**Assessing Carbon Effluxes and Decomposition Rates of Wetland
Reclamation Trials, Fort McMurray, Alberta**

by

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ABSTRACT

Carbon dioxide and methane fluxes were measured from wetland reclamation trials in an effort towards evaluating their potential for sustainable development through their potential for organic carbon accumulation. The treatments included live transplanted peat and stockpiled peat mixed with mineral soil at varying depths placed either in winter or summer. Fluxes were concurrently measured in a nearby natural peatland June through August 2010. Soil samples were incubated in the laboratory to assess the potential decomposability of these treatments. Emissions of CO₂ and CH₄ were greatest in the live treatment both in the *in-vivo* and *in-vitro* condition. These results were attributed in part to a greater quantity of labile soil organic matter that had high decomposability. CO₂ emissions were not particularly high compared to emissions at natural peatlands although this may have been due to the high water table maintained at plots. If the water table drops, higher CO₂ emissions are possible.

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List of Abbreviations

C		Carbon
CO ₂		Carbon dioxide
CH ₄		Methane
SOC		Soil Organic Carbon
C _{min}		Mineralized Carbon
SOC		Soil Organic Carbon
SOM		Soil Organic Matter
BD	g cm ⁻³	BulkDensity
φ	V _v V _T ⁻¹	Soil Porosity
pH		Acidity measured
g		Grams
s		Second
m		Meter
d		Day
μmol	10 ⁻⁶ mol	Micromole
nmol	10 ⁻⁹ mol	Nanomole
°C		Temperature in degrees Celsius
N		Nitrogen
P		Phosphorus
K		Potassium
Ca		Calcium
Mg		Magnesium
Fe		Iron
Mn		Manganese
Cu		Copper
Zn		Zinc
NH ₄ ⁺		Ammonium
NO ₃ ⁻		Nitrate
S		Sulphur
B		Boron
Pb		Lead
Al		Aluminum
Cd		Cadmium
p		Probability value
R ²		Correlation coefficient
df		Degree of freedom

1. INTRODUCTION

Along with industrialization, the global demand for fuel is imposing great pressure on terrestrial ecosystems and this has precipitated many environmental concerns. One concern is the loss of natural ecosystems with their potential to act as sinks for greenhouse gases (GHGs) such as carbon dioxide (CO₂). As a result of anthropogenic activities such as deforestation and topsoil disturbance caused by land-use changes, the natural ecosystem's potential to act as a sink has been progressively reduced.

Peat forming wetlands cover approximately 5% of the global surface area (Matthews & Fung, 1987) and 14% of the landscape in Canada (NWWG, 1988) and are unique ecosystems with a high capability for carbon sequestration (Roulet, 2000). Northern Canada is covered with approximately 1.24×10^6 km² wetlands in boreal forest ecosystems (Joosten & Clarke, 2002). In Canada, these ecosystems store about half of the country's soil carbon stocks (Tarnocai, 1998). Globally, peatlands store approximately 455 Pg of carbon which is about one-third of the global soil carbon pool (Gorham, 1991; Yu et al., 2001).

The surface area covered with boreal forests in Alberta is about 381 000 km² while Canada's oil sands deposits in this province lie underneath approximately 140 200 km² of this immense natural ecosystem. Approximately 18% of Alberta's land consists of wetlands (Vitt et al., 1996), among which 90.4% are peatlands (bogs and fens) within the boreal forest region of northern Alberta (OSWWG, 2000). At present, surface mining in the oil sands covers about 4800 km². Surface mining involves the clear

cutting of boreal landscape which leads to irreversible disturbance of peatlands. Environmental protection policies require oil sands producers to reclaim disturbed lands to “equivalent capability, which is a sustainable landscape with productivity that is equal or greater than prior to oil sands development” including the ability to support native vegetation and wildlife (CAPP, 2000).

Consequently, oil sands producers are developing wetland reclamation strategies for disturbed sites. Part of successful peatland reclamation would involve these ecosystems accumulating organic carbon in the soil (as peat) as a result of slow rates of decomposition.

A better understanding of carbon cycling within reclaimed wetlands is important to effectively implement reclamation strategies. Thus, this study has focused on identifying which treatments can slow decomposition processes and their associated emissions of carbon dioxide (CO₂) and methane (CH₄) while supporting the growth of peatland vegetation.

Experimental peatland reclamation trials were established by Syncrude Canada Ltd. near Fort McMurray, AB in 2008-09. The treatments included live transplanted peat and stockpiled peat mixed with mineral soil placed in 15, 50 and 100 cm layers either in winter or summer.

To improve the chances for successful peatland reclamation, this study uses field and laboratory measurements of CO₂ and CH₄ exchange to address the following two major questions:

- i. Compared to a nearby natural peatland, which treatment has the same or better potential for carbon sequestration as a result of slow rates of decomposition?

ii. What are the physical/chemical factors that control variations in carbon effluxes (CO_2 and CH_4) in the experimental treatments in question, and a local natural peatland?

These questions are further addressed with the following hypotheses:

i. Plots with transplanted peat (live treatment) are expected to have greater CO_2 and CH_4 emissions in the field and in laboratory incubations than plots with stockpiled peat mixed with mineral soil (stockpiled treatment) due to greater soil organic matter (SOM) content. Further, because the SOM in the stockpiled peat has likely undergone significant decomposition while stored, it may be more recalcitrant than recently transplanted peat which should have some component of fresh plant material as well as humified peat.

ii. The natural treatment will not receive additional watering and is expected to have lower soil moisture than experimental treatments, resulting in greater aerobic respiration rates and the potential for oxidation of CH_4 resulting in negligible CH_4 emissions and greater CO_2 emissions compared to the experimental treatments.

iii. The contribution of root autotrophic respiration to total soil respiration is expected to be greatest in treatments with healthier and greater vegetation densities which are expected for the live and natural treatments. Thus, when roots are excluded, autotrophic respiration will decrease most in live and natural treatments. It is also expected that when roots are excluded, there will be an increase in available nutrients and an increase in fresh plant material (from dead and dying roots), heterotrophic respiration may increase. Thus, there may not be a significant effect of root exclusion on CO_2 emissions.

iv. For treatments with thicker peat (50 cm and 100 cm vs. 15 cm), higher decomposition rates would be expected due to the larger amount of SOM subject to decomposition.

v. In all treatments, both CO₂ and CH₄ effluxes are expected to increase with higher temperatures if moisture conditions remain unchanged. Soil temperature is expected to be most variable in the stockpiled treatment due to lower albedo and greater thermal admittance of the stockpiled substrate when compared with live or natural treatments. Thus greater temporal variations in CO₂ and CH₄ emissions are expected in the stockpiled treatment.

vi. For CH₄ production, conditions must be anoxic with very low redox conditions. With regular additions of aerated water to the experimental treatments, these anoxic conditions may not occur allowing for relatively high rates of CO₂ efflux and low CH₄ emissions despite high water levels.

2. BACKGROUND AND LITERATURE REVIEW

2.1. Wetland definition, classifications and functions

According to National Wetlands Working Group (NWWG, 1997), a wetland is defined as land that is either permanently or temporarily saturated and distinguished by poorly drained soils, hydrophytic vegetation and biological activity that is adapted to and characteristic of a wet environment.

Wetlands can be classified based on different factors such as soil structure, hydrology, vegetation, water quality and/or a combination of these factors. Natural wetlands in Canada have been classified into five distinct groups: bogs, fens, swamps, marshes, and shallow open water wetlands (NWWG, 1997). Among those, shallow open-water wetlands, marshes, and swamps are subdivisions of non-peat forming wetlands (with < 40 cm of accumulated organic material) and bogs, fens, and some swamps are subdivisions of peat forming wetlands (with > 40 cm of accumulated organic material) known as peatlands (Vitt et al., 1996). Based on hydrologic characteristics, wetlands are further classified as either mineratrophic or ombrotrophic (Sjors, 1950). Fens and swamps are mineratrophic, which means that they receive water from ground/surface water and from atmospheric sources. However, bogs are ombrotrophic, which means that their only source of water is from direct precipitation (NWWG, 1997).

Peatlands provide a variety of ecosystem services such as the storage of water, filtration of precipitation while it moves into the groundwater, providing habitat for different species (Mitsch & Gosslink, 2000) and sequestering large amounts of

carbon through a small but persistent imbalance between primary productivity and decomposition (Yu et al., 2001).

2.2. Wetland reclamation, restoration and constructed wetlands

Using the Alberta Environment regulatory definition (CAPP, 2007), wetland reclamation is defined as the "creation of wetlands on disturbed land where they did not formerly exist or where their previous form has been entirely lost" and wetland restoration is "a process of returning wetland function of a remnant wetland site, as it was before disturbance". The Oil Sands Wetlands Working Group (2000) has identified five different wetland types in its reclamation plans, each following specific goals. These include altered wetlands, opportunistic wetlands, constructed wetlands, vegetated watercourses, and littoral zones of lakes.

Altered wetlands refer to landscape features that are not removed by mining but are affected by post-mining activities (e.g. dewatering, drainage) and therefore need restoration measures. Opportunistic wetlands are unplanned wetlands that occur from depressions (e.g. increase in water tables or slow drainage) and consequently, provide some of the wetlands' functions. Vegetated watercourses are designed as channels for transferring water to and from wetlands and provide riparian habitat (e.g., travel corridors, habitat connectivity). Littoral zones are designed along the shores of end pit lakes, to provide shoreline protection and some of the wetlands' functions. Finally, constructed wetlands are landscape features that are created to provide wetlands' primary functions such as flood control, water quality improvement or habitat for wildlife populations (OSWWG, 2000). Constructed

wetlands, such as the wetland reclamation trials discussed in this thesis, are created by anthropogenic modification of the substrates and/or hydrology to form a hydrophyte community and hydric soil.

In wetland reclamation, how substrates are acquired, placed and amended are all of great importance. The substrate, which becomes the upper layer of soil, not only supports the rooting system of wetland vegetation but it is also responsible for water flow, gas fluxes, aeration, nutrient supplies and for microbial activity. Substrate acquisition depends on the reclamation goal for the constructed wetland. It can be composed of peat, stockpiled mineral/peat, overburden, hydric soils salvaged from other wetlands, or some other combination. As the physical and chemical properties of substrates (i.e. their pH, salinity, or clay content; Danielson et al., 1983) vary in different reclamation efforts and are subject to change with time, it takes a long period for newly reclaimed wetlands to achieve the desired outcome (OSWWG, 2000). Sometimes new conditions will also be imposed (i.e. a saline condition created by using oil sands process waters) that require adaptation time.

As any reclamation activity is expensive, especially at large scales, careful and precise studies should be conducted to determine the optimal substrates, hydrology and amendment regimes, based on geology, climatology, and ecology of the region. Further, as hydric soil, hydrophytes and waterlogged condition are the key characteristics of wetland ecosystems, these characteristics should be considered when evaluating the functionality of constructed wetlands (Sistani et al., 1999; Stolt et al., 2000).

2.3. Carbon cycling in peatlands

Carbon accumulation in peatlands is the result of an imbalance between carbon gained through photosynthesis and its loss through ecosystem respiration (both autotrophic and heterotrophic respiration), resulting in exchanges of CO₂, CH₄ and dissolved organic and inorganic carbon (Batjes, 1996). The difference between photosynthesis and ecosystem respiration is known as net ecosystem exchange of CO₂ and it represents the net amount of CO₂ taken-up or released by the ecosystem.

Although there are many different factors that control carbon exchange processes, generally they can be grouped into three categories: biological (e.g. plant species and physiology, microbial community, etc.), physical (e.g. water level, light, moisture, temperature, etc.) and chemical (e.g. oxygen availability, nutrients, etc.).

Northern peatlands, with slow but steady net carbon sequestration are unique natural carbon sinks (Roulet et al., 2007). Carbon sequestration in peatlands is a result of low rates of decomposition due to cold temperatures, waterlogged and anoxic soil conditions with low redox potentials, and plants that are resistant to decomposition (Moore & Basiliko, 2006). Large porous structures within the surface peat (acrotelm) provide its high water storage capacity (Boelter, 1968) and low redox potentials suppress respiration and effectively sequester CO₂ (McNeil & Waddington, 2003).

Anthropogenic disturbance of peatlands (e.g. land use change, mining, peat harvesting, etc.) terminate peat formation (Rodhe & Svensson, 1995) and can transform peatlands into sites that are net sources of CO₂ (Waddington et al., 2002).

These disturbances can also effectively reduce CH₄ emissions (Tuittila et al., 2000). This switch from a carbon sink to source is caused by a loss of vegetation adapted to saturated conditions, decrease in water level and volumetric water content, changes to the soil water retention capacity, and changes in microbial activities related to new aerobic conditions and to increased substrate availability (Waddington et al., 2002).

In disturbed wetlands, carbon sink capacity can be restored by improving and re-establishing the factors that characterize wetlands. Studies have shown that after restoration of some Canadian peatlands, net CO₂ losses were reduced as volumetric soil moisture contents were increased resulting in reduced rates of aerobic microbial decomposition due to lack of oxygen (Waddington, 2000). In Finland, restored peatlands became a net sink for CO₂ after 3 years by controlling water level and vegetation composition (Komulainen & Tuittila, 1999).

Although reclamation and restoration techniques can bring disturbed peatlands towards a status as a net sink of atmospheric CO₂ (Greenwood, 2005), because of the water logged characteristic of peatland soil, they typically remain a net source of CH₄ (Waddington & Day, 2007).

Decomposition of SOM in wetlands depends on microbial activity which is a function of the microbial community, substrate quality and quantity, nutrient availability, and environmental conditions such as temperature and water content. Environmental conditions and nutrient availability also have a great effect on the productivity and species composition of the vegetation community (Wheeler &

Proctor, 2000), which, in turn, determines the quality and quantity of the substrates as a nutrition source for microorganisms.

Soil organic matter consists of about 50% carbon by dry weight and smaller amounts of O, H, N, P and S. Decomposition depends on both quantity and quality of the SOM. In peatlands, decomposition rates are greatest for leaf and root litter from herbs and sedges, followed by arboreal litter and finally, with the slowest rates, *Sphagnum* mosses (Scheffer et al., 2001; Thormann et al., 2001). This classification is based on the chemical structure of organic molecules. The heavier and more complex the C molecule the slower the decomposition rate (Wagner & Wolf, 1998).

Also, the decomposition rate of SOM greatly depends on its C:N ratio. Plant tissues vary widely in their C:N ratios. Organic materials with low C:N ratios tend to decompose faster (Enriquez et al., 1993). However, studies have shown that nitrogen concentration controls the decomposition only when labile SOM is available to support microbial needs (Haynes, 1986). During break down of SOM by microorganisms, N, P and S and other nutrients will be mineralized (Wardle, 2002). These nutrients will be available for plant uptake and microbial immobilization. As plants and microorganisms both use the same pool of N resources (Tilman, 1982), there will be a continuous competition for N and because N is a limited resource in most ecosystems, N becomes a limiting factor for microorganism and plant growth (Kaye & Hart, 1997).

The C:N ratio of SOM usually decreases as decomposition proceeds. Fresh plant litter begins with a high C:N ratio and with the consumption of labile parts of organic compounds, C is lost as CO₂ while N participates in mineralization and immobilization processes. SOM quality decreases with decomposition (or humification). Microbes preferentially consume the labile substrates, and leave the recalcitrant compounds while microbes die and recalcitrant components in their cell walls (e.g. chitin) increase the recalcitrant humic compounds in the soil (Berg & Staaf, 1980; Melillo et al., 1982).

C:N ratio describes the size and distribution of C and N pools (Tateno & Chapin, 1997). Most experiments have shown that although C:N ratio or the nitrogen concentration of SOM are very important factors for decomposition, the C:N ratio of SOM cannot directly influence the decomposition rates (Haynes, 1986; Prescott et al., 1999; Hobbie & Vitousek, 2000). The quality of SOM is a better predictor of decomposition rate (Berg & Staaf, 1980; Melillo et al., 1982).

Slow decomposition can occur for other reasons. For example slow decomposition of *Sphagnum* moss is a result of an increase in acidity due to organic acids produced as *Sphagnum* decays (Johnson & Damman, 1993). Generally acidity results in both poor vegetation and poor microbial biomass production which leads to a decrease in the input of labile SOM to the soil and microbial respiration, respectively (Primavesi, 1984). In peat, the break-down of moss structures also decreases the number and size of pores and lowers the hydraulic conductivity (Boelter, 1969; Clymo, 1984) and, consequently, the new anoxic condition slows the decomposition rate (Johnson et al., 1990).

Although the role of microorganisms and the quality of peat substrate after reclamation are important factors influencing decomposition and emission of CO₂ and CH₄ in constructed wetlands these mechanisms are poorly understood (Glatzel et al., 2004). Based on field studies, Glatzel et al. (2004) and Waddington et al. (2001) reported that peat in mined peatlands have lower CO₂ production potential than nearby natural sites, which can be the result of exploitation and removal of peat, clear cutting of vegetation and suppression in microbial activities related to limited nutrient availability (Wind-Mulder et al., 1996). Studies have shown that soils in created wetlands as part of reclaimed wetlands usually have higher bulk density and lower OM compared to the natural wetlands (Bishel-Machung et al., 1996; Shaffer & Ernst, 1999; Stolt et al., 2000). Poor vegetation growth in reclaimed wetlands can lead to reduced litter quantity and with less labile SOM available to microbes (Hunter & Faulkner, 2001), which can effectively decrease the microbial biomass (Duncan & Groffman, 1994).

Most available nutrients within peat are derived from decomposition of fresh plant and microbial matter and can also be introduced by adding intermediate substrates usually known as amendments (Segers, 1998; Whalen, 2005; Lai, 2009). Adding amendments is very common in promoting reclaimed wetland's function. As an example an increase in electron acceptors like nitrate (NO₃⁻) can decrease CH₄ emissions as a result of suppressed methanogenesis while an increase in ammonium (NH₄⁺) inhibits CH₄ oxidation and increases CH₄ emissions (Bubier, 1995).

The few studies on reclaimed wetlands showed that SOM and some other soil characteristics such as porosity are not increasing with time as expected (Bishel-Machung et al., 1996; Shaffer & Ernst, 1999). Thus, it has been suggested that amendments such as leaf compost, sewage sludge, and bio-solid materials is the best way to help accelerate the development of wetland functions in constructed wetlands such as freshwater marshes (Stauffer & Brooks, 1997; Bailey et al., 2007). As an example, organic amendments used in a reclaimed marsh had a significant effect by increasing soil water content and nitrate availability (Stauffer & Brooks, 1997). In another case, organic amendments applied to a reclaimed fresh-water marsh led to an increase in microbial activity (Duncan & Groffman, 1994). Some peatland reclamation trials, such as those described in this thesis and others (e.g. Wieder et al., 2011), involve laying down peat or mineral soil mixed with peat from various sources with the goal of more quickly achieving hydric soils that will support peatland vegetation.

As plants provide the organic material that becomes peat and the substrates for microbial activity, proper re-vegetation of reclaimed peatlands is imperative. Studies have shown that re-vegetation and elevation of the water table in reclaimed wetlands causes a noticeable increase in fresh SOM within soils (Brake et al., 1999) and enhanced CO₂ sequestration (Weider et al., 2010). These alterations can effectively influence nutrient balance and improve physicochemical properties of the peat (De Mars & Wessin, 1999; Baum et al., 2003), which leads to an increase in size and activity of microbial populations and, consequently, to carbon transformations (Francez et al., 2000).

The most important environmental conditions that limit decomposition in peatlands are oxygen availability, temperature, and acidity. Accordingly, laboratory experiments have shown that CO₂ emissions generally increase after lowering water level (Blodau et al., 2004), increasing pH (Chapman & Thurlow, 1998) and increasing temperature (Bergman et al., 1999).

Temperature has a significant effect on soil microbial activities and as a result CO₂ and CH₄ emissions from the soil. Generally, as temperatures decrease, decomposition slows down and vice versa (Moore et al., 1999; Trofymow et al., 2002). In the field, CO₂ and CH₄ emissions from peatland soils are often observed to increase exponentially with temperature (Alexander, 1961; Lloyd & Taylor, 1994; Moosavi et al., 1996; Conen et al., 2008). An increase in CO₂ and CH₄ emissions at higher temperatures are due to the stimulation of microbial respiration (Winkler & Cherry, 1996) and the increase in rates of chemical reactions at high temperatures (McKenzie et al., 1998).

However, this increase has a limit and microbial respiration and biological oxidation systems will be inactivated at sufficiently high temperatures (Lloyd & Taylor, 1994). Further, CH₄ fluxes may decrease with increasing temperature when associated with a decrease in water table depth and increased aeration in the peat as a result of enhanced evapotranspiration (Moore & Roulet, 1993; Roulet et al., 1993). Meanwhile, any factor, such as precipitation that leads to elevation of the water table, increases CH₄ emissions by providing anoxic zones. As natural ecosystems are always experiencing a combination of environmental factors, the

carbon emissions would depend on the overall effect of the changes on CO₂ and CH₄ production and CH₄ oxidation processes (e.g. Bohn et al., 2007).

