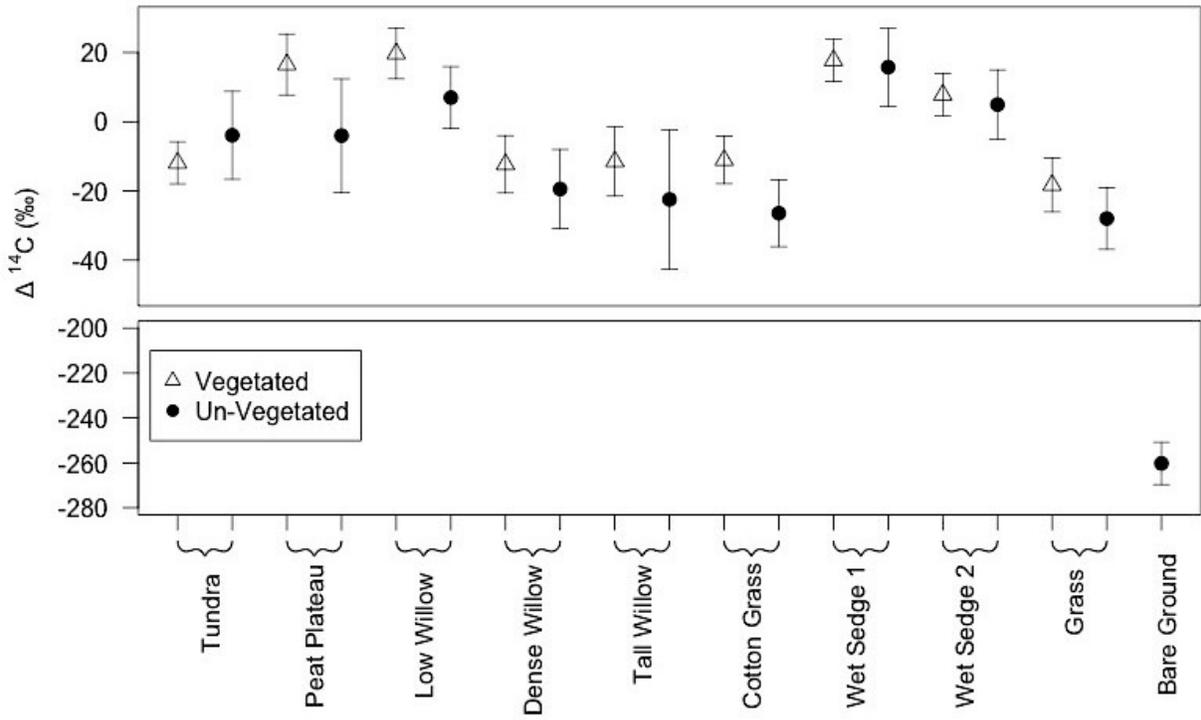


Figure 4.13 $\Delta^{14}\text{C}$ of respired CO_2 for vegetated and un-vegetated collars at each site. Error bars represent the measurement uncertainty.

5 Chapter: Discussion

The emissions of CO₂, CH₄, and N₂O within the Ilisarvik drained lake basin were highly variable. Each measurement site possessed different characteristics that may affect microbial activity, plant respiration, and gas transport to the atmosphere and thus, emission rates. This chapter will discuss the relative importance of these factors in the context of observations from other subarctic and Arctic ecosystems.

5.1 Carbon dioxide fluxes

Ecosystem respiration (ER), the sum of both autotrophic and heterotrophic respiration processes may be increased by optimization of soil conditions such as soil composition, moisture and temperature that can positively affect microbial activity and plant growth (Vourlitis and Oechel, 1999; Davidson and Janssens, 2006; Oberbauer et al., 2007).

Air and soil temperature were found to be important variables controlling CO₂ emissions at the study site with stronger relationships when vegetation was present. Oberbauer et al. (2007) explained that an increase in temperature can stimulate photosynthesis, which increases the availability of photosynthate to support cell growth and reproduction but also increase microbial substrate supply into the soil. Higher temperatures should also lead to greater rates of autotrophic respiration, depending on substrate availability (Bunce, 2007). Thus, higher temperatures can increase CO₂ emissions from both above and belowground sources. Without the aboveground plant parts (due to clipping), autotrophic respiration is likely decreased considerably from belowground roots such that heterotrophic respiration is the main contributor to ER (Peterson and Billings, 1975) and may include the decomposition of the roots from the clipped plants. Since soil temperatures vary less than air temperature, variation in CO₂ emissions in vegetated collars is higher than in