In peatlands the position of the water table could be above, at, or below the peat depending on time and space. These permanent or temporary variations have a direct effect on both oxygen availability and soil moisture and provide a moisture-aeration regime, which not only depends on the water table, but also on pore structure, ratio of total pore spaces filled with water versus air, and the oxygen content of the water (Rydin & Jeglum, 2006). Macroporosity in peats lead to rapid infiltration, however, the *Sphagnum* mosses with dead hyaline cells that do not have the capability of water transportation hold the water between the leaves, stems of mosses by capillary forces and provide additional water storage (Rydin & Jeglum, 2006). This stored water will move along the capillary gradient due to matric potential, capillary fringe and the atmospheric vapour pressure deficit (Boelter, 1969; Clymo, 1984; Rydin & Jeglum, 2006).

Aerobic decomposition and CO₂ production rates can decrease when soil moisture drops below 50% or increases above 100% of the dry mass (Haynes, 1986). In peatlands, soil matric potential has important effects on soil microbial activity through controlling nutrient supplied by diffusion (Killham, 1994; Stark & Firestone, 1995). Low matric potential limits both nutrient availability and water accessibility to microbes. In saturated conditions however, lack of aerated pore space leads to anoxic conditions and accordingly inhibits aerobic decomposition (McNeil & Waddington, 2003). Note that short-term, infrequent fluctuations between oxic and anoxic conditions can strongly increase decomposition rates as a

result of an increase in dissolved organic carbon (DOC) produced during aerobic decomposition followed by diffusion of nutrients and their accessibility to microbes in saturated peat (Moore & Dalva, 2001). However, frequent moisture fluctuations reduce microbial populations and result in a decline in decomposition (Clein & Schimel, 1994). This occurs as rewetting dry peat can lead to high concentrations of synthesized osmotic metabolites by microbes and create a gradient causing many microbial cells to burst and die.

Soil redox potential, which indicates oxidation potential and availability of terminal electron acceptors (Lafleur, 2009), must be below -200 to -400 mV (Zehnder, 1978) for methanogens to have a priority to other decomposers. Methanogens are classified into hydrogenotrophs and acetotrophs using hydrogen and acetate for CO₂ reduction and CH₄ production, respectively. The contribution of each group in methanogenesis depends on temperature and substrate availability (Westermann, 1993). Usually acetotrophic methanogens contribute in 85- 90 % of the CH₄ production at temperatures between 10 °C - 15 °C and as the temperature rises, the activity of hydrogenotrophic methanogens increase (Avery et al., 1999). Also as the vegetation coverage and consequently SOM increase, the contribution of acetotrophic methanogens rise. In recent carbon isotope studies in northern peatlands, results have shown that the acetate pathway is dominant during the summer as a result of increased acetate levels (Avery et al, 1999; Chasar et al, 2000).

Not all the CH₄ produced in the peatland is emitted to the atmosphere. A considerable amount of it may be consumed by methanotrophic bacteria in a CH₄

oxidation zone (Hanson & Hanson, 1996; Segers, 1998). Usually, the potential for CH₄ oxidation is larger than the potential for CH₄ production (Segers, 1998). Water table level typically has a direct negative relationship with the CH₄ oxidation zone. Overall, the thicker the CH₄ production zone, the thinner the CH₄ oxidation zone and vice versa (Whalen, 2005).

Changes in water table level will affect the availability of substrate for aerobic and anaerobic decomposition. Whichever process consumes the mineralized carbon from the soil substrate pool most effectively reduces the substrate availability for the other one.

CH₄ emission to the atmosphere is through the three main pathways of diffusion, ebullition and plants. CH₄ emissions through diffusion is usually slow in comparison to other two pathways (Kiene, 1991; Lai, 2009) although it plays an important role in total emissions to the atmosphere (Whalen, 2005). Ebullition or the sudden release of CH₄ in the form of bubbles occurs in water saturated peat layers mostly as a result of changes in water level (Strack et al., 2005), barometric pressure (Kellner et al., 2004; Comas et al., 2008), temperature (Beckmann et al., 2004), mechanical disturbance (Fechner-Levy & Hemond, 1996) or during spring thaw via its release under the ice CH₄ (Moore & Knowles, 1990, Hargreaves et al., 2001). These factors cause the partial pressure of CH₄ in the water to exceed the hydrostatic pressure of the water column and ebullition occurs (Schiitz & Seiler, 1989; Chanton & Dacey, 1991). Plants also provide a pathway for CH₄ through aerenchymous tissue. Aerenchymous tissue which is adapted to waterlogged soils transports oxygen into the root zone for oxidation of CH₄ (Chanton et al., 1992),

through the same route, CH₄ is transported into the atmosphere. The contribution of aerenchymous species to total CH₄ emissions has been estimated to be ~ 25-97% (Whalen, 2005).

In studies from reclaimed peatlands, the rates of CH₄ emissions are often low. These low rates were not related to less CH₄ production but to higher oxidation rates because of well aerated surface peat and also less vegetation with aerenchymous tissues having the ability to transport CH₄ to the atmosphere and bypass methanotrophs (Magnusson, 1993; Moore & Dalva, 1997; Francez et al., 2000).

In peatlands, decomposition and CO₂ production usually decreases with soil depth due to both greater recalcitrance of substrates (e.g. lower quality substrates) and to less favorable environmental conditions (e.g. high moisture and low temperature) (Belyea, 1996). In addition, the soil's physical characteristics such as its bulk density and porosity are not favorable for decomposition at depth. Deeper soils have higher bulk density and lower porosity which limits the rates of gas diffusion limiting microbial development and growth (Groot, 1998; Holden et al., 2004).

Peat disturbance often leaves behind these deeper peat soils with lower decomposability. However, some studies have shown that reclamation with appropriate re-vegetation improves the quality and quantity of SOM at the top layers and results in higher decomposition rates in comparison to sites with poor vegetation or vegetation with recalcitrant compounds (Glatzel et al., 2004). Similarly, Waddington et al. (2003) showed that after peatland reclamation, these

reclaimed sites will remain sources of C for years and suggested that the source of these emissions were new vegetation and straw mulch used as amendments.

As described here, a great number of factors are important in influencing decomposition rates in peatland soils, and thus are important to consider when developing strategies to construct peatlands where carbon will accumulate as peat.

3. MATERIALS AND METHODS

3.1. The study area

The Athabasca oil sand deposits in northeastern Alberta (Figure 3.1) lie within the Boreal Plains ecozone of Canada (Devito et al., 2005). The forests in this region include aspen (*Populus tremuloides* Michx.), poplar (*P. balsamifera* L.), spruce (*Picea glauca* (Moench) Voss, black spruce (*Picea mariana* (Mill.) Britton, Sterns & Poggenb), jack pine (*Pinus banksiana* Lamb), balsam fir (*Abies balsamea* (L.) P. Mill) and tamarack (*Larix laricina* (Du Roi) Koch) (Rowe, 1972). The area is also habitat for a large number of fauna, including nearly 236 bird species with 66 species of waterfowl, more than 43 mammal species (Westworth, 1993) and many different amphibians and reptiles, fish, insects, arachnids, etc. that are directly or indirectly dependent on the boreal ecosystem (OSWWG, 2000).

Natural soils in the oil sands are classified as orthic grey luvisols, organic, and organic cryosols, eluviated dystic, eluviated eutric brunisols, brunisolic gray luvisols, and gleyosolic soils (Syncrude Canada Ltd., 1984). In addition, moisture regimes in these soils vary from xeric to subhydric (The Oil Sands Vegetation Reclamation Committee, 1998). Thus the region supports a variety of landscapes from bogs and fens to forests.

The climate in this region is boreal cold temperate (Mulligan & Gignac, 2001). As recorded at the Environment Canada weather station located at the Fort McMurray Airport (56° 39'N, 111° 13'W), the mean daily temperature ranges from -18.8 °C in January to +16.8 °C in July. Only 2 months of the year are frost-free and the growing season is relatively short. The mean annual precipitation is 456

mm with 342 mm of rain and 156 cm of snow (Environment Canada, Climate Normal 1971-2000).

The oil sands consist of fine grained quartzitic bitumenous sand and sandstone originating from delta deposits from an ancient tropical sea, consolidated by glacier formations (Stolte et al., 2000), saline sodic Cretaceous shale known as Clearwater shale, glacier sediments and recent sediments (Sincruide Canada Ltd., 1978).

Figure 3.1. Location of the three main oil sands deposits in Alberta, Canada. Source: [Einstein, 2006] (http://en.wikipedia.org/wiki/File:Athabasca_Oil_Sands_map.png).

It has been estimated that the mined area near Fort McMurray will exceed 1406 km² by the year 2023 (Cooper & Lee, 2003) and result in the destruction of a considerable area of boreal peatlands. According to government policies, oil sands producers are required to return these disturbed lands to equivalent (or greater) capabilities of what existed prior to oil sands development. Consequently, research on how to develop effective and sustainable techniques to create peatlands is ongoing. During the summer and winter of 2008-09 Syncrude Canada constructed an experimental site to examine its wetland reclamation strategies for the Fort McMurray, Alberta area.

This study was carried out on a site about 40 km NNW of Fort McMurray, Alberta (57° 03' N, 111° 39' W), near the SW end of the Mildred Lake basin (Figure 3.2). Large, U-shaped 20 m × 10 m "cells" (plots) were excavated within which 28 different experimental peatland reclamation treatments were established (Figure 3.3). Treatments involved using either stockpiled mineral/peat soil mixtures (stockpiled) or live peat transplants (live) from a local peatland ecosystem. The three depths of substrate were 15 cm, 50 cm, and 100 cm, and they were either applied with or without compaction and in either winter or summer. The vegetation in live peat transplants were transferred to the plots along with the peat directly from the local peatland and the vegetation in stockpiled plots were peatland plant species grown from seed in greenhouses and planted by hand (see Table 3.1 for a list of the species found within the treatments). To minimize disturbance during study of the plots, boardwalks were built across the plots.

Figure 3.2. Location of the U-Shaped Cells in Syncrude Canada Ltd. "Mildred Lake" Settling Basin, Fort McMurray, Alberta, Canada (Google Earth Pro, 2010).

Figure 3.3. The U-shaped cells ("plots") at the experimental site (Google Earth, 2010).

Table 3.1. Vegetation in live transplanted peat and stockpiled mineral/peat plots in spring-summer, 2010 (Bloise, 2010).

Live transplanted peat			Stockpiled mineral/peat		
Achillis millifolia	Funaria hygrometrica	Salix	Achillis milifolia	Chenopodium album	Poa compressa
Agropyron interior	Hawksbeard	Salix serrissima	Agropyron enlongatum	Cicuta maculata	Poa interior
Agropyron smithii	Hordeum jubatum	Scirpus microcarpus	Agropyron smithii	Clematis occidentalis	Polygonum lapathifolium
Agrostis scabra	Hylocomium splendens	Scirpus validus	Agropyron trachycaulum	Corydalis aurea	Populus tremuloides
Algae	Juncus bufonius	Showy aster	Agrostis scabra	Crepis tectorum	Potentilla norvegica
Beckmannia	Kalmia polifolia	Smilicina trifolia	Alopecurus aequatis	Epilobium angustifolium	Ranunculus sceleratus
Betula glandulosa	Kobresia	Sonchus arvensis	Amaranthus retroflexus	Epilobium ciliatum	Rhianthus borealis
Birdsfoot trefoil	lamb's quarters	Sphagnum fuscum	Aster conspicuus	Epilobium glandulosum	Rosa acicularis
Carex aenea	large horsetail	Taraxacum officinale	Beckmannia syzigachne	Equisetum scirpoides	Rubus idaeus
Carex aurea	Larix laricina	Triglochin palustris	Bromus ciliatus	Frageria vesca	Rumex occidentalis
Carex canescens	Ledum groenlandicum	Typha latifolia	Calamagrostis canadensis	Galium trifidum	Salix
Carex diasperma	Leptobryum pyriforme	Vaccinium caespitosum	Caltha palustris	Galium triflorum	Salsola kali
Carex sartwellii	Marchantia polymorpha	Vaccinium vitis-idaea	Carex aquatilis	Geranium bicknellii	Scirpus microcarpus
Carex vaginata	marsh skullcap	water foxtail	Carex aurea	Glyceria grandis	Scirpus validus
Cladina mitis	Melilotus alba		Carex bebbii	Hordeum jubatum	Scutellaria galericulata
Cladonia sp.	Oxycoccus microcarpus		Carex canescens	Juncus bufonius	Sisyrinchium montanum
Cow-wheat	Picea mariana		Carex concinna	Juncus tenuis	Solidago Canadensis
Dicranum undulatum	Pleurozium schreberi		Carex crawfordii	Kobresia simpliciuscula	Sonchus arvensis
Drepanocladus fluitans	Poa interior		Carex diandra	Lotus corniculatus	Stellaria longifolia
Eliocharis palustris	Pohlia nutans		Carex diasperma	Lysimachia thyrsiflora	Taraxacum officinale
Epilobium angustifolium	Polytrichum strictum		Carex interior	Medicago sativa spp	Trifolium pretense
Epilobium ciliatum	Populus tremuloides		Carex sartwellii	Melampyrum lineare	Triglochin maritima
Epilobium glandulosa	Potentilla norvegica		Carex siccata	Melilotus alba	Triglochin palustris
Eriophorum vaginatum	Ptilium criste-castrensis		Carex utricularia	Melilotus officinale	Typha latifolia
flakey white lichen	Ranunculus sceleratus		Carex vaginata	Monolepis nuttalliana	Urtica dioica
Fragaria vesca	Rubus chamaemorus		Ceratophyllum demersum	Neslia paniculata	Vicia Americana

Among the 28 treatment plots at the Syncrude Canada experimental site, 12 were selected for this current study (Table 3.2). These plots had either summer or winter placements of live transplanted peat or stockpiled mineral/peat mix treatments in three different depths. These treatments were not compacted except in plot 8. However, near surface soil physical characteristics such as bulk density in 2010 were not significantly different between plot 8 (0.28 g cm^{-3} for 0 – 15 cm) and the uncompacted stockpiled plots (0.27 g cm^{-3}), which suggested there was no lasting effects of compaction (data from Pugsley, 2011). In addition, two plots, nearly 6 m apart, were established in a peatland area approximately 7 km from the research plots to act as experimental controls. This natural peatland site was drained to some extent and there was a drainage ditch $\sim 25 \text{ m}$ from the experimental plots. Two gas sampling collars were installed in each treatment and control plots.

Table 3.2. Experimental plots and associated treatments chosen for study in 2010.

	Depth: 15 cm		Depth: 50 cm		Depth: 100 cm	
	Winter Placement	Summer Placement	Winter Placement	Summer Placement	Winter Placement	Summer Placement
Transplanted live peat	Plot 11	Plot 06	Plot 18	Plot 02	Plot 12	Plot 19
Stockpiled mineral/peat	Plot 28	Plot 07	Plot 17	Plot 08 (Compacted)*	Plot 10	Plot 04
Natural peatland	Plot A and B					

* As the 50 cm, summer placement, stockpiled plot without compaction was submerged under water, the compacted treatment was used instead.

During the experiment, the plots were irrigated with fresh water on a regular basis to maintain a high and constant volume of water to the plants transplanted or planted with the goal of improving plant growth and survival. Thus, the frequency of irrigation varied depending on evaporation rates and the amount of precipitation. The water was from an artificial pond to the west of the site (Figure 3.3), which was regularly filled by container trucks with fresh water from Mildred Lake. This water was then pumped into the piping system and circulated to the northern and southern pipes to irrigate the plots.

3.2. Field measurements

3.2.1. Collar installation

On May 21, 2010, two gas sampling collars were installed in each plot 1 to 2 m from each other (Figure 3.4). These collars were 15 cm tall, with a diameter of 30 cm and were made from SDR35 12" PVC sewer pipe. The bottom edge was sharpened to facilitate their installation in the soil. Special care was paid to minimize soil disturbance within and around the collar areas. Collars were inserted to a depth of ~ 10 to 13 cm. All vascular vegetation was clipped at the soil surface within the collars on May 22 and when any new vegetation regrew to limit the contributions of above ground autotrophic respiration to the flux measurements. The top edge of the collar had a narrow groove in which the chambers were placed in to ensure a gas-tight seal during the measurements. CO₂ and CH₄ fluxes were both measured using the same collars but with two different techniques. At the beginning of the experiment 28 of these collars were installed in the 12 treatment

plots and 2 natural control plots. The first half of the experiment from May 31 to July 1 is called the "early season" (DOY 151 - 182). During this period, each pair of collars in a given plot were simple replicates.

Approximately half way through the season, on July 4, one of the collars in each plot was replaced with a root exclusion (RE) collar with the undisturbed collar designated as the "control" (CT) collar. This second half of the experiment from July 7 to August 18 is called the "late season" (DOY 188 - 230).

Figure 3.4. Arrangement of collars in the experimental plots.

The RE collars were designed to allow for gas and water flow belowground but isolate and sever any roots growing into the soil within the collar areas. After some time, fluxes from these collars should represent only heterotrophic respiration. The PVC pipes used for the RE collars were cut to different lengths to extend either to the base or to ~ 50 cm depth in the various plots. Collars were 17.5 cm length for the 15 cm depth plots, 52.5 cm for the 50 cm and 100 cm depth plots and 42.5 cm for the natural plots (which approximately coincided with the base of the peat layer). Approximately 2 cm wide slots were cut with a dado blade approximately every 3

cm along the depth of the collar beginning 20 cm from the top of the collar. The surface of the collars was then covered with a 1 μm mesh (Plastok[®] Meshes and Filtration Ltd.) to restrict the growth of fine roots into the collar area.

3.2.2. Flux measurements

Soil respiration was measured with two different chamber systems on the collars between June 3 and August 18, 2010. The first system was a non-steady state flow-through chamber system which measured CO₂ efflux from the soil (Figure 3.5a). This chamber was 40.2 cm tall and 29.9 cm in diameter and made of clear acrylic. It was covered with a black plastic bag to darken the interior and limit any photosynthetic CO₂ uptake of non-vascular species such as lichen and mosses that were not clipped. This opaque chamber had a mixing fan to restrict the development of a temperature or gas concentration gradient within the chamber volume during the two minutes that was required for a typical measurement. The chamber was connected in a loop to an infrared gas analyzer (IRGA) (LI-840, LI-COR BioScience Inc., Lincoln, NE, USA). Flow to the IRGA and through the system were set at about 700 cc/min and 2 L/min, respectively, using flow meters (Rate-Master[®], RMB, Dwyer Instruments Inc., Michigan City, IN, USA) to adjust the flow rate established with a pump (model TD-4X2N, Brailsford & Company, Inc., Antrim, NH, USA). Prior to each measurement, the chamber air was cleared by waving it into the wind, the collar groove was filled with water and the chamber was immediately set in this groove to ensure an air tight seal. During a two-minute measurement, CO₂ and water vapour concentrations and chamber air temperature

were recorded every second on a data logger (CR23X Micrologger, Campbell Scientific Inc., Logan, UT, USA).

All raw data were transferred to a Matlab program for flux calculation using the following formula:

$$F_{\text{calc.}} = \frac{P_{\text{bar}}}{1 + \frac{H_2O}{1000}} \times \frac{1000 \times V_{\text{in}}}{R \times T \times A} \times \frac{dc}{dt} \quad (1)$$

where, $F_{\text{calc.}}$ denotes the CO₂ flux ($\mu\text{mol m}^{-2} \text{ s}^{-1}$), P_{bar} is the average barometric pressure for the day (Pa), H_2O represents the average water vapour concentration (mmol mol^{-1}), V_{in} denotes the volume of the chamber (m^3), A represents the area of the chamber (m^2), T is the average chamber air temperature (K), R is the Universal gas constant ($8.31 \text{ J K}^{-1} \text{ mol}^{-1}$) and dc/dt the rate of change in CO₂ concentration ($\mu\text{mol mol}^{-1} \text{ s}^{-1}$) as calculated from least squares linear regression.

To adjust the volume for each collar situation, the depth of the standing water inside the collars were recorded every time flux measurements were made. Also the distance from the soil surface inside the collar to the top rim of the collar was recorded every week, by measuring four to five different locations for each collar. The averages from these measurements were used to update the volume ($V_{\text{New}} = \pi \times \text{radius}^2 \times (\text{height of chamber} + \text{depth of soil surface to top of the collar} - \text{depth of water in collar})$) and recalculate the fluxes (F_{New}), $F_{\text{New}} = V_{\text{New}} \times F_{\text{calc}} / V_{\text{in}}$.

These measurements were made in a randomized order on all the collars twice a week on rain-free days with a total of 25 sets of measurements over the summer.

The second type of chamber system was a non-steady state, static system used to measure CO_2 efflux and CH_4 exchange (Figure 3.5.b). Two acrylic chambers, each 40.2 cm and diameter 29.9 cm, were made opaque by covering the chambers with black plastic in order to eliminate photosynthetic CO_2 uptake. These chambers had a small vent tube on the top to avoid pressure fluctuations when setting the chamber on the collar. A ~ 50 cm length of tubing was mounted into the lid of the chamber and connected to a set of three-way stopcocks. The first stopcock had a membrane to allow needle insertion and sampling. The other stopcock was attached to a 60 cc syringe to allow mixing of the air inside the chamber.