unvegetated collars. A study by Peterson and Billings (1975) found similar results when comparing above and below ground ER where variation in fluxes were lower at sites without surface vegetation. At the unvegetated collars, most of the sites' CO₂ emissions increased with soil temperature. Only emissions from the unvegetated Peat Plateau and Bare Ground sites had a small negative and non-significant correlation. This might be due to other factors such as organic matter content which was very low at Bare Ground (9.6%), thus even if soil temperature increased, there might not be enough substrate available for an increase in decomposition. For Peat Plateau the slightly negative correlation may be due to low temperature throughout the field season (from 2.3 °C to 5 °C) being insufficient to stimulate measurable microbial decomposition. Alternatively, soil temperature was only measured at a depth of 10 cm which might not represent the main forcing since ER is distributed through the soil profile. In other words, correlation of CO₂ fluxes with soil temperature might be positive and stronger with temperature measured at different depths.

In contrast, the relationship between CO₂ emissions and VWC was stronger for unvegetated collars. This suggests VWC has a greater impact on below ground ER than aboveground and is likely more important for heterotrophic than autotrophic respiration processes. Increasing moisture levels may enhance CO₂ emission because water films in the soil enable the diffusion of substrates for microbial decomposition such as soluble organic C and extra cellular enzymes (Davidson and Janssen, 2006). In contrast, drier environments are more nutrient-limited and can limit microbial activity (Shaver and Chapin, 1991). Other studies have found a negative relationship between VWC and CO₂ fluxes (Fan et al., 1992; Vourlitis and Oechel, 1999; Oberbauer et al., 2007). This was explained by the correlation between higher and deeper water table (Vourlitis and Oechel, 1999) and reduced respiration under waterlogged, anoxic conditions (Oberbauer et al., 2007). Although respiration-soil moisture relations typically show an optimum

value, in this study this was not the case among sites or within a site. Vegetation well adapted to saturated soils ensured fresh OM inputs to the soil while anaerobic decomposers continued respiring while the aerenchyma tissues transporting CH₄ to the atmosphere likely helped support CO₂ production by aerobic microbes within the rhizosphere (Le Mer et Roger, 2001).

Over the field season, CO₂ effluxes varied from ~ 0.5 to 13 μmol m⁻² s⁻¹ (0.5 to 13.5 g C m⁻² day⁻¹) with a mean value of 3.3 μmol m⁻² s⁻¹ (3.4 g C m⁻² day⁻¹). Vegetated collars had significantly greater CO₂ emissions than unvegetated with emissions from both autotrophic and heterotrophic respiration. Radiocarbon Δ¹⁴C highlighted the rapid cycling and emissions of recently sequestered C with more modern CO₂ from collars with vegetation at most sites. Means of Δ¹⁴C for vegetated collars and unvegetated collars were 7.63 Δ¹⁴C ‰ and -25.94 Δ¹⁴C ‰ respectively. Without above ground vegetation, microorganisms decompose soil organic carbon that is available including older substrate. For example, even a small contribution (4.5%) of old C (-260 Δ¹⁴C ‰, based on respired CO₂ from the bare soil site) and a larger contribution (95.5%) of modern C (20 Δ¹⁴C ‰, based on atmospheric CO₂) results in a mean of 7.4 Δ¹⁴C ‰, equivalent to what was observed at the vegetated collars. Schuur et al. (2009) observed contributions of ~ 0.5 to 44% from old C in respired CO₂ depending on thaw depth. In this study, there was no overall correlation with thaw depth although the bare ground site with deepest thaw had the most negative Δ¹⁴C as noted above. Hicks Pries et al. (2017) showed that soil from all depths of a profile respired modern C despite increasingly older soil C fractions with depth. This study was conducted in a coniferous forest in non-permafrost terrain so turnover time of soil organic matter may be faster than in permafrost terrain where the substrate may be frozen seasonally for a longer period each year, limiting microbial activity.

The highest fluxes for vegetated collars were found at the Tundra site followed by the Wet Sedge and Low Willow sites with fluxes above $10 \mu\text{mol m}^{-2} \text{s}^{-1}$. For the unvegetated collars, the highest fluxes were found at Wet Sedge, Grass, Cotton Grass and Low Willow with values above $5 \mu\text{mol m}^{-2} \text{s}^{-1}$. These higher fluxes may be explained by a number of factors. The Tundra site had the highest aboveground plant biomass while the Wet Sedge 1 & 2 and the Grass sites had the highest LAI (Table 4.2), both of which suggest more plant material or more photosynthates to support respiration (Zona et al., 2010). The lowest fluxes were found at the Bare Ground with a mean value of $1.9 \mu\text{mol m}^{-2} \text{s}^{-1}$. This rate might be explained by having nearly zero plant biomass and LAI (Table 4.2) as well as the lowest soil OM% (Table 4.3) and $\Delta^{14}\text{C}$ (Figure 4.13). At that site, only heterotrophic respiration can contribute to ER, the substrate for decomposition is in limited supply and by virtue of age, may be older more humified material that is more recalcitrant and resistant to decomposition (Schuur et al., 2009). However, Biasi et al. (2005) have shown that even old OM may be relatively labile. In that study, under higher temperatures, this older material was preferably respired in Arctic soils. Despite these linkages between variables for certain sites, neither biomass nor LAI correlated with mean or median CO_2 fluxes among all the basin and tundra sites. This might be explained by other interacting environmental factors, such as VWC, soil temperature or substrate availability.

Similarly, although a relationship between thaw depth and CO_2 emissions was expected (Zona et al., 2010; Harden et al., 2012; Belshe et al., 2013; Kuhry et al., 2013; Schuur et al., 2015), there was none in this study. Again, the great variation in soil and vegetation conditions at Ilisarvik likely offsets or confounds the effect of deeper thaw depth and warmer soils on respiration. For example, by controlling for temperature using Eq 3 and removing the above ground vegetation,

unvegetated collar R_{10} values correlated significantly and positively with VWC, Ca, Mn and S. These nutrients may be supporting microbial growth.

5.2 Methane fluxes

Multiple studies show higher CH_4 fluxes at sites with higher soil moisture (Fan et al., 1992; Vourlitis et al., 1993; Merbold et al., 2009). In this study, the Wet Sedge sites had highest CH_4 emissions since the water table was near or above the ground, which would promote anaerobic conditions and methanogenesis while limiting methane oxidation. VWC in the top 20 cm of soil remained high (over 60%) at these sites and thus, there was no temporal relationship between VWC and CH_4 emissions. At the Wet Sedge sites, the unvegetated collars had significantly lower CH_4 effluxes than the vegetated collars. Among vegetated collars across the study area, LAI was slightly correlated with CH_4 emissions. Some studies have found a positive relationship between biomass and CH_4 emissions (Bartlett et al., 1992; Verville et al., 1998; Christensen et al., 2000). Christensen et al. (2000) explain this relationship by the presence of vascular tissues that acts as a conduit for CH_4 to the atmosphere, thus with an increase of root density, CH_4 emissions should increase as well. The plant species with aerenchyma tissues that can act in this way are mainly from the *Poaceae*, *Polygonaceae* or the *Cyperaceae* families (Yamauchi et al., 2012; Laan et al., 1989; Visser et al., 2000). Thus, sedges are often associated with higher CH_4 emissions compared to other vegetation types at a given site (Fan et al., 1992; Bubier et al., 1995). Verville et al. (1998) showed that when sedges were removed, the emissions of CH_4 decreased by 60% at a sedge meadow tundra site in Alaska. The authors explained this result by an initial interruption of gas transport of CH_4 via the roots and rhizomes. The next growing season, the reduced CH_4 was then explained by a change in available substrate for methanogenesis (Verville et al., 1998).

Acetoclastic methanogens require acetic acid to act as a terminal electron acceptor which can release CO₂ and CH₄ through acetate fermentation. The other major pathway of CH₄ production is CO₂ reduction with hydrogen (Kotsyurbenko et al., 2004). While moss-colonized bogs mainly produce CH₄ by CO₂ reduction, wet sedges such as *Carex* spp. mainly produce CH₄ by acetate fermentation (King et al., 1998; Le Mer and Roger, 2001).

Fluxes of CH₄ from the wet sites (Wet Sedge 1 & 2) varied between ~ 7 to 355 nmol m⁻² s⁻¹ (equivalent to 9.7 to 492 mg CH₄ m⁻² day⁻¹ assuming constant efflux) with a mean value of 97 nmol m⁻² s⁻¹ (134.4 mg CH₄ m⁻² day⁻¹) over the field season which is within the range of fluxes observed at other study sites with wet soils and similar vegetation types. Fan et al. (1992) observed a mean flux of 29 mg CH₄ m⁻² day⁻¹ in wet subarctic tundra in Alaska, Christensen et al. (2000) observed a mean flux of 151.2 mg CH₄ m⁻² day⁻¹ in a fen and grassland site located in a valley in Greenland, and Bartlett et al. (1992) observed fluxes varying between 15.6 to 426 mg CH₄ m⁻² day⁻¹ in the Yukon-Kuskokwim delta, Alaska.