Figure 3.5. Two different systems used for CO_2 and CH_4 efflux measurements during the field experiment. (a) Non-steady state flow-through chamber system used to measure CO_2 efflux. (b) Non-steady state, static system used to measure CO_2 and CH_4 fluxes.

During a measurement, the collar groove was filled with water, the chamber was placed on the collar, and 5 gas samples 5 minutes apart were taken after manually

mixing the air within the chamber volume by pumping the 60 ml syringe 3 times. For each air sample, 27 mL of air was pulled out by syringe and 24 mL was injected into a 12 mL pre-evacuated vial (Exetainer, Labco. Ltd., Buckinghamshire, UK). The vials had been prepared with ~ 2 mL of magnesium perchlorate as moisture absorbent and capped with a PTFE/silicone septa and a rubber septum. The labeled vials were stored at ambient condition and then sent to Carleton University for CH₄ and CO₂ analysis using a gas chromatograph (GC) (Varian CP 3800, CA, USA) equipped with an electron capture detector (ECD) and a flame ionization detector (FID).

The CH₄ and CO₂ fluxes were then calculated using Eq 1 above with H₂O set to zero and the rate of change in either CH₄ or CO₂ (dc/dt) determined from least squares linear regression of the 5 gas samples obtained over the 20 min measurement period. These measurements were made in a randomized order on all the collars approximately once every 5 days on rain-free days with a total of 17 sets of measurements over the summer.

For both types of flux measurements, the rates of change in concentrations were scrutinized to ensure the quality of the measurements. All 'outliers' including CO₂ concentrations with >100 ppm increase and CH₄ concentrations with >30 ppm increase over a 5 minute period, along with the ebullition events were omitted from the calculated CO₂ and CH₄ diffusive fluxes. Ebullition events were identified by sudden release of high concentrations of CH₄ during the measurement period and often on the first or second gas sample i.e. within 5 min of setting the chamber onto the collar. The weight of the chamber and possible movement of the collars may

have disturbed the soil and led to release of trapped CH₄. Ebullition rates were calculated using the increase from ambient CH₄ (set to 2.0 ppm if not observed during the measurement) to the maximum CH₄ concentration observed and the time it took to reach this level. For example, if the first gas sample had the highest CH₄ concentration, fluxes were calculated assuming it took 5 min for this emission of CH₄ to occur.

A comparison of CO₂ fluxes from the flow-through and the static chamber systems found CO₂ effluxes to be statistically different ($F_{(1)} = 346.3$, $P = 0.001$) and significantly greater in the flow-through vs. static system at the natural plots (mean and standard deviation of the static system efflux = $2.2 \pm 0.2 \mu\text{mol m}^{-2} \text{s}^{-1}$, and flow-through efflux = $9.7 \pm 1.0 \mu\text{mol m}^{-2} \text{s}^{-1}$). At the natural plots, the water table was always at least ~ 20 cm below the surface and with the relatively turbulent conditions in the flow-through chamber, this system may have “flushed” out CO₂ built up in the highly porous aerated surface peat. Consequently, over a 2 min period, the flux as measured with the flow-through system did not represent the diffusive flux of CO₂ from the peat at the natural plots and was not used in further analyses. In the case of the static system, the long measurement time and minimal mixing was assumed to better capture the true diffusive flux at the natural plots. At the stockpiled plots, there was no significant difference in fluxes from either system ($F_{(1)} = 1.8$, $P = 0.16$; static system efflux = $1.5 \pm 0.8 \mu\text{mol m}^{-2} \text{s}^{-1}$; flow-through efflux = $1.6 \pm 1.0 \mu\text{mol m}^{-2} \text{s}^{-1}$) likely due to the much higher water table (at or within a few cm of the surface in most plots). Even with vigorous mixing, the slow

diffusion of CO₂ through the saturated peat would minimize any effects by the chamber system on CO₂ flux measurements. However, this was not true for live treatments and although the differences between two systems were small, they were significantly different ($F_{(1)} = 89.7$, $P = 0.001$; static system efflux = $3.5 \pm 1.2 \mu\text{mol m}^{-2} \text{s}^{-1}$; flow-through efflux = $5.2 \pm 1.7 \mu\text{mol m}^{-2} \text{s}^{-1}$). This may reflect a difference in the environmental conditions on the days when the two flux systems were used.

3.2.3. Soil temperature, moisture and pH measurements

Soil temperature at different depths and soil moisture content were measured on every flux sampling day to interpret the relationship between soil temperature and moisture with soil respiration.

Chromel-constantan thermocouple (Tc) profiles were installed between pairs of collars in each plot. The Tc profile consisted of thermocouples fixed to a wooden dowel at 2, 5, 10, 20 and 50 cm for plots with 50 cm and 100 cm depth and 2, 5, and 10 cm for plots with 15 cm depth. Soil temperatures were then recorded by using a digital thermometer (Model HH82A, OMEGA Engineering, Inc., USA).

Soil temperature and moisture were also recorded at 10 cm soil depth using a digital thermometer and a Hydrosense soil water measurement system (model CS620, Campbell Scientific, Inc. Canada). To minimize soil disturbance inside the collars, soil temperature and volumetric water content measurements were made next to each collar, in a location with surface characteristics similar to what was found in the collar. The Hydrosense unit was not calibrated to the various soil types. Instead, the porosity of the soil samples was determined from soil samples

(see below) and for VWC values that were higher than the actual soil porosity, VWC was adjusted by using the following formula:

$$VWC_{\text{approx}} = \left(\frac{VWC_R}{VWC_{\text{Max}}} \right) \times \phi \times 100\% \quad (2)$$

where, VWC_{approx} denotes the approximate VWC (%), VWC_R represents the recorded VWC (%), VWC_{Max} is the maximum recorded VWC (%), and ϕ denotes the soil porosity ($V_v V_T^{-1}$).

Daily changes in a plot's water table before and after irrigation were recorded. The water table depth was found by comparing the height from the top of the pipe to the ground (i.e. the datum level), with the pre-irrigation water level, which was the height from the top of pipe to the water level inside the pipe. Further adjustments to the water table record were made using the automated water table measurements (10 min frequency) in some of the water table pipes (Faubert, 2011). Some discrepancies between the distance between the surface and the water table between the pipes and the collars occurred because of the heterogeneous nature of the substrate surface in the plots and also because of the land slope between location of pipes and collars. The water table at the collars was further adjusted if needed using daily measurements of water height in the collars during the experiment.

pH data from live and stockpiled plots were obtained from water samples extracted in 2010 and analyzed for a suite of water quality parameters (Oswald, 2011). pH for the natural plots was measured in the laboratory by placing dry soil

samples in deionized water and measuring the ‘clear’ suspension with a pH meter (Model Accumet AP85, Fisher Scientific, Canada). Also to examine how comparable these laboratory readings were with those assessed from water samples, several dry soil samples from live and stockpiled treatments were examined. The laboratory pH measurements were similar but ~ 0.3 less than what they were in the field.

3.2.4. Nutrient supply rate measurement

For measuring soil nutrient supply rate during this experiment, Plant Root Simulator (PRS)TM probes (Western Ag Innovations Inc., Saskatchewan, AB, Canada) were used. PRSTM had anion and cation exchange membranes that were separately covered in a plastic encasement with the thickness of approximately 0.4 cm and a window cut for exposure of exchange membrane (17.5 cm^2 considering both sides of the probe) to the soil.

The anion exchange membrane adsorbed nutrients, including borate ($\text{B}(\text{OH})_4^-$), nitrate (NO_3^- -N), phosphate (H_2PO_4^-) and sulfate (SO_4^-) and also micronutrient metals such as copper (Cu^{2+}), iron (Fe^{3+}), manganese (Mn^{2+}) and zinc (Zn^{2+}). The cation exchange membranes were responsible for adsorption of nutrients such as ammonium (NH_4^+ -N), magnesium (Mg^{2+}), potassium (K^+), and calcium (Ca^{2+}). The amount of nutrient ions adsorbed on the probe at the end of the burial period represented the nutrient supply rate available to soil microbes and plants for the duration of the burial. This was expressed in units of micrograms of nutrient adsorbed per 10 cm^2 of membrane surface over the burial time.

All the probes in the experimental and natural plots were installed on May 27 and 28, 2010, respectively. The probes were placed vertically at the surface (0-5 cm) and at a 5-12 cm depth in the soil. At any stage of the experiment, before and after installation, the probes were kept in a cool condition, away from chemical materials, sunlight and heat. In total eight sets of probes were installed in each plot (Figure 3.6). Each set included an anion and cation probe. The installation arrangement was as follows:

- 1 cation and 1 anion probe below surface (0-5 cm) inside each of the collars
- 1 cation and 1 anion probe in depth (5-12 cm) inside each of the collars
- 2 cations and 2 anions probes below surface (0-5 cm) between the pair of collars
- 2 cations and 2 anions probes in depth (5-12 cm) between the pair of collars

Figure 3.6. The arrangement of probes in each plot, during the experiment (May 28 - August 18, 2010).

The probes buried outside the collars were used to assess nutrient supply rates in the presence of plants and evaluate the competition between plants and microbes for nutrient absorption.

On July 7 all 112 surface probes (56 sets) were removed and new sets were replaced in the same slots to determine changes in nutrient supply during the second half of the experiment when the RE collars were in place. To minimize soil disturbance which could cause errors in daily flux measurements, the deeper probes were not changed. All surface probes in the first and second installations were left in the soil for an equal length of time. All probes were removed August 17 and sent to the Western Ag Innovation laboratory for processing. There, the probes were eluted using a 0.5N HCl solution for 1 hour, after which the eluates were analyzed calorimetrically by flow injection analysis (FIA) (Bran and Lubbe, Inc., Buffalo, NY) and with Inductively Coupled Plasma (ICP) spectroscopy (PerkinElmer Optima 3000-DV, PerkinElmer Inc., Shelton, CT) to measure different nutrient concentrations.

3.3. Laboratory incubations and soil analyses

At the end of the experimental season on August 17, 2010 soil grab samples were taken from each plot from 0-5 cm and 5-10 cm depths in three different locations with at least 15 cm distance from one another. Samples with a known volume were also collected using a sampler (Wardenaar corer, Eijkelkamp Agrisearch Equipment BV, Netherlands) specifically designed for peat soil. Profiles were placed in wooden frames and covered with plastic wrap. All samples were shipped to Carleton University where they were stored frozen.

The grab samples were used in incubations to examine potential rates of microbial respiration in the different plots and relate these to the different soil

characteristics such as SOM content, C:N ratio, and bulk density (from core samples).

For the incubation analysis, frozen samples were defrosted overnight. Subsamples (~20 g) were placed in separate pre-weighed Buchman funnels with a sterile membrane 0.45 μm filter. The soils were wetted with deionized water and brought to a uniform matric potential by applying suction using a vacuum pump (Model DOA-P104-AA, Gast, USA) at ~ 68 kPa. The funnel bottoms were removed and the soil samples were then weighed and placed in wide-mouth 1 L Mason jars. The jars were capped with the snap lids with a rubber septa installed in the centre of the lid for gas sampling during the incubation period. A needle was inserted into the septa to supply compressed air at 1 L min^{-1} for 5 min and a second for venting to flush out the jars and bring them to approximately ambient CO_2 concentrations. Two rounds of incubations were carried out. The first set began September 17, 2010 with a total of 16 samples (14 samples from depth of 0-5 cm of the first replication, and two empty jars as blanks). The second set began November 26 2010, with a total of 30 samples, (2 blanks, 28 samples from depths 0-5 and 5-10 (14, each)). The jars were incubated in the dark at room temperature and sampled on days 1, 2, 4, 7, 14, 21, 33, 49, 62 and 97. On each sampling date room air temperature was recorded (typically between 20 and 21.5 $^{\circ}\text{C}$) and the jars were weighed, sampled and then flushed with compressed air (as described above). Sampling involved removing 30 mL of headspace air with a needle and 25 mL was immediately injected into a 12 mL pre-evacuated vial and analyzed on the GC as described for the static non-flow through chamber measurements above.

To calculate the rate of CO₂ production (microbial respiration), the volume of the headspace was measured for each individual incubation jar as the volume of the jar (1 L) subtracted by the volume of funnel filled with soil. The rate of CO₂ production was calculated as,

$$F_{CO_2} = cV \left(\frac{P}{RT} \right) \times m_c \times \frac{m_s}{d} \quad (3)$$

where *c* is CO₂ of sample (ppm), *V* is volume of headspace in the jar (L), *P* is pressure (101.3 kPa), *R* is the ideal gas constant (J mol⁻¹ K⁻¹), *T* is incubation temperature in Kelvin, *m_c* is the molecular weight of C, *m_s* is dry soil weight (g) of sample, and *d* is number of days since last sampling. The total C mineralized (*C_{min}*, mg C g dry soil⁻¹) was taken as the cumulative *F_{CO2}* for the duration of the incubation. When *C_{min}* was expressed as mg C g soil organic C⁻¹, the soil weight (*m_s*) is replaced by soil organic carbon content (SOC, g).

At the completion of the incubation, the incubated soil samples were weighed and then oven dried at 85 °C for 24 hours to determine gravimetric water content. Volumetric water content was estimated using the soil volume determined above but note that this was a disturbed soil sample such that the volume was likely less than it would have been in the field. Samples were crushed and homogenized and subsamples were used for the following soil analyses. SOM fraction by mass was assessed using the loss-on-ignition method where SOM is oxidized by heating the soil at 550 °C for 4 hours. C:N elemental analysis was done by dry combustion on a LECO TruSpec-CN analyzer (TruSpec[®], LECO Corporation, St. Joseph, Michigan, USA) at Queen's University.

Bulk densities of the cored soil samples were assessed by calculating the ratio of oven-dry weight of soil per unit of its bulk volume. Soil porosity (ϕ), was calculated from bulk density using a particle density of 1.3 g cm^{-3} .

3.4. Statistical analyses

Statistical analyses were carried out using the program JMP (version 8.0.2, SAS, 2009, Institute Inc., Cary, NC, USA). The dependent variables were CO_2 and CH_4 effluxes and CO_2 production during incubation. Treatment factors included the plot characteristics such as soil depth (15, 50, 100 cm), placement season (summer or winter), soil type (live, stockpiled, natural), and collar type (CT or RE). Independent environmental variables included water table depth, soil volumetric water content, soil temperature and nutrient availability. In this study, these environmental variables were studied in both correlational manner (implying no causality) and using regression analysis (implying causality).

Standard least squares fit, Student's t and Tukey's HSD tests were used to analyze and compare the significance of CO_2 and CH_4 flux variations under different treatments and environmental factors in the field. Multivariate analysis of variance (MANOVA) was performed by applying repeated measures ANOVA, F-test and Wilks' Lambda test to study between and within subjects effects and, consequently, to determine the significance of changes in CH_4 and CO_2 effluxes and interactions between factors/crossed factors. CH_4 effluxes were adjusted to positive values with a uniform addition of $6.1 \text{ nmol m}^{-2} \text{ s}^{-1}$ and log transformed to better meet assumptions of normality. Incubation results were analyzed using

ANOVA. For all tests, an α level of 0.05 was used as the level for significance. It is important to note that treatment combinations were not replicated and as such, statistical analyses were only intended to highlight differences in response variables for the plots at the Syncrude Canada experimental site.

Spearman's correlation coefficient test was carried out to demonstrate the relationship between environmental variables and CH₄ and CO₂ fluxes.

For data analysis, fluxes measured with both the flow-through and static chamber methods were treated in two individual parts. The first part of the study investigated fluxes recorded from two different CT collars (CT#1 and CT#2) as replicates with 11 sets of flow-through measurements and 7 sets of static measurements. In the second part, fluxes from two different collar types (CT#1 and RE) were analyzed with 14 sets of flow-through measurements and 10 sets of static measurements.

As described above, the flux data from the field were screened for quality and prior to statistical analysis. For the second part of the study, when a flux from either the CT#1 or RE collar was missing in a given plot on a given date, the flux from the other collar was also omitted to avoid biasing the results.

4. RESULTS

4.1. Field Experiment

4.1.1. CO₂ fluxes

For the following analysis, the season was separated into two parts: early and late. During the early part of the season (DOY 151-182) fluxes recorded from two different CT collars (CT#1 and CT#2) were treated as replicates and in the late part of the season (DOY 188 - 230), the CT #2 collar was replaced with a RE collar. The goal of this arrangement was to be able to compare the effect of root exclusion on C emission rates while environmental variables continued to vary through the season. Further, the two different methods used to measure CO₂ effluxes were analyzed separately as the effluxes measured from the natural plots using the flow-through method were not included in this analysis (see Methods Section 3.4.).

There was a significant treatment, depth and placement effect on CO₂ effluxes with the flow-through method (Table 4.1) and similarly, there was a significant treatment and placement effect on CO₂ effluxes with the static method during both parts of the season (Table 4.2). The static method of measuring CO₂ effluxes resulted in a treatment × collar interaction effect that was significant when the RE collars were in place (Late season in Table 4.2 & Figure 4.1b) with a slightly greater CO₂ efflux for the RE collars in the live treatment only. Otherwise, there was no significant collar effect when the RE collars were in place indicating that excluding roots did not significantly increase or decrease CO₂ effluxes. Summer placements were significantly higher both in live and stockpiled treatments and with both methods. In general, live plots had the greatest CO₂ effluxes while the stockpiled plots had the

lowest effluxes for either flux measurement method (Figure 4.1 & Appendix Table A.T1). The natural plots had intermediate effluxes that were only significantly different from both the stockpiled and live treatments in the late season as measured using the static method (Figure 4.1b). The significant placement and depth effects and related interaction effects were primarily driven by the live, 15 cm, summer placement (average static efflux $\pm$ 1 SE on DOY 151-182: $7.3 \pm 0.2 \mu\text{mol m}^{-2} \text{s}^{-1}$ and on DOY 188-230: $7.8 \pm 0.2 \mu\text{mol m}^{-2} \text{s}^{-1}$). Lowest static effluxes were in stockpiled, 50 cm, summer placement (DOY 151-182: $1.3 \pm 0.2 \mu\text{mol m}^{-2} \text{s}^{-1}$). However, stockpiled treatments were uniformly low regardless of placement season and soil depth and as a result, there were no significant differences among placement and depth in this treatment (for more details see Appendix Table A.T1 and Figures A.F1 – A.F4).

Figure 4.1. Average CO₂ emissions ($\mu\text{mol m}^{-2} \text{s}^{-1}$) measured by the flow-through system (a) and with the static system (b) during the early (DOY 151-182) and late (DOY 188 -230) seasons. Error bars represent +1 standard deviation. Different letters denote significant differences between interactions of treatments and collars (CT = control, RE = root exclusion) for either the early or late part of the season.

Table 4.1. ANOVA results for CO₂ emission measurements by the flow-through chamber method for the two parts of the season based on fully factorial analysis of treatments (live and stockpiled), collar types, soil placement season and soil depth. Note that interactions with collars were only significant during the early part of the season and as these were simply replicates at this time and not a significant factor effect on their own, the interactions are not presented here. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T1.

Early (DOY 151-182)				Late (DOY 188-230)			
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	398.0	<0.0001*	Treatment	1	506.9	< 0.0001*
Collar	1	2.0	0.1	Collar	1	0.002	0.9
Depth	2	9.5	0.0001*	Depth	2	5.4	0.0045*
Placement	1	4.5	0.0344*	Placement	1	29.6	<0.0001*
Treatment × Depth	2	14.7	<0.0001*	Treatment × Depth	2	10.3	<0.0001*
Treatment × Placement	1	14.2	0.0002*	Treatment × Placement	1	15.2	<0.0001*
Depth × Placement	2	10.1	<0.0001*	Depth × Placement	2	6.6	0.0014*
Treatment × Placement × Depth	2	27.4	<0.0001*	Treatment × Placement × Depth	2	42.0	<0.0001*

Table 4.2. ANOVA results for CO₂ emission measurements by the static chamber method for the two parts of the season based on analysis of treatments and collar types for live, stockpiled, and natural treatments along with placement and soil depth for live and stockpiled treatments. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T2.