The majority of sites at Illisarvik were sinks for CH₄ despite wet soils (VWC between 40 and 60%). Based on near surface soil bulk density values and assumed particle densities of 2.65 g cm⁻³ for mineral soils and 1.3 g cm⁻³ for organic soils, these VWC represented 59 to 94 % saturation over the top 20 cm of soil suggesting that there could be sufficient oxygen in most soils to limit CH₄ production and/or oxidize any CH₄ produced. Methane fluxes for these drier sites varied between -2.83 and 17.7 nmol m⁻² s⁻¹ (- 3.85 and 24.5 mg CH₄ m⁻² day⁻¹) with a mean of 0.54 nmol m⁻² s⁻¹ (0.74 mg CH₄ m⁻² day⁻¹). These fluxes are similar to those measured by Bartlett et al. (1992) at drier sites of an upland tundra along a 300 m transect, with values varying from -2.1 to 18.1 mg CH₄ m⁻² day⁻¹. A study by Merbold et al. (2009) investigated the impact of drainage on CH₄ fluxes during the growing season and found there was a reduction from 320.5 mg CH₄ day⁻¹

to $< 12.8 \text{ mg CH}_4 \text{ day}^{-1}$ following drainage. According to the authors, the residence time of CH_4 becomes shorter under aerobic conditions due to methanotrophic bacteria that oxidize CH_4 to CO_2 .

In contrast to CO_2 fluxes, temporal variations in CH_4 fluxes were not significantly correlated with air or soil temperature at the majority of sites. Only the vegetated collars of Wet Sedge 1 & 2 had significant and positive correlations. Dunfield et al. (1993) found that when temperature is above $15 \text{ }^\circ\text{C}$, CH_4 emissions can be enhanced, since methanogenesis is more stimulated by an increase in temperature compared to methanotrophy in Subarctic peat soils. When temperatures are low, the authors found that CH_4 remained low, regardless of an increase in temperature. Many studies found that an increase in temperature did not affect CH_4 emissions (Verville et al., 1998; Christensen et al., 2000; Merbold et al., 2009). Similarly, spatial variations in CH_4 fluxes were not correlated with thaw depth in contrast to findings by other studies (Vourlitis et al., 1993; Verville et al., 1998).

CH_4 fluxes were significantly correlated to the rates of absorbed Fe, Mn and B for both collar types. Fe and Mn mobility increases with decreasing redox conditions as iron and manganese oxides are reduced by anaerobic bacteria. Greater mobility of Fe and Mn is a sign of reducing conditions that might be suitable for methanogenesis as a metabolic pathway.

5.3 The Bare Ground site

The Bare ground site appeared to be unique from all the other sites in terms of soil characteristics and greenhouse gas fluxes. Since Illisarvik was drained in 1978, the Bare Ground site has never been colonised to any great extent by vegetation. The pond next to this site tends to expand seasonally, flooding the sandy 'beach' and impacting the soil chemistry of the site. For example, this site had relatively high electrical conductivity and lowest pH and highest Al ion

availability of all the sites in the basin. A drainage channel connected to the Beaufort Sea had to be dug in 2003 to decrease the water level. This may explain the low pH where regular drainage has led to a loss of base cations and acidification (Sollins et al., 1988). Without plant activity at this site, the mean age of respired C (2300 yr BP) likely reflects lakebed C sources.

Only this site emitted N₂O (mean of 0.66 ± 0.1 nmol m⁻² s⁻¹). PRS nutrient data indicated that available nitrate was 41.32 µg 10 cm⁻² month⁻¹ compared with negligible values at all the other sites. However total %N was not higher at this site compared to the others. The nitrate would have come from ammonium through nitrification, an aerobic process that also may release N₂O (Schmidt, 1982). However, since there is no vegetation to take up ammonium or nitrate, it may be consumed through denitrification under anaerobic conditions and result in the release of N₂O (Aulakh et al., 1992). According to Ma et al. (2007) when water content is below 60%, nitrification is dominant while beyond 60% it is denitrification. Even if Arctic soils are considered to be poor in nitrogen content, Chapin et al. (1995) explained that an increase in temperature of a tundra tussock environment is expected to increase nitrogen availability due to changes in mineralization, vegetation growth, biomass and NPP.