Early (DOY 151-182)				Late (DOY 188-230)			
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	38.2	< 0.0001*	Treatment	2	117.6	< 0.0001*
Collar	1	0.1	0.7	Collar	1	0.3	0.5
Treatment × Collar	2	0.1	0.9	Treatment × Collar	2	5.0	0.0073*
Placement	1	16.6	<0.0001*	Placement	1	67.6	< 0.0001*
Depth	2	1.8	0.2	Depth	2	1.7	0.2
Treatment × Depth	2	5.7	0.0038*	Treatment × Depth	2	7.8	0.0005*
Treatment × Placement	1	0.3	0.5	Treatment × Placement	1	2.7	0.1
Depth × Placement	2	3.4	0.0330*	Depth × Placement	2	0.2	0.7
Treatment × Placement × Depth	2	7.5	0.0008*	Treatment × Placement × Depth	2	19.8	< 0.0001*

Temporal trends in CO₂ emissions were subtle with fluxes tending to increase at the beginning of the early season and decrease at the end of the late season with peak fluxes ~ DOY 200 to 210 (Figure 4.2). By DOY 215, emissions from the stockpiled treatment reached their lowest values while emissions from the live treatments were lowest at the beginning of the early season. Repeated measures ANOVA (RM ANOVA) was used to identify any linear trends through time that may differ among treatments. Time was a significant factor for all but the late season static flux measurements, but time only significantly interacted with treatment for the early season flow-through flux measurements (Table 4.3 and 4.4) when CO₂ emissions were increasing in the live treatment and slightly decreasing in the stockpiled treatment (Figure 4.2a). As with the previous ANOVA results, CO₂ emissions on average were significantly different between live and stockpiled treatments in both flow-through and static measurement systems (Table 4.3 and 4.4). Collar effects (i.e. differences among CT collar replicates in early season and CT and RE collars in late season) and also treatment × collar interactions were not significantly different for either measurement system.

Table 4.3. RM ANOVA results for CO₂ emission measurements by the flow-through chamber method for the two parts of the season based on analysis of treatments (live and stockpiled), collar types, time and their interactions. F statistics correspond to the Wilks' lambda test. Asterisks denote significant differences ($\alpha = 0.05$).

Early (DOY 151-182)					Late (DOY 188-230)				
Effect	NumDF	DenDF	F-Value	Prob > F	Effect	NumDF	DenDF	F-Value	Prob > F
Treatment	1	20	40.4	<0.0001*	Treatment	1	18	51.8	< 0.0001*
Collar	1	20	0.2	0.6	Collar	1	18	0.03	0.8
Treatment × Collar	1	20	0.03	0.8	Treatment × Collar	1	18	0.2	0.6
Time	10	11	14.2	<0.0001*	Time	13	6	8.6	0.0073*
Time × Treatment	10	11	6.9	0.0017*	Time × Treatment	13	6	3.1	0.08
Time × Collar	10	11	1.0	0.0017*	Time × Collar	13	6	0.5	0.5
Time × Collar × Treatment	10	11	2.9	0.0467*	Time × Collar × Treatment	13	6	0.8	0.8

Table 4.4. RM ANOVA results for CO₂ emission measurements by the static chamber method for the two parts of the season based on analysis of treatments (live, stockpiled, and natural), collar types, time and their interactions. F statistics correspond to the Wilks' lambda test. Asterisks denote significant differences ($\alpha = 0.05$).

Early (DOY 151-182)					Late (DOY 188-230)				
Effect	NumDF	DenDF	F-Value	Prob > F	Effect	NumDF	DenDF	F-Value	Prob > F
Treatment	2	22	11.9	0.0003*	Treatment	2	16	15.3	0.0002*
Collar	1	22	0.03	0.8	Collar	1	16	0.1	0.7
Treatment × Collar	2	22	0.03	0.9	Treatment × Collar	2	16	0.8	0.4
Time	6	17	6.8	0.0008*	Time	9	8	2.4	0.1
Time × Treatment	12	34	1.2	0.3	Time × Treatment	18	16	1.2	0.3
Time × Collar	6	17	0.4	0.08	Time × Collar	9	8	0.6	0.7
Time × Collar × Treatment	12	34	0.4	0.9	Time × Collar × Treatment	18	16	0.4	0.9

Figure 4.2. Average daily CO₂ emissions (μmol m⁻² s⁻¹) measured by the flow-through system (a) and the static system (b) during the early (DOY 151-182) and late (DOY 188-230) season. Solid lines represent emissions from control collars (CT) #1, and dashed lines represent emissions from CT #2 in early season and root exclusion collars (RE) in the late season. Early and late season have been separated by a grey vertical line. Error bars represent +1 standard error.

4.1.2. CH₄ fluxes

In general, the live treatment had the greatest CH₄ effluxes (full season average ± 1 SE = 39.9 ± 11.1 nmol m⁻² s⁻¹) while the natural treatment had the lowest fluxes with near neutral exchange (-1.8 ± 0.4 nmol m⁻² s⁻¹). The stockpiled treatment had intermediate effluxes (25.8 ± 10.1 nmol m⁻² s⁻¹) that were significantly different from live and natural treatments in both seasons (Figure 4.3 and Tables 4.5 & A.T2). There was no significant collar effect for either season (Table 4.5). The effects of placement season and substrate depth on CH₄ effluxes were inconsistent resulting in significant interactions with treatment (Table 4.5). Overall, the highest effluxes were from the stockpiled 100 cm summer plots and the live 15 cm summer plots in both seasons. In both seasons, the lowest effluxes (from the experimental treatments) were from the stockpiled 15 cm summer and winter plots. All Tukey HSD results are presented in Table A.T2.

Figure 4.3. Average CH₄ emissions (nmol m⁻² s⁻¹) measured by static system during the early (DOY 151-182) and late (DOY 188 - 230) late part of the season. Error bars represent +1 standard deviation. Different letters denote significant differences between treatments for either the early or late part of the season. Although not a significant effect, fluxes are shown for each collar type (CT = control, RE = root exclusion). Statistical analysis was carried out on log transformed fluxes.

Table 4.5. ANOVA results for CH₄ emissions measured by the static chamber method for the two parts of the season based on analysis of treatments and collar types for live, stockpiled, and natural treatments along with placement and soil depth for live and stockpiled treatments. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T2.

Early (DOY 151-182)				Late (DOY 188-230)			
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	18.6	<0.0001*	Treatment	2	97.4	<0.0001*
Collar	1	0.4	0.4	Collar	1	2.9	0.08
Treatment × Collar	2	0.4	0.6	Treatment × Collar	2	1.8	0.1
Placement	1	32.6	<0.0001*	Placement	1	36.3	<0.0001*
Depth	2	3.3	0.0378*	Depth	2	10.4	<0.0001*
Treatment × Depth	2	44.9	<0.0001*	Treatment × Depth	2	39.3	<0.0001*
Treatment × Placement	1	1.4	0.2299	Treatment × Placement	1	0.6	0.4
Depth × Placement	2	1.6	0.1999	Depth × Placement	2	7.0	0.0011*
Treatment × Placement × Depth	2	26.2	<0.0001*	Treatment × Placement × Depth	2	4.6	0.0106*

Through the study period, CH₄ emissions increased through time in the live treatment and to some extent, the stockpiled treatment (Figure 4.4). In the natural treatment however, the actual fluxes recorded in the field declined through time and thus showed slightly greater CH₄ uptake rates by the end of the season ($R^2 = 0.3$, Slope = $-0.02 \text{ nmol m}^{-2} \text{ s}^{-2} \text{ day}^{-1}$). Repeated measures ANOVA was used to identify any linear trends through time that may differ among treatments. In contrast to the results in Table 4.5, the treatment effect was only significant in the late season when time was considered in the analysis (Table 4.6). Time was a significant factor only during the early season while interactions were not significant during either part of the season (Table 4.6). Consequently, the trends through time for the CT#1 and CT#2/RE collars were very similar within a treatment (Figure 4.4).

Table 4.6. RM ANOVA results for CH₄ flux measurements by the static chamber method for the two parts of the season based on analysis of treatments (live, stockpiled, and natural), collar types, time and their interactions. Asterisks denote significant differences ($\alpha = 0.05$).

Early (DOY 151-182)					Late (DOY 188-230)				
Effect	NumDF	DenDF	F-Value	Prob > F	Effect	NumDF	DenDF	F-Value	Prob > F
Treatment	2	22	3.3	0.05	Treatment	2	8	15.2	0.0019*
Collar	1	22	0.08	0.7	Collar	1	8	0.5	0.4
Treatment					Treatment				
×	2	22	0.08	0.9	×	2	8	0.1	0.8
Collar					Collar				
Time	6	17	3.3	0.0226*	Time	8	1	26.2	0.14
Time					Time				
×	12	34	1.6	0.1	×	16	2	5.7	0.15
Treatment					Treatment				
Time					Time				
×					×				
Collar	6	17	0.3	0.8	Collar	8	1	2.2	0.4
Time					Time				
×					×				
Collar	12	34	0.6	0.7	Collar	16	2	0.9	0.6
×					×				
Treatment					Treatment

Figure 4.4. Average daily CH_4 emissions (a) and log transformed* CH_4 emissions (b) from CT and RE collars measured by static system during the early (DOY 151-182) and late (DOY 188-230) season. Solid lines represent emissions from CT #1, and dashed lines represent emissions from CT #2 in early season and RE in the late season. Early and late season have been separated by a vertical grey line. Error bars represent +1 standard error. *Before log transformation, a constant of $6.1 \text{ nmol m}^{-2} \text{s}^{-1}$ was added to all fluxes to avoid negative values.

4.1.2.1. Ebullition

Ebullition events did not occur until DOY 200 (July 19). Although there was not a significant collar effect on diffusive CH₄ emissions (Tables 4.5 and 4.6), there was a tendency for more ebullition events from CT collars than RE collars (Table 4.7). Also among live, stockpiled and natural treatments, stockpiled plots had the highest number of ebullition events. The average ebullition flux was $2208.6 \pm 262.6 \text{ nmol m}^{-2} \text{ s}^{-1}$ ($\pm 1 \text{ SE}$) in live, $2378.2 \pm 345.2 \text{ nmol m}^{-2} \text{ s}^{-1}$ in stockpiled and $1661.1 \pm 602.2 \text{ nmol m}^{-2} \text{ s}^{-1}$ in natural treatments.

Table 4.7. The number of collars with CH₄ ebullition events on each sampling date beginning July 19, 2010.

Date		Jul. 19	Jul. 23	Jul. 27	Jul. 30	Aug. 6	Aug. 10	Aug. 14	Aug. 18
DOY		200	204	208	211	218	222	226	230
Treatment									
Live	CT	-	6	6	-	-	-	-	-
	RE	-	-	-	-	1	2	2	2
Stockpile	CT	1	6	6	-	2	1	2	-
	RE	-	-	-	-	-	-	-	1
Natural	CT	-	2	2	2	-	-	-	-
	RE	-	-	-	-	-	-	-	-

4.1.3. Environmental factors

4.1.3.1. Soil moisture content

Average soil VWC from 0 to 10 cm depth during the course of experiment was $84.5 \pm 5.3\%$ in live and $70.9 \pm 7.5\%$ in stockpiled treatments (Figure 4.5). In the natural treatment, average VWC was $52.6 \pm 5.7\%$ and also showed a slight decline through time while VWC varied little in the live and stockpiled treatments (Figure 4.5), which was a consequence of regular irrigation of the experimental plots to compensate for evaporative water loss. In the natural peatland the only source of water was rainfall and as the summer progressed, warmer temperatures (Figure 4.6) likely caused the decline in VWC.

Figure 4.5. Average volumetric water content (%) for the 0 to 10 cm soil depth, recorded during the field experiment on days when either flow-through or static chamber system measurements were made (May 28 - August 18, 2010). Error bars represent ± 1 standard deviations. Early and late season have been separated by a vertical grey line.

Figure 4.6. Average daily air temperature (°C) (a) and daily total precipitation (mm) (b) during the early (DOY 151-182) and late (DOY 188-230) season. Early and late season have been separated by a grey vertical line. Error bars represent ± 1 standard deviation.

There was a significant difference in VWC among treatments during both seasons on days when fluxes were measured with the flow-through chamber (ANOVA: DOY 151-182: $F_{(1)} = 159.8$, $P < 0.0001$ and DOY 188-230: $F_{(1)} = 317.3$, $P < 0.0001$). However, differences were not significantly different among collars (DOY 151-182: $F_{(1)} = 0.5$, $P = 0.4$ and DOY 188-230: $F_{(1)} = 0.0004$, $P = 0.9$) nor was there a significant treatment $\times$ collar interaction (DOY 151-182: $F_{(1)} = 3.5$, $P = 0.05$ and DOY 188 -230: $F_{(1)} = 1.2$, $P = 0.2$).

There was a significant difference in VWC among treatments and among collars during the early season on days when fluxes were measured with the static chamber (ANOVA: $F_{(2)} = 142.7$, $P < 0.0001$ and $F_{(1)} = 4.4$, $P = 0.03$ respectively) but there was no treatment $\times$ collar interaction ($F_{(2)} = 1.0$, $P = 0.3$). In the late season variations were significant for treatments ($F_{(2)} = 627.9$, $P < 0.0001$) but not for collars or for treatment $\times$ collar interactions ($F_{(1)} = 0.5$, $P = 0.4$ and $F_{(2)} = 1.6$, $P = 0.1$ respectively).

For the full duration of the study, there was no significant difference in VWC in the live treatments on the days when fluxes were measured with the static chamber system vs. the flow-through chamber system ($F_{(1)} = 2.7$, $P = 0.09$). However, VWC was significantly greater in the stockpiled treatments when flow-through measurements were made vs. when static chamber measurements were made ($F_{(1)} = 5.6$, $P = 0.01$), although the average difference was very small ($1.6 \text{ m}^3 \text{ m}^{-3}$).

There was a statistically significant negative correlation between soil CO_2 emission rates measured with the flow-through chamber system and VWC in live peat (Spearman $\rho = -0.12$, $P = 0.03$) but not for stockpiled ($\rho = 0.01$, $P = 0.86$).

There was also a statistically significant negative correlation between soil CO₂ emission rates measured with the static chamber system and VWC in live treatments ($\rho = -0.25$, $P = 0.0002$) but not for stockpiled ($\rho = -0.03$, $P = 0.6$) nor natural treatments ($\rho = 0.05$, $P=0.6$). Figure 4.7 illustrates these relationships with linear least square regressions when significant. However, the proportion of variation in CO₂ emissions explained by variations in VWC is low ($R^2 = 0.02$ for flow-through chamber system and $R^2 = 0.04$ for static chamber system). Details on significance of Spearman's correlations and linear least square regressions within each treatment for the full and separate early and late seasons are presented in Tables A.T3 and A.T4.

Figure 4.7. Relationship between CO₂ effluxes (μmol m⁻² s⁻¹) and volumetric water content (%) for the 0 to 10 cm soil depth on days fluxes were measured with the flow-through chamber system (a) and static chamber system (b). Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

There was a statistically significant positive correlation between CH₄ emission rates and VWC in stockpiled treatment ($\rho = 0.56$, $P < 0.0001$) and for the natural peatland ($\rho = 0.52$, $P=0.0001$). In live peat treatment this correlation was not significant ($\rho = 0.05$, $P = 0.47$). Details on significance of Spearman's correlations have been presented in TableA.T4.

Figure 4.8. Relationship between log transformed CH₄ efflux (nmol m⁻² s⁻¹) and volumetric water content (%). Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

4.1.3.2. Water table depth (WTD)

Average WTD during the course of the experiment was 8.2 ± 6.8 cm in live, -1.2 ± 4.0 cm in stockpiled treatments and not measured in the natural plots (Figure 4.9). The recorded WTDs were based on water levels in the plots, not the collars. Although the water table depths were higher in the live plots, the collars were situated on hummocks that tended to be above the water table in most cases. For

stockpiled plots however, the peat/mineral surfaces were flat and there were more days in which collars were below water. The number of days in which peat surfaces in the collars were submerged were 4 days in live, 12 in stockpiled and 0 in natural treatments.

For days when fluxes were measured using the flow-through system, there was a significant difference in WTD between live and stockpiled treatments in both seasons (ANOVA: DOY 151-182: $F_{(1)} = 164.6$, $P < 0.0001$ and DOY 188-230: $F_{(1)} = 209.3$, $P < 0.0001$).

For days when fluxes were measured using the static system, there was a significant difference in WTD between live and stockpiled treatments in both seasons (DOY 151-182: $F_{(1)} = 130.2$, $P < 0.0001$ and DOY 188-230: $F_{(1)} = 150.4$, $P < 0.0001$).

For the full duration of the study, there was no significant difference in WTD in the live ($F_{(1)} = 0.07$, $P = 0.7$) and stockpiled treatments ($F_{(1)} = 0.3$, $P = 0.5$) on the days when fluxes were measured with the static chamber system vs. the flow-through chamber system.

Figure 4.9. Average water table depths (cm) recorded during the field experiment (May 28 - August 18, 2010). Negative values indicate water tables below the surface. Error bars represent ± 1 standard deviations.

There was a negative correlation between CO_2 effluxes measured using the flow-through system and WTD in both live and stockpiled treatments (Figure 4.10a) indicating an increase in CO_2 efflux with higher water levels, however this correlation was statistically significant only for the stockpiled treatment (stockpiled: Spearman $\rho = -0.19$, $P = 0.0009$ and live: Spearman $\rho = -0.05$, $P = 0.33$). The same results were found for CO_2 effluxes measured using the static chamber system (Figure 4.10b) where the relationship was significant only for the stockpiled treatment ($\rho = -0.30$, $P < 0.000$) and not for the live treatment ($\rho = -0.11$, $P = 0.09$). Figure 4.10 illustrates these relationships with linear least square regressions when significant. The proportion of variation in CO_2 emissions explained by variations in WTD in the stockpiled treatment is 20% for fluxes

measured with both flow-through and static chamber systems. Details on significance of Spearman's correlations and linear least square regressions within each treatment and within each season are presented in Tables A.T3 and A.T4.

Correlations between CH₄ effluxes and WTD were significantly negative in stockpiled peat ($\rho = -0.2$, $P = 0.003$) with $R^2 = 0.2$ for the associated linear regression indicating that an increase in water level explains some of the corresponding increase in CH₄ efflux. In contrast, CH₄ effluxes and WTD were significantly positively correlated in live peat ($\rho = 0.2$, $P = 0.003$) and associated with a low R^2 of 0.05 (Figure 4.11). Details on correlations have been presented in Table A.T4.

Figure 4.10. Relationship between total CO₂ effluxes (μmol m⁻² s⁻¹) and water table depths (cm) on days when the flow-through chamber was used (a) and on days when the static chamber system was used (b). Lines represent the linear least square regressions for cases when the relationship is significant (P < 0.05).

Figure 4.11. Relationship between total CH₄ efflux (nmol m⁻² s⁻¹) and water table depth (cm) on days when the static chamber system was used. CH₄ effluxes are on logarithmic scale. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

4.1.3.3. Soil temperature

Using the soil temperatures recorded from the thermocouple profile during days when either static or flow-through flux measurements were made, near-surface (2, 5 and 10 cm) soil temperature increased through the study period until approximately DOY 212 while deeper soil temperatures continued to warm later into the study (Figure 4.12).

Over the study period, the average temperature at 2 cm depth was greatest in the dry natural treatment and least in the stockpiled treatment (19.8 ± 4.1 °C in live, 18.9 ± 1.4 °C in stockpiled and 21.1 ± 0.5 °C in natural treatments). This pattern reversed at depth with greatest soil temperature at 50 cm depth in the stockpiled

treatment and least in the natural plots (12.1 ± 0.4 °C in live, 13.0 ± 0.5 °C in stockpiled and 10.4 ± 0.2 °C in natural) (Figure 4.12).

On days when fluxes were measured using the flow-through system, there was a significant treatment effect for soil temperatures at 2 cm depth in late season with higher temperature in the live treatment. There was also a significant treatment effect for soil temperatures at 5 and 10 cm depth in early season with highest temperature in stockpiled treatment. Soil temperatures at 20 and 50 cm depths were significantly different in stockpiled and live treatments during both early and late seasons with higher temperatures in stockpiled (Table 4.8).

For the days when fluxes were measured using the static system, soil temperatures at 2 cm depth in late season, at 5 and 10 cm depth in early season, and at 20 and 50 cm depth during both early and late seasons were significantly different (Table 4.9). However, there were few consistent patterns. For the 2 cm depth, temperatures were greatest in the natural and least in stockpiled treatment. For the 5 cm depth, temperatures were greatest in the natural and least in the live treatment. For the 10 cm depth, temperatures were greatest in the stockpiled treatment and least in the live treatment. For the 20 and 50 cm depths, temperatures were greatest in the stockpiled and least in the natural treatment.

For the full duration of the study, there was no significant difference in soil temperature at any depth among the treatments on the days when fluxes were measured with the static chamber system vs. the flow-through chamber system (Live-2 cm: $F_{(1)} = 0.3$, $P = 0.5$, Live-5 cm: $F_{(1)} = 1.1$, $P = 0.2$, Live-10 cm: $F_{(1)} = 0.8$, $P = 0.3$, Live-20 cm: $F_{(1)} = 2.0$, $P = 0.1$, Live-50 cm: $F_{(1)} = 0.6$, $P = 0.4$,

Stockpiled-2 cm : $F_{(1)} = 0.1$, $P = 0.6$, Stockpiled-5 cm $F_{(1)} = 0.2$, $P = 0.6$,
Stockpiled-10 cm: $F_{(1)} = 0.3$, $P = 0.5$, Stockpiled-20 cm: $F_{(1)} = 1.3$, $P = 0.2$,
Stockpiled-50 cm: $F_{(1)} = 0.8$, $P = 0.3$).

Figure 4.12. Average soil temperatures from 2, 5, 10, 20 and 50 cm depths, recorded during the field experiment (May 28 - August 18, 2010). Error bars represent ± 1 standard deviations.

Table 4.8. ANOVA results for soil temperature (°C) at 2, 5, 10, 20 and 50 cm depth when fluxes were measured using the flow-through chamber system. Asterisks denote significant differences ($\alpha = 0.05$).

Early (DOY 151-182)				Late (DOY 188-230)			
2 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	2.1	0.1	Treatment	1	8.0	0.0048*
5 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	4.1	0.0432*	Treatment	1	0.05	0.8
10 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	14.4	0.0002*	Treatment	1	3.3	0.06
20 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	19.7	<0.0001*	Treatment	1	13.4	0.0003*
50 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	1	31.6	<0.0001*	Treatment	1	42.7	<0.0001*

Table 4.9. ANOVA results for soil temperature (°C) at 2, 5, 10, 20 and 50 cm depth when fluxes were measured using the static chamber system. Asterisks denote significant differences ($\alpha = 0.05$).

Early (DOY 151-182)				Late (DOY 188-230)			
2 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	2.4	0.08	Treatment	2	6.1	0.0024*
5 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	7.4	0.0008*	Treatment	2	0.2	0.7
10 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	6.2	0.0024*	Treatment	2	0.8	0.4
20 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	12.3	<0.0001*	Treatment	2	58.5	<0.0001*
50 cm depth							
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	16.4	<0.0001*	Treatment	2	188.3	<0.0001*

CO₂ effluxes as measured by both the flow-through and the static chamber system correlated positively with soil temperature. The Spearman correlation results are outlined in Appendix Table A.T3 and Table A.T4 Figures 4.13 and 4.14 highlight the response of CO₂ flux to soil temperatures at different depths as measured by the flow-through and static system at the different treatments. For the flow-through measurements, these positive correlations were significant at 2, 5, 10 and 20 cm in live treatment with very low R² values associated with linear regressions of 0.07, 0.06, 0.08, 0.09 respectively and at 10, 20 and 50 cm in stockpiled treatment also with very low R² values of 0.02, 0.007, 0.008 respectively (Table A.T3, Figure 4.13). For the static chamber measurements, these positive correlations were significant at all depths in live treatment with R² values of 0.3, 0.3, 0.3, 0.1 and 0.05 as the depth increased, also at 2, 5 and 50 cm in stockpiled treatment with R² values of 0.09, 0.05, 0.02 respectively and at 2, 5 and 10 cm in natural treatment with R² values of 0.1, 0.08, 0.05 respectively (Table A.T4, Figure 4.14). Consequently, the most variation in CO₂ flux explained by variations in soil temperature occurred in the live treatment with near surface soil temperature and when using the static vs. the flow-through chamber system.

CH₄ emissions from live and stockpiled treatments were found to have a significant positive correlation with soil temperatures at all depths. Maximum and minimum R² values were 0.4 at 20 cm and 0.1 at 2 cm depth in the live treatment, 0.1 in 20 and 50 cm and 0.006 in 2 cm depth in stockpiled. However CH₄ emissions from the natural treatment had a significant negative correlation with soil temperature at 10, 20 and 50 cm depths with R² values of 0.04 , 0.1, 0.2

respectively (Figure 4.15). Details on significance of Spearman correlations have been presented in Table A.T4.

Figure 4.13. Relationship between total CO₂ effluxes (μmol m⁻² s⁻¹) and soil temperatures (°C) at different soil depths on days when the flow-through chamber was used. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

Figure 4.14. Relationship between total CO₂ effluxes (μmol m⁻² s⁻¹) and soil temperatures (°C) at different soil depths on the static chamber system was used. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

Figure 4.15. Relationship between total CH_4 effluxes ($\text{nmol m}^{-2} \text{s}^{-1}$) and soil temperatures ($^{\circ}\text{C}$) at different soil depths on days when the static chamber system was used. CH_4 effluxes are on logarithmic scale. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

4.1.3.4. Bio-nutrient availability

There was a significant difference in nutrient availability among treatments at early and late surface (~ 5 cm) and deep (~ 12 cm) installations in most cases. The exceptions were NO_3^- and Cd in early and late surface and deep probes, Fe and Cu in early and late surface probes, Mn in early surface probes. Nutrient availability based on collars was not significantly different except for NH_4^+ , NO_3^- and P recorded from deep probes, also K in early and late surface probes. Nutrient availability among treatment $\times$ collar interactions were not significantly different at any depth. Details of the ANOVA to examine treatment, collar and treatment $\times$ collar interactions have been presented in Tables 4.10 for total N, NH_4^+ , NO_3^- and A.T6 for the remaining ions.

Figure 4.16 shows early and late surface and also deep N availability. The availability of other nutrients and other metals has been presented in the appendix Figures A.F7– A.F19. Generally nutrient availability in deep probes was found in the following order: live > stockpiled > natural with the exception of Ca, Mg, and B being higher in stockpiled. Also S, Al and Cd rates were found to be slightly higher at the natural site in comparison to live and stockpiled treatments. Rates of NH_4^+ were generally higher in comparison to NO_3^- as a result of denitrification and low mineralization/nitrification in waterlogged soils. There was a positive correlation between average plot and nutrient availability for many nutrients including N. Soil water likely had a significant effect on ion movement and mineralization rates in the soil. Correlations between environmental factors and nutrient availability have been shown in Tables A.T7.

Table 4.10. ANOVA results from separate statistical analyses of surface (~ 5 cm) and deep (~ 12 cm) N availability, based on treatments, collar and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T7.

Total N ($\mu\text{g}/10\text{cm}^2/82$ days)											
Surface probe (~ 5 cm) – Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F
Treatment	2	14.18	<0.0001*	Treatment	2	1.17	0.32	Treatment	2	34.27	<0.0001*
Collar	2	0.54	0.58	Collar	2	1.10	0.34	Collar	2	7.2	0.0025*
Treatment × Collar	4	1.17	0.34	Treatment × Collar	4	0.49	0.74	Treatment × Collar	4	0.54	0.7
NO ₃ ⁻ -N ($\mu\text{g}/10\text{cm}^2/82$ days)											
Surface probe (~ 5 cm) – Early				Surface probe (~ 5 cm) – Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F	Effect	DF	F Ratio	Prob > F
Treatment	2	2.90	0.06	Treatment	2	1.65	0.20	Treatment	2	2.78	0.07
Collar	2	0.13	0.87	Collar	2	0.27	0.75	Collar	2	8.31	0.0012*
Treatment × Collar	4	0.34	0.84	Treatment × Collar	4	0.54	0.70	Treatment × Collar	4	4.24	0.0070*
NH ₄ ⁺ -N ($\mu\text{g}/10\text{cm}^2/82$ days)											
Surface probe (~ 5 cm) – Early				Surface probe (~ 5 cm) – Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F	Effect	Effect	F Ratio	Prob > F
Treatment	2	15.35	<0.0001*	Treatment	2	7.58	0.0019*	Treatment	2	30.97	<0.0001*
Collar	2	0.39	0.67	Collar	2	3.00	0.06	Collar	2	5.08	0.0119*
Treatment × Collar	4	1.32	0.28	Treatment × Collar	4	0.44	0.77	Treatment × Collar	4	1.68	0.17

Figure 4.16. NH_4 capture by PRSTM probes in surface (0 -5 cm) (a) and deep (10 - 12 cm) (b) treatments (live peat, stockpiled mineral/peat and natural) and from control, root exclusion and between collars. Probe supply rate for surface probes is in microgram per 10 cm² per 41 $\pm$ 1 days (Early: June 28 - July 7, Late: July 7 - August 17, 2010) and for deep probes is in microgram per 10 cm² per 82 $\pm$ 1 days (between June 28 and August 17, 2010). Error bars represent +1 standard deviation. Levels not connected by same letter are significantly different in interactions between treatment groups and collar types. Different capital letters denote significant differences for each measurement duration.

* Method detection limit for $\text{NH}_4^+ - \text{N}$ is 2 μg per 10cm² per 41 days of burial.

* Method detection limit for $\text{NH}_4^+ - \text{N}$ is 2 μg per 10cm² per 82 days of burial.

There was a significantly positive correlation between CO₂ and CH₄ emission rates measured by flow-through and static chambers in the field with total Nitrogen (NH₄ and NO₃) availability from the deep probes due to the available labile SOM.

There was a significant positive correlation between CO₂ (static chamber: $r = 0.69$, $p < 0.0001$, Figure 4.17) and CH₄ fluxes ($r = 0.64$, $p = 0.000$) in the field and total N supply rate as measured with the deep PRS probes inside the gas flux collars. More details on the correlations between fluxes and nutrient supply rates are given in Table A.T8.

Figure 4.17. Relationship between the average CO₂ efflux recorded by the static system (μmol m⁻² s⁻¹) and total Nitrogen availability (μg/10 cm²/82 days). Each symbol represents the average CO₂ flux in the field for the full measurement period from each plot (2 collars) and total Nitrogen availability from deep probes inside the collars. Line represents the linear least square regressions when the relationship is significant ($P < 0.05$).

4.2. Laboratory Incubations

4.2.1. Potential biodegradability of peat

$C_{\min}$ both per g dry soil and per g C were significantly different among treatments but there was no significant difference between the two sample depths nor were treatment $\times$ sample depth interactions significant (Table 4.11).

$C_{\min}$ for the live treatment was significantly greater than either stockpiled or natural treatments. However, $C_{\min}$ in the stockpiled and natural treatments were not significantly different (Figure 4.18 and Table A.T9).

Table 4.11. ANOVA results for mineralized C respired from treatments during the incubation period, based on analysis of treatments (live, stockpiled, and natural), and soil sample depths (~ 5 and ~10 cm) and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T9.

Mineralized C (mg C-CO ₂ g ⁻¹ dry soil d ⁻⁹⁷)				Mineralized C (mg C-CO ₂ g ⁻¹ C d ⁻⁹⁷)			
Effect	df	F Ratio	Prob > F	Effect	df	F Ratio	Prob > F
Treatment	2	10.2	0.0007*	Treatment	2	7.5	0.003*
depth	1	2.0	0.1	depth	1	2.5	0.1
Treatment				Treatment			
$\times$	2	0.05	0.9	$\times$	2	0.1	0.8
depth				depth			

Figure 4.19 shows the average respiration rate in live, stockpiled and natural peat soil samples during 97 days of incubation and the potential for continuation of emissions after this incubation period. $C_{\min}$ rates were greatest at the beginning of the laboratory soil incubation experiment and declined over time. Time series for $C_{\min}$ rates for the individual soil samples and results averaged for each sample depth are presented in appendix Figures A.F20- A.F27.

Figure 4.18. Average mineralized C expressed as $C_{\min}$ per mass dry soil ($\text{mg C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-97}$) in panel (a) and $C_{\min}$ per mass soil C ($\text{mg C-CO}_2 \text{ g}^{-1} \text{ C d}^{-97}$) in panel (b). Different letters denote significant differences between interactions of treatments and depths. Error bars represent ± 1 standard deviation. Grey and white bars represent the emissions from 0- 5 and 5-10 cm depths, respectively.

Figure 4.19. Average microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from soil samples at 0-5 and 5-10 cm depths during 97 days of incubation. Blue, red and green lines represent the emissions from live, stockpiled and natural treatments respectively. Error bars represent ± 1 standard deviation.

N content of these soils (before incubation) was significantly greatest in stockpiled and least in live treatment. Carbon content on the other hand was significantly greater in live treatment (Tables 4.12, 4.13 and Figure 4.20). Carbon, N and C:N ratios were not significantly different among sample depths nor were treatment $\times$ sample depth interactions significant (Table 4.21). Details are shown in Appendix Table A.T10

Table 4.12. ANOVA results for C, N and C:N based on analysis of treatments (live, stockpiled, and natural), and soil sample depths (0- 5 and 5-10 cm) and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects are presented in Appendix Table A.T10.

Mineralized C (mg C-CO₂ g⁻¹ dry soil d⁻⁹⁷)			
Effect	df	F Ratio	Prob > F
C (%)			
Treatment	2	9.88	0.0009*
Sample depth	1	0.08	0.7765
Treatment × Sample depth	2	0.02	0.9742
N (%)			
Treatment	2	12.8	0.0002*
Sample depth	1	0.08	0.7757
Treatment × Sample depth	2	0.08	0.9210
C:N Ratio			
Treatment	2	23.4	<0.0001*
Sample depth	1	0.45	0.5058
Treatment × Sample depth	2	0.08	0.9187

Figure 4.20. Average C (a), N (b) and C:N ratios (c) in incubated treatments. Different letters denote significant differences between treatments. Error bars represent ± 1 standard deviation. Grey and white bars represent averages 0-5 and 5-10 cm depths, respectively.

There was a general tendency for $C_{\min}$ as expressed as g C/ g dry soil to be greater with greater SOM indicating the importance of substrate on respiration rates (Figure 4.21). However, there was no significant correlation between these variables.

Figure 4.21. Relationship between total mineralized carbon per gram soil (mg C-CO₂ g⁻¹ dry soil) and organic matter content (g g⁻¹) in incubated soil samples. Each symbol represents either values from 2 soil samples at 0-5 cm depth or the result from a single soil sample at 5-10 cm depth from each plot.

Among plots within a treatment, there was only a significant correlation between $C_{\min}$ and pH for the incubated samples from the live treatment. Figure 4.22 illustrates this relationships with linear least square regression for the live treatment ($R^2 = 0.6$).

Figure 4.22. Relationship between total mineralized carbon ($\text{mg C-CO}_2 \text{ g}^{-1} \text{ C}$) and pH in incubated soil samples. Each symbol represents either the average from 2 soil samples at 0-5 cm depth or the result from a single soil sample at 5-10 cm depth from each plot. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

There were significant differences in bulk density and porosity among treatments. While bulk density of the stockpiled treatment was significantly greater than live and natural treatments, there was no difference between these latter two (Table 4.13). Given that porosity is derived from the bulk density measurements, the same pattern was also observed for this variable (Table 4.13). There was no significant difference in bulk density or porosity between the 0-5 cm and 5-10 cm depths. As summarized in Table 4.13, soil characteristics were similar between live and natural treatments. However, fluxes were similar between stockpiled and natural treatments and generally much lower than those in the live treatment.

Table 4.13. Comparison between C:N ratio, C and N content, soil porosity, bulk density, and carbon mineralization rates in incubated soil treatments, average static chamber system CO₂ and CH₄ fluxes, average flow-through system CO₂ and the average total N availability for the ~12 cm probes. Different letters denote significant differences between treatments. Values in parenthesis show 1 standard deviation from the mean.

Factors	Treatments		
	Live	Stockpiled	Natural
N (%)	0.8 (0.2) ^B	1.5 (0.4) ^A	0.9 (0.1) ^B
C (%)	50.2 (1.9) ^A	39.6 (8.6) ^B	52.3 (1.2) ^A
C:N	67.4 (19.6) ^A	27.4 (3.1) ^B	56.9 (7.4) ^A
Bulk Density (g cm ⁻³)	0.094 (0.022) ^B	0.265 (0.045) ^A	0.111 (0.026) ^B
Porosity (V _v V _T ⁻¹)	0.927 (0.018) ^A	0.796 (0.035) ^B	0.914 (0.020) ^A
Average total N for the ~12 cm probes (µg/10cm ² /82 days)*	13.0 (3.0) ^A	5.8 (2.2) ^B	5.0 (1.7) ^B
Average CO ₂ respired during 97 day incubation (mg C-CO ₂ g ⁻¹ dry soil)	152.9 (80.3) ^A	55.6 (58.6) ^B	41.5 (14.7) ^B
Average CO ₂ respired during 97 day (mg C-CO ₂ g ⁻¹ soil C)	309.0 (166.3) ^A	144.9 (162.7) ^B	79.2 (27.7) ^B
Average CO ₂ flux (µmol m ⁻² s ⁻¹) measured by static chamber system	3.5 (1.2) ^A	1.5 (0.8) ^B	2.2 (0.6) ^C
Average CO ₂ flux (µmol m ⁻² s ⁻¹) measured by flow-through system	5.2 (1.7) ^A	1.6 (1.0) ^B	-
Average CH ₄ flux (nmol m ⁻² s ⁻¹) measured by static chamber system	39.9 (45.6) ^A	25.8 (41.5) ^B	-1.8 (1.5) ^C

* Values are based on the N recorded from CT, RE & outside the collars.

Among plots within a treatment, there was only a significant correlation between C_{min} and %C for the samples from the live treatment. Figure 4.23 illustrates this relationships with linear least square regression for the live treatment ($R^2 = 0.4$). N% and C:N ratios were not significantly correlated with C_{min} in any treatment (Figure 4.23). Details on significance of Spearman correlations and linear least square regressions of C%, N% and C:N within each and among all treatments has been presented in Table A.T11.

Figure 4.23. Relationship between total mineralized carbon (mg C-CO₂ g⁻¹ C) and % C (a), %N (b), C:N ratio (c) in incubated soil samples. Each symbol represents either the average from 2 soil samples at 0-5 cm depth or the result from a single soil sample at 5-10 cm depth from each plot. Lines represent the linear least square regressions for cases when the relationship is significant ($P < 0.05$).

Figure 4.24 illustrates the relationship between the average CO₂ efflux measured by the static chamber system in the field with the average C_{min} measured during the laboratory incubation for each plot (n = 2, R² = 0.5).

As the environmental factors in the laboratory and field conditions were totally different the positive correlation between these two fluxes clearly shows the effect of the treatment characteristics such as bulk density, porosity, C:N, N% and above all soil labile carbon content on the microbial decomposition.

Figure 4.24. Relationship between the average CO₂ efflux recorded by the static system in the field (μmol m⁻² s⁻¹) and the average mineralized carbon during laboratory incubations (mg C-CO₂ g⁻¹ dry soil). Each symbol represents the average CO₂ flux in the field for the full measurement period from each plot (2 collars) and the average rate of mineralized carbon from incubated samples from 3 soil samples from each plot (2 from 0-5 cm and 1 from 5-10 cm depths). Line represents the linear least square regressions when the relationship is significant (P < 0.05).

5. DISCUSSION

5.1. CO₂ Fluxes

In this study, the live treatment had the greatest CO₂ emissions both in the field and in the laboratory incubation experiment. The strong relationship between C_{min} in the laboratory, where soil moisture and temperature were controlled, and average CO₂ flux in the field, where various environmental conditions differed among the treatments, suggests that environmental conditions were less important than the potential biodegradability of the soil in determining decomposition rates. This was taken as evidence that other soil properties such as the greater amount of SOC and the potential for more labile SOC were important in driving greater decomposition rates from the live treatment compared to stockpiled and natural treatments.

Compared to the live treatment, lower CO₂ emissions were observed both in the field and in the laboratory from the stockpiled treatment. The soil in this treatment was composed of mineral/peat soil mix, had less SOC with a lower C:N ratio and the potential for more recalcitrant SOC presumably due to greater humification of the organic material. Humification is the process by which organic matter is transformed into humus, the relatively stable end-product of microbial decay (Zech & Kogel-Knabner, 1994).

During the decomposition process, part of the mineralized N and labile C usually become inaccessible due to consumption by microbes and there is a loss of labile organic matter by leaching while there are additions of the recalcitrant

components of dead microbes' cell walls. Consequently, the SOM quality deteriorates (becomes more recalcitrant) (Berg & Staaf, 1980; Melillo et al., 1982).

The decomposition of SOM is affected by the C:N ratio which is the relative proportion of the two elements. Although the process of decomposition requires more C than N, both elements are necessary and both impact the rates of mineralization and immobilization. During decomposition of SOM, the C:N ratio influences the amount of N mineralized for each C respired and the amount of N immobilized by decomposers (Pineiro et al., 2006). When there is C available, decomposition slows once N is largely immobilized. Some microbes will then die thereby replenishing the source of available N. The rest of the microbes start using the remainder N to form new cell materials. During cell wall formation C is used to provide energy and lost as CO₂, consequently, the amount of C is reduced while N from dead microbes is recycled (Tateno & Chapin, 1997). When C is less than what is required for converting available N into protein, microbes use the available C and release the excess N as ammonia (Tateno & Chapin, 1997). SOM with high C:N generally has reduced decomposition rates compared to SOM with low C:N (Prescott et al., 2005). High C:N ratios are associated with greater N immobilization because microbes require mineral N to be able to continue metabolizing SOM with high C content (Janssen, 1996). Since microbes need N and other elements for decomposition, if there is little N in the soil, decomposition will be slowed down. For SOM with low C:N, there is potentially sufficient N relative to C for decomposition to proceed relatively unhindered (Agren et al. 1992; Hobbie & Vitousek, 2000).