5.4 Variation of Carbon emissions compared to other sites

The variation in CO₂ fluxes at the study site was between 0.56 and 12.18 µmol m⁻² s⁻¹ (2.13 to 46.31 g CO₂ m⁻² day⁻¹) over the field season. This range is similar to ER measured on polygonal terrain located on the Tuktoyaktuk Coastlands with values between 0.07 and 46.79 g CO₂ m⁻² day⁻¹ (Martin et al., 2017). Martin et al. (2017) measured emission from three different polygonal terrain features; non-degraded which did not have ice wedges, moderate degraded ice wedges which had wet troughs and highly degraded ice wedges with melt ponds. These sites varied greatly

in wetness (moist soils to standing water) and in the types of plant species, which helps explain the broad range of CO₂ emissions measured. These ranges are higher than values found at other sites in the Arctic. Zona et al. (2010) found a range between 6.5 and 8.5 g CO₂ m⁻² day⁻¹ for three similar young (<50 yr BP) drained lake basins in the Arctic Coastal Plain of northern Alaska. This low variation may be explained by site selection, where the authors found similar sites to focus their analyses on temporal instead of spatial variation. Welker et al. (2000) found ER values between 62 and 1142 g CO₂ m⁻² season⁻¹ (~ 0.66 and 11.42 g CO₂ m⁻² day⁻¹) for the summer season of 1995/1996 at sites including moist tussock and dry heath tundra located in the Canadian Low Arctic. The two sites selected by the authors were 150 m apart and results of the whole season might be not representative for a daily comparison. Oberbauer et al. (2007) found values ranged from ~ 1.94 to 14.44 g CO₂ m⁻² day⁻¹ for wet meadow, moist meadow and dry heath within a low land site in the High Arctic (78°N) for the years 2000/2001. Even though there was variation in VWC among sites, the small variation in fluxes may be explained by undeveloped soils with a thin organic layer (0-15 cm) and mean July temperature of 5°C which are factors that can limit ER (Oberbauer et al., 2007).

The variation in CH₄ fluxes at the study was between -2.83 and 356.79 nmol m⁻² s⁻¹ (-3.9 to 494.53 mg CH₄ m⁻² day⁻¹) over the field season. Bartlett et al. (1992) found similar variation in CH₄ fluxes with values ranging between -2.1 to 426 mg CH₄ m⁻² day⁻¹ along three transects, each including a dry upland tundra and a wet meadow tundra at the bottom of the slope, located in the Yukon-Kuskokwim delta. Christensen et al. (2000) also found a similar range of CH₄ emissions at a heterogeneous tundra site with *Cassiope* heath, grassland, continuous fen, hummocky fen and *Salix arctica* snowbed, all located in a valley of the High Arctic (74°N). Values varied between -0.96 to 372 mg CH₄ m⁻² day⁻¹. Higher ranges were found by Martin et al. (2017), with values

between 0.12 to 1390 mg CH₄ m⁻² day⁻¹ from melt ponds of high-centered polygon near Inuvik and the other near Tuktoyaktuk, NT. However, these high values are only from the melt ponds with standing water 10-90 cm deep and no root vegetation in the pond centres. In contrast, Laurion et al. (2010) found a lower range of emission with values between 0.48 and 90.1 mg CH₄ m⁻² day⁻¹ for thaw ponds located in the Subarctic (55°N) and the High Arctic (73°N). The authors explain their low values by a lack of ebullition and even might be underestimated. A lower range of CH₄ emissions was also found by Morrissey and Livingston (1992) on the North Slope of Alaska, with values between 0 and 286.5 mg CH₄ m⁻² day⁻¹. These values come from a very broad range of sites (57) varying from well drained to wet tundra in order to represent both local and regional scales.

To summarize, CO₂ emissions were found to be influenced by above ground vegetation and temperature in expected ways. However, some factors that can influence C cycling such as presence and amount of OM and thaw depth did not correlate with emissions in this study. CH₄ emissions were positively influenced by greater VWC and presence of sedges, both common results throughout the literature. However, no significant relationship with CH₄ was found for any sites regarding thaw depth, OM content and soil temperature. These results highlight how spatial and temporal variability in emissions may be controlled by multiple factors and some relationships are likely masked or offset by others. N₂O emissions were negligible for every site except at the Bare Ground where constant flooding and very acidic conditions also resulted in limited plant growth and thus, cycling of older C compared to all the other sites. Despite a high spatial variability in C emissions throughout the basin, greenhouse gas emissions from this study site remained more or less within the range found at other sites across the Southern Arctic.