Although the C:N ratio and C and N content of the soils are all important factors for controlling decomposition rate, the quality of the SOM is always a better predictor of decomposition rate (Berg & Staaf, 1980; Melillo et al., 1982). During decomposition the loss of C is primarily from consumption of labile substrates while there may be little change in the recalcitrant C pool (Prescott et al., 1999). This would lead to an increase in the C:N ratio of the labile fraction of the SOM while the recalcitrant C levels remain unchanged. Since the labile SOM is the fraction of SOM that participates in decomposition (Trumbore, 2000; Taneva et al., 2006), the overall soil C:N ratio is not necessarily a good indicator of decomposition rates and may lead to misinterpretation of the process (Davidson et al., 2000; Melillo et al., 2002). As a result, it is important to group the SOM pool into fractions with different lability and associated C:N ratios instead of focusing on the overall C:N ratio (Davidson et al., 2000; Melillo et al., 2002; Knorr et al., 2005). In this study, the C:N ratios of the live and natural treatments' soils were > 50 suggesting there had been an accumulation of organic matter (Van Cleve et al., 1983) which exceeded decomposition rates. In contrast, the stockpiled soil C:N ratio was lower due to the decomposition that would have occurred during storage in the absence of inputs of fresh plant litter with their high C:N ratios. Consequently, although the average C:N value of the stockpiled soil is in the range (20 – 30) often cited as optimum for decomposition (Haynes, 1986), relatively low rates of decomposition in the stockpiled vs. live soil is likely due to the reduced quality of the stockpiled SOM. These issues may also explain why in this study,

there was no significant correlation between C_{min} and %N, %C or total C:N except for in the live treatment where C_{min} decreased with %C ($R^2 = 0.4$).

One potential reason for differences in SOM quality between the live and natural treatments may be related to the type of plant material that makes up the SOM. Highly recalcitrant vegetation such as *Sphagnum* mosses decompose more slowly and effectively reduce decomposition rates (Hajek, 2009). Slow decomposition of *Sphagnum* is a result of acids that inhibit microbial activity produced as *Sphagnum* decays (Johnson & Damman, 1993). There is effectively chemical protection of *Sphagnum* cell wall polysaccharide by recalcitrant compounds including polyphenolic network polymers and a surface coating of lipids (Van Breemen, 1995). Further, the break-down of moss tissue decreases the porosity and pore size and lowers the hydraulic conductivity of the peat (Boelter, 1969; Clymo, 1984). Consequently, the new anoxic condition slows the decomposition rate (Johnson et al., 1990). Although the vascular plants were clipped from inside the collars in this study, the peat surface inside the collars was covered with *Sphagnum* mosses in the natural treatment collars but not in the live treatment collars. Although not quantified, the organic material from the live treatment may have been made up of a greater proportion of non-*Sphagnum* plant tissue than in the natural treatment resulting in the lower rates of C_{min} observed in the laboratory experiment despite similar C, N and C:N values.

In the field and in the laboratory, CO₂ emissions from plot #6 (15 cm, summer) from the live treatment was the highest among all the treatment plots. The C:N ratio of this plot was high which again suggests that despite high total C:N ratio,

the labile SOM fraction may have had lower C:N ratio. This plot was also slightly more alkaline (pH ~7.5) than other live treatment plots (pH ~ 5.5- 6.5). Slight alkalinity can lead to a more favorable pH environment for decomposition because both microbial activity and vegetation growth will increase (Smolders et al., 2006). Although aboveground plants were clipped in the collars, the germinations inside the collars of plot #6 had to be clipped regularly suggesting autotrophic respiration from living below ground roots may have been significant. During incubations, these previously live roots may have provided fresh, labile substrate for decomposition. Plot #12 (100 cm, winter) had the lowest C_{min} of the live treatment plots. Although this treatment had high SOM (0.9 ± 0.04 , $\pm$ SE), it had relatively low pH (~5) which may have led to lower emission rates. Studies have shown that acid conditions both in the peatland soils and in the laboratory result in poor microbial biomass production which leads to decrease in microbial respiration (Farrell & McDonnell, 1986; Kok et al., 1990; Primavesi, 1984). However, despite greater pH ranging from 6.8 to 7.8 in the stockpiled treatment, there was uniformly low C_{min} among all plots. High emission rates from plot #4 (100 cm, summer, stockpiled) in the field experiment may be attributed to greater amounts of total N in comparison to the other stockpiled plots (Berg & Staaf, 1980; Melillo et al., 1982) and would suggest that substrate quality and nutrient availability were more important than pH in determining microbial decomposition rates.

In the field, the spatial and temporal variations in the emissions of CO₂ were related to these variations in SOC quantity and quality but also to variations in soil moisture and temperature. Moisture/aeration regime in the peat soil is influenced

by water table depth, the proportion of total pore spaces filled with water vs. air, and the oxygen content in the water (Rydin & Jeglum, 2006). Aerobic decomposition and CO₂ production rates can decrease when soil moisture drops below 50% or increases above 100% of the dry mass (Haynes, 1986). Generally aerobic decomposition and CO₂ production has a parabolic relationship with soil VWC (Waring & Schlesinger, 1985), with greater effects in the upper soil layers (Silvolva & Ahlholm, 1989). Aerobic decomposition is inhibited when there is a lack of oxygen supply for microbes. The rate of oxygen diffusion is 10,000 times slower in water than in air (Armstrong, 1979) and the solubility of oxygen in water decreases as temperature increases. Thus, in wet warm soils, water inhibits the oxygen supply to microbes while any available oxygen is rapidly depleted by root and microbial respiration (Chapin et al., 1991). Studies have shown that CO₂ fluxes increase with decreasing water table in intact peat from Canadian wetlands (Moore & Dalva, 1993). However, at the other extreme, very dry conditions may inhibit microbial respiration. In peat soils, soil matric potential drops rapidly as VWC decreases below field capacity (Doran et al., 1991; Orchard & Cook, 1983). Soil matric potential, which is a combination of capillary and adsorptive forces, has an important effect on soil microbial activity by controlling nutrient supply to microbial populations (Killham, 1994) which are mostly abundant and active in the root zone (Paul & Clark, 1996).

In this study, the CO₂ emissions in the field remained relatively high in the fully saturated live treatments when compared with the saturated or nearly saturated stockpiled treatment and the relatively dry natural treatment. In the live treatment,

the regular addition of piped in water may have replenished the oxygen supply to some extent in these experimental plots. The plots were irrigated regularly (~ every 2-5 days) with water from an artificial pond, which was regularly filled by water container trucks. This water was then pumped in to the piping system and circulated to the lateral pipes to irrigate the plots. Microbial respiration would then be expected to deplete the oxygen rapidly (Rydin & Jeglum, 2006). Although the same water was applied to the stockpiled treatment, decomposition rates may have remained low due to the poor quality of SOM as discussed above. In the natural treatment VWC was typically between 40 and 70 % integrated over the top 20 cm where precipitation was the only source of water. It was likely that VWC was much lower in the top few cm due to the high porosity of the surface peat leading to low matric potential that may have led to limited nutrient availability and water accessibility to microbes, and played a role in limiting decomposition rates in the natural plots in the field experiment. However, there was little response to variations in VWC. In other studies, the introduction of oxygenated water into drained peat resulted in rapid oxidation of organic matter (Moore & Dalva, 1993). After rewetting dry peat, high concentrations of osmotic metabolites synthesized by microbes create a gradient causing many microbial cells to burst. This results in nutrient availability after the first rain or irrigation. Thus, short-term drying-wetting cycles, such as rain storms can strongly stimulate decomposition in the litter layer and surface soils if the cycles are infrequent (as generally occurs in soils). Frequent moisture fluctuations however, reduce microbial population numbers and result in a decline in decomposition (Clein & Schimel, 1994). Since

there was relatively low $C_{\min}$ from natural treatment soils during the incubation experiment despite all soils being brought to a similar matric potential, the indication was that other factors such as poor quality SOM or perhaps reduced numbers of microbes may have limited CO_2 emissions from these soils.

The soil properties in the different treatments also differed in terms of bulk density and porosity. Higher organic matter content in the live and natural treatments resulted in lower bulk density and higher porosity. Although not directly assessed, it would be expected that these treatments had more and larger macropores. Larger macropores increase water infiltration and microbial activity (De Neve & Hofman, 2000; Pengthamkeerati et al., 2005). Although bulk density was significantly higher in stockpiled treatment in comparison to live and natural treatments (0.3 versus 0.1 g cm^{-3}), differences in the bulk density and porosity (0.8 versus $0.9 \text{ V}_v \text{ V}_T^{-1}$) were not large enough to be considered as controlling factors in this study.

In the field, CO_2 and CH_4 emissions from peatland soils are often observed to increase exponentially with temperature (Alexander, 1961; Lloyd & Taylor, 1994, Moosavi et al., 1996; Conen et al., 2008). This increase in emissions at higher temperatures is partly due to the increased rate of chemical reactions (McKenzie et al., 1998) and the mineralization of more nutrients for microbial activity as a result of enhanced microbial respiration (Winkler & Cherry, 1996). Microbial respiration and biological oxidation systems will be inactivated at sufficiently high temperatures (Lloyd & Taylor, 1994). Also, decomposition will only increase with temperature rise in the presence of moisture, while in dry environments, as the

temperature rises the decomposition rate tends to drop (Bosatta & Agren, 1998) as drought limits the microbial activity and the diffusion of nutrients in the soil pore space (Papendick & Campbell, 1981; Robertson et al., 1997). It is also important to consider that the temperature sensitivity of microbial decomposition depends on SOM quality (Bosatta & Agren, 1998). It has been suggested that the decomposition of high quality SOM is less temperature sensitive than for low quality SOM (Fierer et al., 2005). In this study, CO₂ emissions tended to increase with increasing temperature but the relationships were weak. This was most noticeable for the natural treatment, in part because as the growing season progressed, higher temperatures coincided with a decrease in VWC.

Nitrogen availability controls decomposition rates where labile SOM is available to support microbial needs (Haynes, 1986). Nitrogen availability in the form of NH₄⁺ recorded by the PRS™ probes in the field collars was significantly greater in the live treatment with no significant difference between stockpiled and natural treatments. However, the N% in elemental analysis of soil treatments at both 0-5 and 5-10 cm depths was significantly greater in the stockpiled treatment with no significant difference between natural and live treatments. The reason is that the PRS™ probes give a measure of available or mineralized N in the form of NH₄⁺ and NO₃⁻ absorbed from the soil solution while elemental analysis shows total N in both organic and inorganic forms. With greater soil moisture in the field and potentially more labile SOM both promoting greater decomposition, there may have been a net mineralization of N in the live treatment. In addition, there was sufficient soil moisture to carry these ions to the probes installed at these relatively

shallow depths. In the natural treatment, relatively low rates of decomposition may have resulted in a net immobilization of N and/or less soil moisture would have resulted in limited ion movement to the probe resins. This explanation also likely explained the reduced H_2PO_4^- in the natural vs. live treatments. Low levels of NO_3^- relative to NH_4^+ in all treatments reflected the large potential for denitrification and low mineralization/nitrification in waterlogged soils (Swift et al., 1979; Haynes, 1986).

As plants and microorganisms both use the same pool of nutrient resources (Tilman, 1982), there will be competition for limiting nutrients. Because N is a limited resource in most ecosystems and particularly in peatlands, N becomes a limiting factor for microorganism and plant growth (Kaye & Hart, 1997). Lower capture rates of N, K, P, Fe, Mn, Zn, and Pb outside of the collars were likely the result of plant root and microbial competition with probes for nutrient adsorption, while this competition inside the collars was limited largely to probes and microbes although some plant growth remained despite continual clipping of above ground vegetation in some collars.

In terms of the effects of placement timing on CO_2 emissions, there was no significant difference between soil physical characteristics among summer and winter plots for either stockpiled or live treatments. Fluxes measured using the flow-through chamber method in the stockpiled and live treatments were used to analyze placement effect. There was no significant difference among stockpiled plots regardless of depth or placement season while live plots did differ significantly resulting in a significant treatment, placement, and depth interaction.

Higher emissions recorded from summer live placement were largely attributed to the influence of plot #6 (live, summer, 15 cm) (Table A.T1). As discussed above, this plot was more alkaline than the other live plots which may have lead to a more favorable pH environment for microbial activity but this appeared to be unrelated to the placement season.

Comparing CO₂ emissions recorded from different treatment depths in the field experiment showed that there were slightly higher emissions in the 15 cm thick peat. Further, this treatment effect was statistically significant when fluxes were measured with the flow- through chamber system, but not with the static chamber system. Placement depth only appeared as a significant factor in the live treatments. Again, plot #6 (live, summer, 15 cm) largely influenced this pattern with inconsistent patterns among the other 5 live plots. As a result, there was no strong evidence that decomposition rates were greater with shallower or deeper substrates. In peatlands, decomposition is greatest within the root zone (Paul & Clark, 1996) and usually decreases at deeper depths in the peat profile (Fierer et al., 2005; Jin et al., 2008; Johnson et al., 1990) due to greater substrate recalcitrance (lower quality substrates) and to less favorable environmental conditions for microbial activity (e.g. high moisture, lowered redox potential and low temperature) (Belyea, 1996). Consequently, reduced CO₂ production at depth may have limited any differences among plots with deeper vs. shallower substrates.

Vegetation coverage in the treatments was observed to be considerably different in terms of density and composition of plant species (data not shown). It was clearly evident that live treatment plots with transplanted vegetation had much

higher plant populations and variety with more woody and perennial species with greater root biomass compared to the stockpiled treatment where vegetation was scattered and mostly annual and perennial herbs and sedges. The two collar types were expected to influence autotrophic respiration from roots in different ways. The CT collars were ~10 cm deep and would have severed roots near the surface. The RE collars extended approximately to the base of the substrate and were expected to sever and exclude all roots from within the collar area. Previous root exclusion studies both in northern peatlands (Makiranta et al., 2010) and northern forest soils (Hogberg et al., 2002; Chen et al., 2002, Lytle & Cronan, 1998) have shown that root exclusion initially enhances CO₂ emissions due to the introduction of dead organic matter to the soil for microbial decomposition. However, as the labile SOM participates in soil respiration, CO₂ fluxes will gradually decrease in root excluded peat due to the omission of autotrophic root respiration along with lack of labile SOM for soil heterotrophic respiration (Harmon et al. 1990, 1999; Chen, 1999). In this study, collar type was either not a significant effect or showed inconsistent patterns. Also, CO₂ emissions for both collar types had a positive trend over time in live and a negative trend over time in stockpiled treatments. This shows that the higher emissions observed in the second part of the experiment, after root exclusion, were not necessarily related to the RE collar installation because the CT collar exhibited the same emission trend.

Generally CO₂ emission rates measured in darkened chambers on undisturbed and vegetated northern peatlands (in the absence of plant photosynthesis) range from 1 to 7 $\mu\text{mol CO}_2\text{-C m}^{-2} \text{ s}^{-1}$ in summer (Martikainen et al., 1995; Moore, 2001;

Moore, 2002). However, soil respiration studies (Heinemeyer et al., 2011) have shown that total soil respiration is frequently under-estimated using these methods because soil collar insertions sever roots and lead to an underestimate of root autotrophic respiration.

Studies of heterotrophic soil respiration in drained forested peatlands in Finland had average summer fluxes of 0.27 to 0.35 g CO₂ m⁻² h⁻¹ (1.7 to 2.2 μmol CO₂-C m⁻² s⁻¹) where the decrease in water level and increase in soil temperature significantly increased the soil respiration rates (Minkinen et al., 2007; Makiranta et al., 2008). Further, greater root density in comparison to most natural peatlands may have increased the CO₂ emissions there. In contrast, peat from a natural peatland in the Northern Pennines, UK was sampled and had roots removed (Heinemeyer et al., 2011). Average midday heterotrophic respiration rates were 1.2 μmol CO₂-C m⁻² s⁻¹ near the surface and about 0.5 μmol CO₂-C m⁻² s⁻¹ at 10 to 20 cm below the surface.

In this study the emission rates from the experimental treatments (stockpiled: 1.5 μmol CO₂-C m⁻² s⁻¹, live: 3.5 to 5.2 μmol CO₂-C m⁻² s⁻¹) were similar or greater to those from a drained Finnish peatland that is clearly experiencing degradation of the peat and loss of soil organic C. They were larger than the heterotrophic rates of the UK peatland but within the range of total respiration rates measured elsewhere. Since the vegetation was clipped in this study, the CO₂ emissions were predominantly heterotrophic and should be lower than total respiration. These results suggest that heterotrophic respiration and thus decomposition rates may not appear to be particularly high compared to those at natural peatlands although this

may have been due to the high water table maintained at these plots. If the water table drops in future, considerably higher CO₂ emissions are possible, especially in the live treatment, which may result in a net loss of soil organic C from these reclaimed peatland trials.

5.2. CH₄ Fluxes

In this study CH₄ emissions in the field were significantly greater in live vs. stockpiled or natural treatments. Higher emissions from the live treatment were likely a result of waterlogged conditions when compared to the drier natural treatment plots while there was more SOM, greater nutrient availability and as discussed above, likely more labile SOM when compared to the waterlogged stockpiled treatment.

For methanogenesis to occur, oxygen depletion must have occurred resulting in a redox potential below -200 to -400 mV (Zehnder, 1978), which describes the availability of terminal electron acceptors in the soil (Lafleur, 2009). Oxidation of CH₄ occurs in a zone close to the water table where oxygen and CH₄ concentrations are high. Usually the potential for CH₄ oxidation by methanotrophs is larger than the potential for CH₄ production by methanogens (Segers, 1998). An increase in electron acceptors like NO₃⁻ decreases CH₄ emissions as a result of suppressed methanogenesis while an increase in NH₄⁺ inhibits in situ CH₄ oxidation due to competition for methane monooxygenase (the enzyme used to oxidize CH₄) and increases CH₄ emissions (Bubier, 1995; Hanson & Hanson, 1996). Thus, it can be concluded that in live and stockpiled treatments, the higher water table generally at

or even above the surface resulted in a greater volume of peat in which methanogenesis could occur and resulted in limited or no volume of peat in which methanotrophy could occur. In both of these treatments NO_3^- -N was low while there was more available NH_4^+ -N in the live treatment to support methanogenic activity. In the drained natural peatland, a low WTD and VWC would have resulted in less volume of peat involved with anaerobic decomposition for CH_4 production while there was a relatively large potential for methanotrophy in the overlying aerated peat. Thus, there was a net uptake of CH_4 in this treatment. Further, methanogens prefer labile C substrates. CH_4 production decreases when labile substrates are depleted (Chanton et al., 1995; Hornibrook et al., 1997; Charman et al., 1999; Clymo & Bryant, 2008). CH_4 flux can be stimulated by adding intermediate substrates (Segers, 1998; Whalen, 2005; Lai, 2009), in order to increase the labile C content of the soil (recent C bound by vegetation) for increasing rates of decomposition (Rovira, 1969).

As with CO_2 emissions, temperature directly influences CH_4 production because metabolic processes such as methanogenesis are strongly temperature dependent. However, it has only minor influence on CH_4 consumption (Segers, 1998; Whalen, 2005). This is because methanotrophy is limited by CH_4 and O_2 diffusion rates rather than kinetics (Smith et al., 2000). Temperature also influences ebullition (Segers, 1998; Whalen, 2005). A temperature rise will decrease CH_4 and O_2 solubility. Since oxidation rates may not increase as quickly as CH_4 production, ebullition may occur.

According to Segers (1998) and Whalen (2005), there is great variation in the temperature sensitivity of CH₄ production. CH₄ emissions from live and stockpiled treatments were found to have a significant positive correlation with soil temperature at all depths. However CH₄ exchange from natural peatlands had a significant negative correlation with the soil temperatures at 10, 20 and 50 cm depths indicating greater rates of CH₄ oxidation with increasing temperature. As the season progressed, evaporation rates likely increased with temperature and WTD (not measured) and VWC decreased in these plots. Consequently, it was likely that CH₄ production was reduced while CH₄ oxidation was enhanced due to both greater activity by methanotrophic bacteria and a greater oxidation zone.

In live and stockpiled treatments, however, the increase in the soil temperature increased CH₄ emissions. This was not only related to a favorable temperature for microbial activity and ion movements but also to the frequent irrigation carried out in the experimental plots to avoid water loss. There was a positive correlation between WTD and CH₄ emissions in live peat. Laboratory studies on peat columns have shown that there is a strong relationship between CH₄ flux and WTD (Moore & Dalva, 1993). The duration of a substrate's exposure to microbial decomposition under favorable environmental condition is also important (Moore & Knowles, 1989; Moore & Dalva, 1993) due to the availability of labile organic matter to microbes (Nilsson, 1992). Accordingly, in live and stockpiled treatments, where WTD was closer to the surface for longer periods of time, the probability of more labile organic matter being available to microbes was greater.

Generally CH_4 is produced in the anaerobic waterlogged area of the peat, and oxidized in the aerated surface peat. The exception occurs when water level is above the peat surface where this condition may lead to lower CH_4 emissions, as a result of CH_4 consumption in the oxygenated water column (Bubier, 1995). Negative correlation between stockpiled treatment WTD and CH_4 emissions was likely due to the mismatch between WTD determination at wells in the plots that were not necessarily representative of the conditions at the collars. Significant positive correlations between VWC and CH_4 emissions in both live and stockpiled treatments shows the importance of soil moisture on anaerobic activities.