6 Chapter: Conclusion

Arctic environments are important C stores due to low mean annual temperatures and poorly drained soils (Belshe et al. 2013). The presence of continuous permafrost prevents active microbial decomposition of C at depth (Davidson and Janssens, 2006). However, C emissions in the Arctic have received attention in recent studies because Arctic ecosystems may shift from a sink to a source of C as the Arctic climate continues to change (Belshe et al. 2013). Higher temperatures could lead to an increase of active layer depth which would make available C stored and enhance microbial decomposition (Harden et al., 2012). Higher temperatures may affect ice rich permafrost, and as a result, create or drain thermokarst lakes, depending on topography (Plug et al., 2008). C emissions from drained lake basins of the Low Arctic and their controls are not well documented especially for recent basins (<50 years BP) (Billings et al., 1982; Zona et al., 2010).

This thesis examined the spatial variability of GHG emissions from a recently drained lake basin in the Low Arctic tundra environment in order to test the following hypotheses:

- 1) *Spatial variations in CO₂ and CH₄ emissions are correlated with vegetation abundance, soil organic C content, air and soil temperature, thaw depth, and soil moisture.*

The results of this study confirmed the hypothesis that vegetated collars have higher CO₂ emissions than unvegetated collars. There was not much difference between collars for CH₄ emissions throughout the basin except for Tundra and Wet Sedge 1&2 sites which had significantly higher fluxes at their vegetated collars and Low Willow had significantly higher emissions at the unvegetated collar. The rest of the sites were not affected by presence of vegetation. Most likely

the type of vegetation had a greater impact on CH₄ fluxes, where presence of sedges seemed to enhance CH₄ production and/or transport to the atmosphere. For CO₂ emissions, the main control on temporal and spatial variations in emissions was air temperature and presence of above ground vegetation, respectively. For CH₄ fluxes, the main control on spatial variations was volumetric water content and type of vegetation. Thaw depth, soil temperature, soil organic C and amount of biomass seemed to be important but not uniformly among the sites, possibly due to the high degree of spatial variability in vegetation and soil characteristics.

2) *CO₂ emissions largely reflect cycling of fresh (i.e., modern) plant residues.*

All but the Bare Ground site emitted relatively “modern” CO₂ while the presence of vegetation further increased $\Delta^{14}\text{C}$ values as recently sequestered CO₂ is respired by the plants and decomposed belowground. The Bare Ground site was an outlier with much older respired C due to minimal colonization of this site by vegetation due to periodic flooding and highly acidic conditions, all limiting C cycling.

3) *Compared to other research sites in the Arctic, CO₂ and CH₄ emissions at Illisarvik are similar in magnitude but have greater spatial variability.*

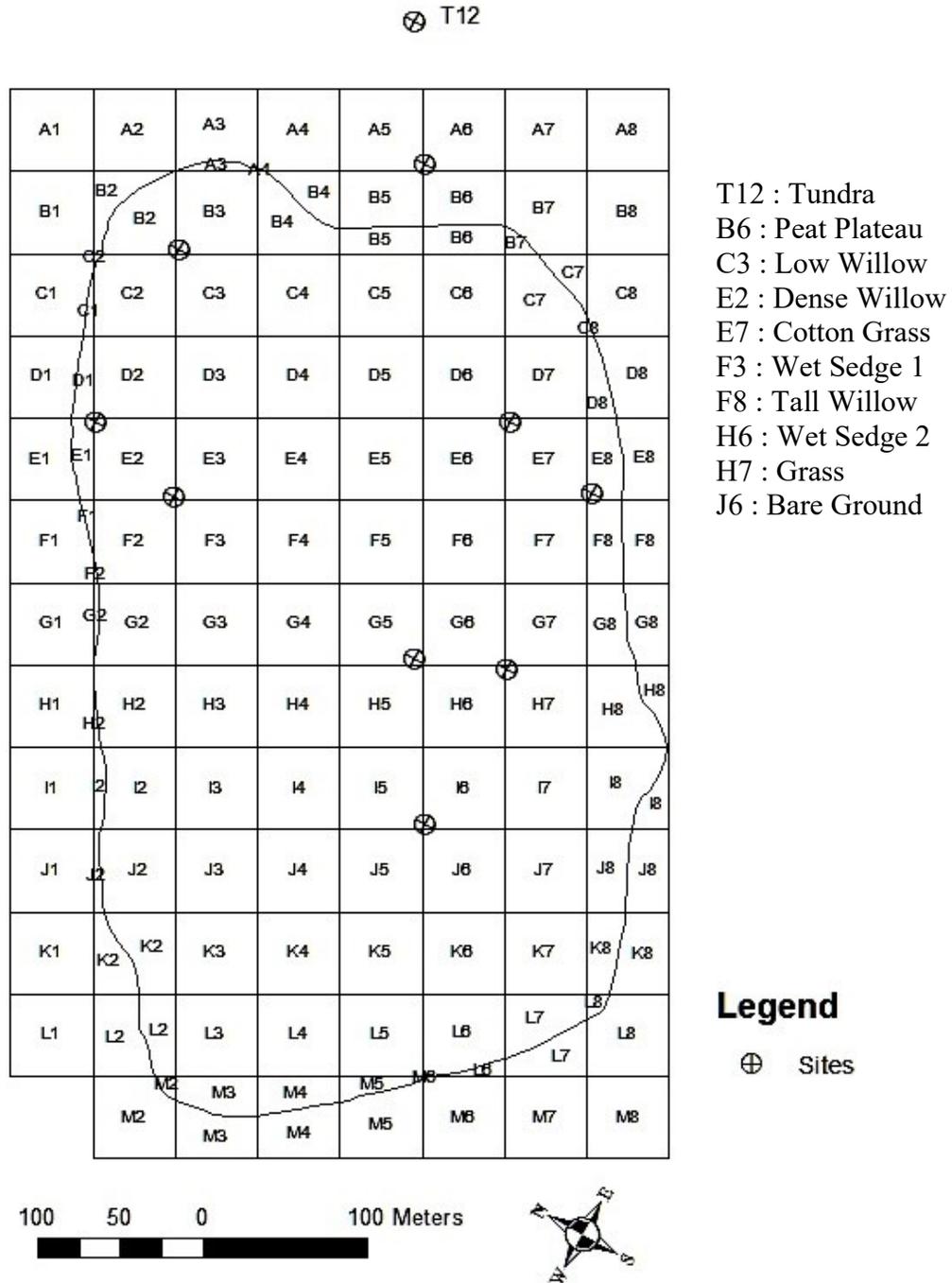
CO₂ emissions measured at Illisarvik were generally in the range found at other sites in the Arctic. Even with the high spatial variability of vegetation, soil moisture and soil characteristics, emissions of CH₄ were within the range expected based on other studies (Bartlett et al., 1992; Morrissey and Livingston, 1992; Christensen et al., 2000; Martin et al., 2017). Variation in CO₂ emissions was similar to a study by Martin et al. (2017) but were greater than at other sites in the Arctic (Bartlett et al., 1992; Christensen et al., 2000; Zona et al., 2010). Illisarvik is highly heterogeneous but does

not appear to be unique in terms of C or N₂O emissions since these emissions are still within or close to the range found at other sites located in the Arctic. Over time, Illisarvik should become a wet sedge moss-peatland, based on other similar sites and become less productive (Ovenden, 1986; Zona et al., 2010).

This study focused on one drained lake over the growing season of 2016. The goal of this study was to characterize spatial variability of vegetation and soil conditions, CO₂, CH₄ and N₂O emissions, and the variables that controlled them. A warming climate is expected to impact thermokarst processes and vegetation distribution throughout the Arctic, which might affect the Arctic C budget. Thaw depth, soil temperature and soil moisture are expected to control C emissions, however results from this study suggest that with a highly heterogeneous terrain, the expected relationships might not always be sufficient to explain temporal and spatial variations in greenhouse gas fluxes. There are multiple environmental variables interacting at the same time making a comparison between sites more complex. More studies are needed to cover the wide variety of Arctic landscapes and landforms to help further constrain estimates of CO₂ and CH₄ emission rates and the key environmental factors that influence them using process models or other scaling methods. However, regardless of the high spatial variability in vegetation and soil characteristics in a drained thermokarst lake basin, C emissions remained more or less within the range found at other non-basin Arctic sites. This implies that an increase in the rate of drainage of thermokarst lakes should not greatly impact estimates of respired CO₂ and CH₄ from the Arctic land surface.

Appendices

Appendix A. A grid of the Study site with sites locations. A solid line defines the border of the basin. Sites are located at the top left corner of a 50x50m grid.