The occurrence of ebullition events started approximately on July 19 (DOY 200) at the beginning of the second half of the experiment period. Ebullition occurs under saturated condition when the partial pressure of the gas in water exceeds the hydrostatic pressure of the water column (Schiitz & Seiler, 1989; Chanton & Dacey, 1991; Schutze et al., 1991). As one of the causes of ebullition is mechanical disturbance (Fechner-Levy & Hemond, 1996), ebullition events may have been associated with the RE collar installations on July 4 (DOY 185). However, ebullition events continued throughout the second half of the growing season. More ebullition at this time could have been due to the warmer soil temperatures enhancing CH_4 production and reaching threshold CH_4 concentrations at depth. There were more ebullition events recorded from the CT collars in comparison to RE collars because the CT collars were not as stable and as deep as the RE collars thus small movements while setting down the measurement chambers on the collars may have led to the escape of CH_4 bubbles. Generally, among treatments,

stockpiled plots had the highest number of ebullition events. In saturated stockpiled plots, there was less water above the peat surface than in the live treatment and consequently, there was potentially less hydrostatic pressure at the depth where methanogenesis was occurring.

Generally CH₄ emission rates from Canadian northern peatland are about 80 nmol m⁻² s⁻¹ (Long & Flanagan, 2009). However, in this study the emission rates in live, stockpiled and natural treatments were 39.9 ± 45.6 nmol m⁻² s⁻¹ (± SD), 25.8 ± 41.5 nmol m⁻² s⁻¹ and -1.8 ± 1.5 nmol m⁻² s⁻¹, respectively. Although part of this lower emission is related to the omission of ebullition events, it is mostly related to the environmental condition. Anoxic and very low redox conditions are the most important factors for CH₄ production. Although water levels in experimental treatments were high, regular additions of aerated water to these treatments did not allow the anoxic condition to last long. Thus despite high water levels CH₄ emissions were relatively low. In the natural treatment, low WTD led to reduction of peat favorable environment for anoxic microbial activity and increase in CH₄ oxidation.

The significant interaction between substrate depth, placement season, and treatment on CH₄ emissions illustrated how these treatment characteristics had inconsistent effects on CH₄ emissions. Effluxes from the stockpiled 100 cm summer plot (plot #4) were greater than all other plots for both seasons. This plot largely resulted in a significant depth effect (100 cm > 50 cm = 15 cm). Since there was no significant difference in soil physical properties (bulk density and porosity) among summer and winter plots for either stockpiled or live treatments, it is

unlikely placement season was important from a mechanistic point of view. Instead, as discussed above, this plot had a relatively high %N compared to the other plots perhaps resulting in less microbial nutrient limitations. Similarly, since the second highest CH₄ emissions were from the live 15 cm summer plot (plot #6), substrate depth did not appear to be an important factor. Instead, this plot was also associated with greatest CO₂ emissions suggesting that activity of both aerobic and anaerobic microbes was enhanced in some way, perhaps by less acidic conditions than other plots.

6. CONCLUSIONS AND RECOMMENDATIONS

Results from this study clearly demonstrate the importance of soil and environmental characteristics on CO₂ and CH₄ emission rates and decomposition of the various substrates that were being tested for wetland reclamation. It was either shown directly or indirectly that CO₂ and CH₄ emissions from the soil are influenced by SOM quality and quantity, nutrient availability, temperature, VWC, water table depth, plant roots and microbial community. Specifically, the following hypotheses were addressed:

i. Plots with transplanted peat (live treatment) did have greater CO₂ and CH₄ emissions in the field and in laboratory incubations than plots with stockpiled peat mixed with mineral soil (stockpiled treatments) due to greater organic matter content. The lower C:N ratio suggested that the organic matter in the stockpiled peat had likely undergone significant decomposition while stored. When the C emissions from the laboratory experiment were normalized by the amount of organic C in the soil, the stockpiled soil mineralization rates were still lower than those of the live treatment suggesting it may consist of more recalcitrant organic material than the recently transplanted peat which presumably had some component of fresh plant material as well as humified peat.

ii. The natural treatment did not receive additional watering and had lower soil moisture compared to the experimental treatments. As expected, this was the only treatment with net CH₄ uptake. However, the saturated live treatment had higher CO₂ emission while the saturated stockpiled treatment had lower CO₂ emissions than the natural treatment. There was evidence that the surface peat of the natural

treatment was dry enough to restrict nutrient availability to microbes and inhibit decomposition. Further, C mineralization during incubation of the live treatment peat was low when compared to the live treatment suggesting some difference in SOM quality or microbe population characteristics that had a role in determining these differences. Finally, the regular irrigation of the live treatment plots with aerated water may have also limited the period of time conditions were anoxic in these plots and allowed for greater CO₂ emissions.

In this study, root exclusion collars did not significantly affect the CO₂ emissions. This may have been due to the difference in plant root biomass and also due to the substitution of autotrophic respiration with heterotrophic respiration. Density and composition of plant species between treatments were considerably different and accordingly the contribution of root autotrophic respiration to total soil respiration should have been greatest in the live and natural treatments with higher plant populations and variety in comparison to the stockpiled vegetation. But as surface roots had already been affected by the CT collar installation, more time may have been needed to observe the impact of severing roots in deeper layers or simply, deeper roots had little impact. However, there were significant differences in nutrients inside and outside collars. Nutrient rates were significantly lower outside of the collars as a result of plant root competition with probes for nutrient adsorption, while this competition inside the collars was limited to probes and microbes.

iii. Comparing CO₂ emissions recorded from different treatment depths in the field experiment showed that there were slightly higher emissions in the 15 cm

thick peat. This treatment effect was statistically significant when fluxes were measured with the flow-through chamber system, but not with the static chamber system. As a result, there was no strong evidence that decomposition rates were greater with shallower or deeper substrates. CH₄ fluxes in live and stockpiled treatments were significantly greater in the 15 cm and 100 cm thick respectively. The different emission could be attributed to generally significantly higher WTD in the 15 cm plots and consequently, potentially greater methanogenesis. For 100 cm stockpiled treatment higher emission rates can be attributed to the higher labile C in these plots vs. 15 and 50 cm depths.

iv. In experimental treatments, CO₂ and CH₄ effluxes increased with higher surface soil temperatures presumably due to the stimulation of microbial activity and the release of more nutrients for microbial activity (Winkler & Cherry, 1996). However, in the natural treatment, as soil temperature increased, soil moisture decreased. CO₂ emissions increased to some extent while CH₄ uptake increased in these plots. Variations in surface temperature were also greatest in the natural treatment where the relatively dry surface peat would have had reduced thermal admittance compared to the saturated experimental treatments regardless of differences in albedo.

v. In this study, although CH₄ emissions in the field were significantly greater in the saturated live and stockpiled treatments vs. dry natural treatment, in all cases they were relatively low. Although redox potential and oxygen levels were not measured in this study, it was possible that the regular addition of aerated water to

the experimental treatments did not allow the anoxic conditions to persist and maintain higher CH₄ production rates.

Comparisons between the aerobic decomposition rates from previous studies in boreal peatlands and the emission rates recorded from the present study show that respiration from these plots was not particularly high compared to respiration rates at natural peatlands. However, this may have been due to the high water table maintained at these plots. The rate of CO₂ emissions from these soils will likely increase if the water table drops in the future. This is most likely for the live treatment with its high potential biodegradability such that under drier conditions, a rapid loss of soil organic C is possible.

For anaerobic decomposition rates however, CH₄ emission rates recorded from this study was significantly lower which is mostly related to regular additions of aerated water and interruption in anoxic condition in the U-shaped cells and generally low WTD and consequently reduction in methanogenesis and increase in methanotrophic activities in the natural plots. However, to fully understand the factors driving these results and evaluate these reclamation treatments in terms of their ability to capture and store C, further investigation is necessary because based on reclamation experiences from the past (OSWWG, 2000) in the long-term, decomposition, vegetation growth and nutrient dynamics will continue to change, and particularly if the management of water levels, vegetation composition and substrate (by adding amendments) is varied. In particular, it was clear (visually) that maintaining water levels above the peat surface was not to the advantage of transplanted live peat vegetation. Further, these high water levels did not inhibit

CO₂ emissions from the live treatment and likely supported significant emissions of CH₄, which is a potent GHG with much higher potential for global warming and also ozone depletion in comparison to CO₂. Thus, as production of CO₂ and CH₄ effluxes are greatly influenced by water table depth, further study to select the optimum water level is highly recommended.

In this study the live treatment appeared to support vegetation growth much better in comparison to the rewetted mineral/peat stockpiled treatment. However, in other trials, mineral soils with amendments can successfully support wetland plant communities (Wieder et al., 2011).

Based on the results of previous reclamation studies, at least a 10 to 15 year timeframe is needed in order to establish, monitor and evaluate reclamation regimes (OSWWG, 2000). As most of the decomposition happens in top layers of the soil, it might be possible to improve the C sequestration by using appropriate watering regimes as mentioned above but also by cultivating hydrophytes, specifically those that can effectively produce recalcitrant substrates or can alter soil properties in favor of low decomposition rates (e.g. by lowering pH or increasing water retention capacity). Further research is recommended to investigate these possibilities.

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APPENDIX

Table A.T1. Results from statistical analyses of CO₂ emission measurements by flow-through system (Early: DOY 151 - 182, Late: DOY 188 - 230) based on full factorial of treatments, collar types, placement season and treatment depth in individual parts of the experiment. Asterisks denote significant differences ($\alpha = 0.05$). Results have been presented only for significant changes. Levels not connected by same letter are significantly different.

DOY 151 - 182						DOY 188 - 230									
Effect test: Treatment						Effect test: Treatment									
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F					
1		398.0		<0.0001*		1		506.9		<0.0001*					
Effect details						Effect details									
Live				A				Live				A			
Stockpiled					B			Stockpiled					B		
Effect test: Treatment depth						Effect test: Treatment depth									
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F					
2		9.5		0.0001*		2		5.4		0.0045*					
Effect details						Effect details									
15 cm				A				15 cm				A			
100 cm					B			50 cm				A	B		
50 cm					B			100 cm					B		
Effect test: Placement season						Effect test: Placement season									
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F					
1		4.5		0.0344*		1		29.6		<0.0001*					
Effect details						Effect details									
Summer				A				Summer				A			
Winter					B			Winter					B		
Effect test: Treatment * Treatment depth						Effect test: Treatment * Treatment depth									
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F					
2		14.7		<0.0001*		2		10.3		<0.0001*					
Effect details						Effect details									
Live, 15 cm				A				Live, 15 cm				A			
Live, 50 cm					B			Live, 50 cm				A			
Live, 100 cm					B			Live, 100 cm					B		
Stockpiled, 100 cm						C		Stockpiled, 100 cm						C	
Stockpiled, 15 cm						C	D	Stockpiled, 15 cm						C	
Stockpiled, 50 cm							D	Stockpiled, 50 cm						C	
Effect test: Treatment*Placement season						Effect test: Treatment*Placement season									
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F					
1		14.2		0.0002*		1		15.2		0.0001*					
Effect details						Effect details									
Live, Summer				A				Live, Summer				A			
Live, Winter					B			Live, Winter					B		
Stockpiled, Winter						C		Stockpiled, Summer						C	
Stockpiled, Summer						C		Stockpiled, Winter						C	

DOY 151 - 182						DOY 188 - 230					
Effect test:						Effect test:					
Treatment depth*Placement season						Treatment depth*Placement season					
DF		F Ratio		Prob > F		DF		F Ratio		Prob > F	
2		10.1		<0.0001*		2		6.6		0.0014*	
Effect details						Effect details					
15 cm, Summer				A		15 cm, Summer				A	
100 cm, Winter					B	50 cm, Summer				A	B
15 cm, Winter					B	100 cm, Summer					B C
50 cm, Summer					B	100 cm, Winter					C
50 cm, Winter					B	15 cm, Winter					C
100 cm, Summer					B	50 cm, Winter					C
Effect test:						Effect test:					
Treatment depth*Collar type						Treatment depth*Collar type					
DF		F Ratio		Prob > F		Not Significant					
2		6.0		0.0027*							
Effect details											
15 cm, CT#1				A							
15 cm, CT#2				A	B						
100 cm, CT#2				A	B						
50 cm, CT#2					B C						
50 cm, CT#1					B C						
100 cm, CT#1					C						
Effect test:						Effect test:					
Treatment * Treatment depth*Placement						Treatment * Treatment depth*Placement					
DF		F Ratio		Prob > F		DF		F Ratio		Prob> F	
2		27.4		<0.0001*		2		42.0		<0.0001*	
Effect details						Effect details					
Live, 15 cm, Summer				A		Live, 15 cm, Summer				A	
Live, 50 cm, Summer					B	Live, 50 cm, Summer					B
Live, 100 cm, Winter					B	Live, 100 cm, Winter					C
Live, 50 cm, Winter					B C	Live, 50 cm, Winter					C
Live, 15 cm, Winter					B C	Live, 15 cm, Winter					C
Live, 100 cm, Summer					C D	Live, 100 cm, Summer					C D
Stockpiled, 100 cm, Summer					D E	Stockpiled, 100 cm, Summer					D E
Stockpiled, 15 cm, Winter					E	Stockpiled, 15 cm, Winter					E F
Stockpiled, 100 cm, Winter					E	Stockpiled, 50 cm, Summer					E F
Stockpiled, 50 cm, Winter					E	Stockpiled, 50 cm, Winter					E F
Stockpiled, 15 cm, Summer					E	Stockpiled, 15 cm, Summer					E F
Stockpiled, 50 cm, Summer					E	Stockpiled, 100 cm, Winter					F
Effect test: Placement season*Collar type						Effect test: Placement season*Collar type					
DF		F Ratio		Prob > F		Not Significant					
1		11.8		0.0007*							
Effect details											
Summer, CT#1				A							
Winter, CT#2				A							
Summer, CT#2				A	B						
Winter, CT#1					B						

DOY 151 - 182				DOY 188 - 230					
Effect test: Treatment depth*Placement season*Collar type				Effect test: Treatment depth*Placement season*Collar type					
DF	F Ratio	Prob > F		Not Significant					
2	12.9	<0.0001*							
Effect details									
15 cm, Summer, CT#1	A								
100 cm, Winter, CT#2	A	B							
15 cm, Summer, CT#2		B	C						
50 cm, Summer, CT#2		B	C						
15 cm, Winter, CT#2		B	C					D	
100 cm, Summer, CT#1			C					D	
15 cm, Winter, CT#1			C					D	
50 cm, Winter, CT#1			C					D	
50 cm, Winter, CT#2			C					D	
50 cm, Summer, CT#1			C					D	
100 cm, Summer, CT#2			C					D	
100 cm, Winter, CT#1								D	

Table A.T2. Results from statistical analyses of CH₄ and CO₂ emission measurements by static system (Early: DOY 151 - 182, Late: DOY 188 - 230) based on treatments, collar types, and their interactions, also based on placement and soil depth and their interactions (for live and stockpiled treatments in the U-shaped cells) in individual parts of the experiment. Asterisks denote significant differences ($\alpha = 0.05$). Results have been presented only for significant changes. Levels not connected by same letter are significantly different.

CO ₂ emission ($\mu\text{mol m}^{-2} \text{s}^{-1}$)									
DOY 151-182					DOY 188 - 230				
Effect test: Treatment					Effect test: Treatment				
DF	F Ratio	Prob > F			DF	F Ratio	Prob > F		
2	38.2	<0.0001*			2	117.6	<0.0001*		
Effect details					Effect details				
Live		A			Live		A		
Natural			B		Natural			B	
Stockpiled			B		Stockpiled			C	
Effect test: Treatment*Collar type					Effect test: Treatment*Collar type				
Not Significant					DF	F Ratio	Prob > F		
					2	5.0	0.0073*		
					Effect details				
					Live, RE		A		
					Live, CT			B	
					Natural, CT			C	
					Natural, RE			C	D
					Stockpiled, CT				D
					Stockpiled, RE				D
Effect test: Placement season					Effect test: Placement season				
Winter		A			Winter		A		
Summer			B		Summer			B	
Effect test: Treatment*depth					Effect test: Treatment*depth				
Live, 15 cm		A			Live, 50 cm		A		
Live, 50 cm		A	B		Live, 15 cm		A		
Live, 100 cm			B	C	Live, 100 cm			B	
Stockpiled, 100 cm				C	Stockpiled, 100 cm				C
Stockpiled, 15 cm				D	Stockpiled, 50 cm				C
Stockpiled, 50 cm				D	Stockpiled, 15 cm				C
Effect test: Treatment* Placement season					Effect test: Treatment* Placement season				
Not Significant					Not Significant				
Effect test: Placement season* depth					Effect test: Placement season* depth				
100 cm, Winter		A			Not Significant				
50 cm, Winter		A							
15 cm, Winter		A							
15 cm, Summer		A							
50 cm, Summer		A	B						
100 cm, Summer			B						

Effect test:					Effect test:				
Placement season* depth*Treatment					Placement season* depth*Treatment				
Live,50 cm,Winter	A				Live,15 cm,Summer	A			
Live,15 cm,Summer	A				Live,50 cm,Summer	A			
Live,100 cm,Winter	A				Live,50 cm,Winter	B			
Live,15 cm,Winter	A	B			Live,100 cm,Summer	B			
Live,50 cm,Summer	A	B	C		Live,100 cm,Winter	B			
Stockpiled,15 cm,Winter		B	C	D	Live,15 cm,Winter	B			
Stockpiled,100 cm,Winter		B	C	D	Stockpiled,100 cm,Summer	B	C		
Live,100 cm,Summer			C	D	Stockpiled,50 cm,Summer		C	D	
Stockpiled,100 cm,Summer			C	D	Stockpiled,15 cm,Winter			D	
Stockpiled,50 cm,Winter			C	D	Stockpiled,15 cm,Summer			D	
Stockpiled,50 cm,Summer				D	Stockpiled,50 cm,Winter			D	
Stockpiled,15 cm,Summer				D	Stockpiled,100 cm,Winter			D	
CH₄ emission (nmol m⁻² s⁻¹)									
DOY 151-182					DOY 189 - 230				
Effect test: Treatment					Effect test: Treatment				
DF	F Ratio	Prob > F			DF	F Ratio	Prob > F		
2	18.6	<0.0001*			2	97.4	<0.0001*		
Effect details					Effect details				
Live	A				Live	A			
Natural		B			Natural		B		
Stockpiled			C		Stockpiled			C	
Effect test: Collar type					Effect test: Collar type				
Not Significant					Not Significant				
Effect test: Treatment*Collar type					Effect test: Treatment*Collar type				
Not Significant					Not Significant				
Effect test: Depth					Effect test: Depth				
100 cm	A				100 cm	A			
15 cm	A	B			50 cm		B		
50 cm		B			15 cm		B		
Effect test: Placement season					Effect test: Placement season				
Summer	A				Summer	A			
Winter		B			Winter		B		
Effect test: Treatment*depth					Effect test: Treatment*depth				
Live,15 cm	A				Live,15 cm	A			
Stockpiled,100 cm	A				Stockpiled,100 cm	A	B		
Live,50 cm	A				Live,50 cm		B		
Live,100 cm		B			Live,100 cm		B		
Stockpiled,50 cm		B			Stockpiled,50 cm			C	
Stockpiled,15 cm		B			Stockpiled,15 cm			D	

Effect test: Placement season* depth						Effect test: Placement season* depth					
Not Significant						100 cm, Summer	A				
						50 cm, Summer	A	B			
						100 cm, Winter		B	C		
						15 cm, Summer			C		
						15 cm, Winter			C	D	
						50 cm, Winter				D	
Effect test: Placement season* depth*Treatment						Effect test: Placement season* depth*Treatment					
Stockpiled, 100 cm, Summer	A					Stockpiled, 100 cm, Summer	A				
Live, 15 cm, Summer	A	B				Live, 15 cm, Summer	A				
Live, 50 cm, Summer		B	C			Live, 50 cm, Summer	A				
Live, 15 cm, Winter			C	D		Live, 15 cm, Winter	A	B			
Live, 100 cm, Winter			C	D	E	Live, 100 cm, Summer	A	B	C		
Live, 50 cm, Winter			C	D	E	Live, 100 cm, Winter	A	B	C		
Stockpiled, 100 cm, Winter				D	E	Stockpiled, 50 cm, Summer		B	C		
Live, 100 cm, Summer				D	E	Stockpiled, 100 cm, Winter		B	C		
Stockpiled, 50 cm, Summer				D	E	Live, 50 cm, Winter			C	D	
Stockpiled, 50 cm, Winter					E	Stockpiled, 50 cm, Winter				D	E
Stockpiled, 15 cm, Winter					E	Stockpiled, 15 cm, Summer					E
Stockpiled, 15 cm, Summer					E	Stockpiled, 15 cm, Winter					E

Table A.T3. Spearman's correlation coefficients showing the relationships between CO₂ efflux (flow-through measurement system) and environmental variables (columns) within each treatment. P-values are given in brackets and significant (P < 0.05) relationships noted by Asterisks.