Appendix B. Table B1. Radiocarbon $\Delta^{14}\text{C}$ (‰) and estimated age of respired CO_2 on August 5.

Sites	Collar	$\Delta^{14}\text{C}$ (‰)	Age (yr) BP
Ambient 1	N/A	18.37(7.26)	-210(57)
Ambient 2	N/A	12.50(10.05)	-164(80)
Tundra	Vegetated	-11.88(6.08)	32(49)
Tundra	Unvegetated	-3.91(12.69)	-33(102)
Peat Plateau	Vegetated	16.53(8.81)	-196 (70)
Peat Plateau	Unvegetated	-4.02(16.35)	-32(132)
Cotton Grass	Vegetated	-11.08(6.83)	25(56)
Cotton Grass	Unvegetated	-26.44(9.63)	151(79)
Tall Willow	Vegetated	-11.48(10.00)	29(81)
Tall Willow	Unvegetated	-22.45(20.10)	118(165)
Dense Willow	Vegetated	-12.30(8.28)	35(67)
Dense Willow	Unvegetated	-19.44(11.43)	94(94)
Low Willow	Vegetated	19.68(7.29)	-221(57)
Low Willow	Unvegetated	6.97(8.94)	-120 (71)
Grass	Vegetated	-18.29(7.74)	84(63)
Grass	Unvegetated	-27.98(8.89)	164 (73)
Wet Sedge 2	Vegetated	7.79(6.16)	-126 (49)
Wet Sedge 2	Unvegetated	4.96(10.02)	-104(80)
Wet Sedge 1	Vegetated	17.73(6.05)	-205(48)
Wet Sedge 1	Unvegetated	15.76(11.26)	-190 (89)
Bare Ground	Unvegetated	-260.23 (9.42)	2357(102)

Appendix C. Table C1. PRS probe results for the field season per site. Units are in $\mu\text{g } 10\text{cm}^{-2} \text{ burial month}^{-1}$.

Site	NO ₃ -N	NH ₄ -N	Ca	Mg	K	P	Fe	Mn	Cu	Zn	B	S	Pb	Al	Cd	Total N
Tundra	1.48	1.08	853.68	375.44	37.71	1.58	9.32	0.45	0.09	1.1	0	10.89	0	2.77	0	2.56
Peat Plateau	1.9	0.92	1194.49	459.98	11.18	1.25	37.24	1.57	0.02	0.05	0.29	19.08	0.13	9.31	0.05	2.82
Low Willow	0	2.96	2353.34	547.19	12.57	4.82	854.81	30.26	0.92	2.33	3.26	77.54	0.96	12.71	0	2.96
Dense Willow	0.72	1.62	1604.2	462.06	13.63	5.7	291.08	13.33	1.32	1.6	1.02	283.96	0.44	10.09	0	2.34
Tall Willow	1.72	5.68	1641.23	426.9	30.56	1.32	59.16	2.66	0.33	0.9	0.58	52.03	0.24	10.93	0.02	7.4
Cotton Grass	1.82	0.8	2006.44	567.46	5.64	0.19	39.4	2.8	0.27	1.39	0.24	1315.14	0	7.12	0.01	2.62
Wet Sedge 1	0.62	5.98	1795.08	432.69	5.71	4.58	801.34	18.77	0	0.37	2.95	140.24	0	11.6	0.04	6.6
Wet Sedge 2	0.98	6.46	1991.07	539.11	7.29	2.56	276.22	30.57	0.01	0.34	1.04	361.32	0	11.58	0.01	7.44
Grass	0.7	1.8	1978.9	758.25	13.3	2.29	207.44	20.51	0.21	0.17	0.74	1112.05	0.05	16	0	2.5
Bare Ground	41.32	1.58	1656.99	603.3	23.86	1.19	36.33	21.27	0.58	1.84	0.59	1127.53	0.05	27.99	0.01	42.9

Appendix D. Table D1. Mean values of carbonates per sites. Units are in % with standard error in parenthesis.

Sites	Carbonates (%) 0-30 cm
Tundra	1.45 (0.63)
Peat Plateau	4.98 (1.97)
Low Willow	0.96 (0.28)
Dense Willow	N/A
Tall Willow	1.51 (0.57)
Cotton Grass	0.77 (0.16)
Wet Sedge 1	2.77 (0.84)
Wet Sedge 2	1.03 (0.46)
Grass	N/A
Bare Ground	1.95 (0.2)

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