	DOY	Volumetric water content (%)	Water table depth (cm)	Thermocouple profile (°C)				
				2 cm depth	5 cm depth	10 cm depth	20 cm depth	50 cm depth
Live	151 - 182	-0.1428 (0.1023)	-0.0305 (0.7282)	0.2715 (0.0016*)	0.2010 (0.0208*)	0.3323 (<0.0001*)	0.3918 (0.0002*)	0.3098 (0.0033*)
	188 - 230	-0.1715 (0.0262*)	-0.0933 (0.2291)	0.3395 (<0.0001*)	0.2198 (0.0042*)	0.2312 (0.0026*)	0.1708 (0.0717)	-0.2964 (0.0015*)
	151 - 230	-0.1252 (0.0301*)	-0.0554 (0.3386)	0.3161 (<0.0001*)	0.2348 (<0.0001*)	0.2409 (<0.0001*)	0.2441 (0.0005*)	0.0904 (0.2028)
Stockpiled	151 - 182	-0.0387 (0.6599)	-0.1897 (0.0294*)	0.2328 (0.0072*)	0.0596 (0.4974)	-0.0964 (0.2717)	0.0046 (0.9664)	0.1313 (0.2228)
	188 - 230	0.1603 (0.0416*)	-0.1834 (0.0195*)	0.1719 (0.0287*)	0.1676 (0.0330*)	0.0709 (0.3714)	0.0165 (0.8666)	-0.0481 (0.6245)
	151 - 230	0.0103 (0.8606)	-0.1918 (0.0009*)	0.0697 (0.2333)	-0.0635 (0.2781)	-0.2143 (0.0002*)	-0.2337 (0.0010*)	-0.2646 (0.0002*)
All treatments	151 - 182	-	-	0.0017 (0.9775)	-0.0809 (0.1902)	-0.0954 (0.1221)	-0.1152 (0.1279)	-0.1913 (0.0110*)
	188 - 230	-	-	0.2134 (<0.0001*)	0.0487 (0.3777)	-0.0232 (0.6749)	-0.0828 (0.2234)	-0.4250 (<0.0001*)
	151 - 230	-	-	0.1076 (0.0087*)	-0.0133 (0.7455)	-0.0768 (0.0613)	-0.0997 (0.0480*)	-0.2103 (<0.0001*)

Table A.T4. Spearman's correlation coefficients showing the relationships between CO₂ and CH₄ efflux (Static measurement system) and environmental variables (columns) within each treatment. P-values are given in brackets and significant (P < 0.05) relationships noted by Asterisks.

FACTOR		DOY	Volumetric Water content (%)	Water table depth (cm)	Thermocouple profile (°C)				
					2 cm depth	5 cm depth	10 cm depth	20 cm depth	50 cm depth
Live	CH ₄ (nmol m ⁻² s ⁻¹)	151 - 182	-0.1052 (0.3411)	0.0734 (0.5070)	0.3195 (0.0031*)	0.3231 (0.0027*)	0.4029 (0.0001*)	0.5617 (<0.0001*)	0.4987 (<0.0001*)
		188 - 230	-0.0442 (0.6496)	0.3049 (0.0013*)	0.1984 (0.0395*)	0.1069 (0.2708)	0.1857 (0.0544)	0.3972 (0.0007*)	0.3295 (0.0061*)
		151 - 230	0.0516 (0.4770)	0.2094 (0.0036*)	0.4027 (<0.0001*)	0.4344 (<0.0001*)	0.5243 (<0.0001*)	0.6892 (<0.0001*)	0.6697 (<0.0001*)
	CO ₂ (μmol m ⁻² s ⁻¹)	151 - 182	-0.2000 (0.0681)	-0.4384 (<0.0001*)	0.4166 (<0.0001*)	0.4207 (<0.0001*)	0.4297 (<0.0001*)	0.3328 (0.0122*)	0.2338 (0.0829)
		188 - 230	-0.3585 (<0.0001*)	0.0984 (0.2872)	0.6227 (<0.0001*)	0.5458 (<0.0001*)	0.5400 (<0.0001*)	0.3433 (0.0020*)	-0.2228 (0.0514)
		151 - 230	-0.2546 (0.0002*)	-0.1171 (0.0961)	0.5312 (<0.0001*)	0.4768 (<0.0001*)	0.4603 (<0.0001*)	0.3760 (<0.0001*)	0.1757 (0.0431*)
Stockpiled	CH ₄ (nmol m ⁻² s ⁻¹)	151 - 182	0.5930 (<0.0001*)	-0.1616 (0.1419)	0.1887 (0.0856)	0.2675 (0.0139*)	0.4351 (<0.0001*)	0.5370 (<0.0001*)	0.4345 (0.0008*)
		188 - 230	0.5474 (<0.0001*)	-0.3015 (0.0023*)	0.0783 (0.4388)	0.0491 (0.6274)	0.1737 (0.0840)	0.3053 (0.0142*)	0.2654 (0.0341*)
		151 - 230	0.5573 (<0.0001*)	-0.2117 (0.0039*)	0.1527 (0.0386*)	0.1721 (0.0195*)	0.2693 (0.0002*)	0.4773 (<0.0001*)	0.4430 (<0.0001*)
	CO ₂ (μmol m ⁻² s ⁻¹)	151 - 182	-0.1631 (0.1382)	-0.2641 (0.0152*)	0.4257 (<0.0001*)	0.3666 (0.0006*)	0.2864 (0.0083*)	0.2204 (0.1027)	0.1761 (0.1941)
		188 - 230	0.1217 (0.1891)	-0.2966 (0.0011*)	0.4175 (<0.0001*)	0.4307 (<0.0001*)	0.4219 (<0.0001*)	0.4491 (<0.0001*)	-0.1414 (0.2169)
		151 - 23	-0.0331 (0.6396)	-0.3083 (<0.0001*)	0.2818 (<0.0001*)	0.1511 (0.0318*)	0.0352 (0.6189)	0.0501 (0.5655)	-0.2203 (0.0105*)

FACTOR		DOY	Volumetric Water content (%)	Water table depth (cm)	Thermocouple profile (°C)				
					2 cm depth	5 cm depth	10 cm depth	20 cm depth	50 cm depth
Natural	CH ₄ (nmol m ⁻² s ⁻¹)	151 - 182	0.0074 (0.9702)	-	-0.1283 (0.5152)	-0.0093 (0.9624)	-0.2092 (0.2854)	-0.1131 (0.5666)	-0.2026 (0.3012)
		188 - 230	0.4379 (0.0076*)	-	-0.0317 (0.8543)	0.0462 (0.7890)	-0.0364 (0.8329)	0.0267 (0.8774)	0.1373 (0.4247)
		151 - 230	0.5208 (0.0001*)	-	-0.2319 (0.0652)	-0.2380 (0.0583)	-0.4757 ($<0.0001^*$)	-0.5109 ($<0.0001^*$)	-0.5479 ($<0.0001^*$)
	CO ₂ (μmol m ⁻² s ⁻¹)	151 - 182	-0.2966 (0.1253)	-	0.5243 (0.0042*)	0.5287 (0.0038*)	0.3453 (0.0719)	0.3459 (0.0714)	0.3777 (0.0475*)
		188 - 230	0.2789 (0.0855)	-	0.3952 (0.0128*)	0.3770 (0.0180*)	0.3809 (0.0167*)	0.2388 (0.1432)	0.2639 (0.1046)
		151 - 230	0.0536 (0.6667)	-	0.4138 (0.0005*)	0.3929 (0.0010*)	0.2614 (0.0326*)	0.1339 (0.2800)	0.1051 (0.3974)
All treatments	CH ₄ (nmol m ⁻² s ⁻¹)	151 - 182	0.5237 ($<0.0001^*$)	0.1734 (0.0246*)	0.0247 (0.7312)	0.0740 (0.3029)	0.1498 (0.0361*)	0.4570 ($<0.0001^*$)	0.4310 ($<0.0001^*$)
		188 - 230	0.7244 ($<0.0001^*$)	0.3181 ($<0.0001^*$)	-0.0416 (0.5177)	-0.0449 (0.4855)	0.0570 (0.3756)	0.4936 ($<0.0001^*$)	0.5393 ($<0.0001^*$)
		151 - 230	0.6665 ($<0.0001^*$)	0.2535 ($<0.0001^*$)	0.0390 (0.4140)	0.0838 (0.0792)	0.1758 ($<0.0002^*$)	0.5162 ($<0.0001^*$)	0.4948 ($<0.0001^*$)
	CO ₂ (μmol m ⁻² s ⁻¹)	151 - 182	0.2771 ($<0.0001^*$)	0.1282 (0.0976)	0.2793 ($<0.0001^*$)	0.2218 (0.0018*)	0.1441 (0.0438*)	0.0845 (0.3211)	0.0393 (0.6448)
		188 - 230	0.4593 ($<0.0001^*$)	0.4008 ($<0.0001^*$)	0.3476 ($<0.0001^*$)	0.2187 (0.0003*)	0.1719 (0.0042*)	0.0717 (0.3176)	-0.3049 ($<0.0001^*$)
		151 - 230	0.3965 ($<0.0001^*$)	0.2997 ($<0.0001^*$)	0.2839 ($<0.0001^*$)	0.1636 (0.0004*)	0.0919 (0.0460*)	0.0549 (0.3154)	-0.1003 (0.0672)

Table A.T5. ANOVA results from separate statistical analyses of surface (0-5 cm) and deep (10-12 cm) nutrient availability, based on treatments, collar and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Tukey HSD test results for all significant effects tests are presented in Appendix Table A.T7.

Ca²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	17.21	<0.0001*	Treatment	2	20.18	<0.0001*	Treatment	2	11.01	0.0002*
Collar	2	2.56	0.09	Collar	2	2.61	0.08	Collar	2	0.63	0.53
Treatment × Collar	4	0.89	0.47	Treatment × Collar	4	1.39	0.25	Treatment × Collar	4	0.49	0.74
Mg²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	6.49	0.0042*	Treatment	2	8.09	0.0014*	Treatment	2	6.84	0.0033*
Collar	2	2.87	0.07	Collar	2	2.22	0.12	Collar	2	0.23	0.78
Treatment × Collar	4	1.24	0.31	Treatment × Collar	4	1.45	0.23	Treatment × Collar	4	0.37	0.82
K⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	29.41	<0.0001*	Treatment	2	23.87	<0.0001*	Treatment	2	239.08	<0.0001*
Collar	2	3.44	0.0438*	Collar	2	3.79	0.0327*	Collar	2	1.05	0.35
Treatment × Collar	4	1.44	0.24	Treatment × Collar	4	1.64	0.18	Treatment × Collar	4	0.36	0.82

P⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	5.80	0.0069*	Treatment	2	6.25	0.0050*	Treatment	2	20.54	<0.0001*
Collar	2	1.71	0.1955	Collar	2	1.99	0.15	Collar	2	10.09	0.0004*
Treatment × Collar	4	0.63	0.6437	Treatment × Collar	4	0.76	0.55	Treatment × Collar	4	2.3	0.07
Fe^{2+,3+} (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	1.49	0.23	Treatment	2.21	0.12	2.21	Treatment	2	62.15	<0.0001*
Collar	2	0.37	0.69	Collar	0.64	0.53	0.64	Collar	2	0.32	0.72
Treatment × Collar	4	0.30	0.87	Treatment × Collar	0.76	0.55	0.76	Treatment × Collar	4	0.06	0.99
Mn²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	2.03	0.14	Treatment	2	6.08	0.0056*	Treatment	2	19.05	<0.0001*
Collar	2	0.16	0.84	Collar	2	0.07	0.92	Collar	2	0.45	0.63
Treatment × Collar	4	0.36	0.83	Treatment × Collar	4	0.07	0.98	Treatment × Collar	4	0.34	0.84

Cu²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	0.30	0.73	Treatment	2	0.49	0.61	Treatment	2	8.46	0.001*
Collar	2	0.38	0.68	Collar	2	0.52	0.59	Collar	2	0.06	0.93
Treatment × Collar	4	0.54	0.70	Treatment × Collar	4	0.48	0.74	Treatment × Collar	4	0.24	0.91
Zn²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	8.06	0.0014*	Treatment	2	3.56	0.0395*	Treatment	2	61.76	<0.0001*
Collar	2	0.59	0.55	Collar	2	1.33	0.27	Collar	2	0.88	0.42
Treatment × Collar	4	1.30	0.28	Treatment × Collar	4	0.97	0.43	Treatment × Collar	4	0.8	0.53
B⁻ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	3.67	0.0361*	Treatment	2	19.20	<0.0001*	Treatment	2	29.1	<0.0001*
Collar	2	0.09	0.90	Collar	2	0.36	0.70	Collar	2	0.31	0.73
Treatment × Collar	4	0.52	0.71	Treatment × Collar	4	0.64	0.63	Treatment × Collar	4	0.59	0.66

S⁻ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	47.74	<0.0001*	Treatment	2	33.68	<0.0001*	Treatment	2	24.045	<0.0001*
Collar	2	0.79	0.46	Collar	2	1.62	0.21	Collar	2	1.57	0.22
Treatment × Collar	4	0.09	0.98	Treatment × Collar	4	0.34	0.84	Treatment × Collar	4	0.22	0.92
Pb²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	4.17	0.0241*	Treatment	2	5.65	0.0077*	Treatment	2	32.37	<0.0001*
Collar	2	0.73	0.48	Collar	2	0.84	0.44	Collar	2	1.29	0.28
Treatment × Collar	4	0.25	0.90	Treatment × Collar	4	0.30	0.87	Treatment × Collar	4	1.42	0.24
Al³⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	11.55	0.0002*	Treatment	2	3.67	0.0362*	Treatment	2	4.85	0.014*
Collar	2	0.07	0.93	Collar	2	1.05	0.35	Collar	2	0.14	0.86
Treatment × Collar	4	1.43	0.24	Treatment × Collar	4	1.09	0.37	Treatment × Collar	4	0.74	0.56

Cd²⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) - Early				Surface probe (~ 5 cm) - Late				Deep probe (~ 12 cm)			
Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob >F	Effect	DF	F Ratio	Prob > F
Treatment	2	2.57	0.09	Treatment	2	2.56	0.09	Treatment	2	2.25	0.12
Collar	2	1.11	0.34	Collar	2	2.86	0.0715	Collar	2	2.95	0.06
Treatment				Treatment				Treatment			
×	4	0.72	0.58	×	4	1.18	0.33	×	4	1.44	0.24
Collar				Collar				Collar			

Table A.T6. Results from separate statistical analyses of surface (~ 5 cm) and deep (~ 12 cm) nutrient availability, based on treatments, collar and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Results have been presented only for significant changes. Levels not connected by same letter are significantly different.

Total N (µg/10cm²/82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	Not Significant			DF	F Ratio	Prob >F
2	14.18	<0.0001*				2	34.27	<0.0001*
Effect details						Effect details		
Live	A					Live	A	
Stockpiled		B				Stockpiled		B
Natural		B	Natural		B			
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			DF	F Ratio	Prob >F
						2	7.2	0.0025*
						Effect details		
						RE	A	
						CT		B
Out		B						
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
NO3-N (µg/10cm²/82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
Not Significant			Not Significant			Not Significant		

Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			DF	F Ratio	Prob >F
						2	8.31	0.0012*
						Effect details		
						RE	A	
						CT	A	
						Out		B
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
NH4 ⁺ -N (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	15.35	<0.0001*	2	7.58	0.0019*	2	30.97	<0.0001*
Effect details			Effect details			Effect details		
Live	A		Live	A		Live	A	
Stockpiled		B	Stockpiled		B	Stockpiled		B
Natural		B	Natural		B	Natural		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			DF	F Ratio	Prob > F
						2	5.08	0.0119*
						Effect details		
						RE	A	
						Out	A	B
						CT		B

Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
Ca ²⁺ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	17.21	<0.0001*	2	20.1	<0.0001*	2	11.01	0.0002*
Effect details			Effect details			Effect details		
Stockpiled	A		Stockpiled	A		Stockpiled	A	
Live		B	Live		B	Natural	A	B
Natural		B	Natural		B	Live		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
Mg ²⁺ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	6.49	0.0042*	2	8.09	0.0014*	2	6.84	0.0033*
Effect details			Effect details			Effect details		
Stockpiled	A		Stockpiled	A		Stockpiled	A	
Live	A	B	Live	A	B	Live	A	B
Natural		B	Natural		B	Natural		B

Effect test: Collar				Effect test: Collar				Effect test: Collar			
Not Significant				Not Significant				Not Significant			
Effect test: Treatment* Collar				Effect test: Treatment* Collar				Effect test: Treatment* Collar			
Not Significant				Not Significant				Not Significant			
K ⁺ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) – Early				Surface probe (~ 5 cm) – Late				Deep probe (~ 12 cm)			
Effect test: Treatment				Effect test: Treatment				Effect test: Treatment			
DF	F Ratio	Prob >F		DF	F Ratio	Prob >F		DF	F Ratio	Prob > F	
2	29.41	<0.0001*		2	23.87	<0.0001*		2	239.08	<0.0001*	
Effect details				Effect details				Effect details			
Live		A		Live		A		Live		A	
Natural			B	Natural		A		Natural			B
Stockpiled			C	Stockpiled			B	Stockpiled			C
Effect test: Collar				Effect test: Collar				Effect test: Collar			
DF	F Ratio	Prob >F		DF	F Ratio	Prob >F		Not Significant			
2	3.44	0.0438*		2	3.79	0.0327*					
Effect details				Effect details							
CT#1		A		CT		A					
CT#2		A	B	RE		A	B				
Out			B	Out			B				
Effect test: Treatment* Collar				Effect test: Treatment* Collar				Effect test: Treatment* Collar			
Not Significant				Not Significant				Not Significant			
P ⁻ (µg/10cm²/82 days)											
Surface probe (~ 5 cm) – Early				Surface probe (~ 5 cm) – Late				Deep probe (~ 12 cm)			
Effect test: Treatment				Effect test: Treatment				Effect test: Treatment			

DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	5.80	0.0069*	2	6.25	0.0050*	2	20.54	<0.0001*
Effect details			Effect details			Effect details		
Natural	A		Natural	A		Live	A	
Stockpiled		B	Stockpiled		B	Stockpiled		B
Live		B	Live		B	Natural		C
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			DF	F Ratio	Prob > F
						2	10.09	0.0004*
						Effect details		
						RE	A	
						CT	A	B
						Out		B
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
Fe ^{2+,3+} (µg/10cm²/82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
Not Significant			Not Significant			DF	F Ratio	Prob > F
						2	62.15	<0.0001*
						Effect details		
						Live	A	
						Stockpiled		B
						Natural		C

Effect test: Collar	Effect test: Collar			Effect test: Collar		
Not Significant	Not Significant			Not Significant		
Effect test: Treatment* Collar	Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant	Not Significant			Not Significant		
Mn ²⁺ (µg/10cm ² /82 days)						
Surface probe (~ 5 cm) – Early	Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Not Significant	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
	2	6.08	0.0056*	2	19.05	<0.0001*
	Effect details			Effect details		
	Live		A	Live		A
	Stockpiled		B	Stockpiled		A
	Natural		B	Natural		B
Effect test: Collar	Effect test: Collar			Effect test: Collar		
Not Significant	Not Significant			Not Significant		
Effect test: Treatment* Collar	Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant	Not Significant			Not Significant		
Cu ²⁺ (µg/10cm ² /82 days)						
Surface probe (~ 5 cm) – Early	Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment	Effect test: Treatment			Effect test: Treatment		
Not Significant	Not Significant			DF	F Ratio	Prob > F
				2	8.46	0.001*
				Effect details		
				Natural		A
				Stockpiled		B
			Live		B	

Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
Zn ²⁺ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	8.06	0.0014*	2	3.56	0.039*	2	61.76	<0.0001*
Effect details			Effect details			Effect details		
Live	A		Live	A	B	Live	A	
Natural	A	B	Natural	A		Natural	A	
Stockpiled		B	Stockpiled		B	Stockpiled		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
B ⁻ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	3.67	0.0361*	2	19.20	<0.0001*	2	29.1	<0.0001*

Effect details			Effect details			Effect details		
Stockpiled	A		Stockpiled	A		Live	A	
Live		B	Live		B	Natural	A	
Natural	A	B	Natural		B	Stockpiled		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
S ⁻ (µg/10cm²/82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	47.74	<0.0001*	2	33.68	<0.0001*	2	24.045	<0.0001*
			Effect details			Effect details		
Stockpiled	A		Stockpiled	A		Natural	A	
Natural		B	Natural		B	Stockpiled	A	
Live		B	Live		B	Live		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		

Pb ²⁺ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	4.17	0.0241*	2	5.65	<0.0077*	2	32.37	<0.0001*
Effect details			Effect details			Effect details		
Live	A		Live	A		Live	A	
Stockpiled		B	Stockpiled		B	Natural		B
Natural		B	Natural		B	Stockpiled		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		
Effect test: Treatment* Collar			Effect test: Treatment* Collar			Effect test: Treatment* Collar		
Not Significant			Not Significant			Not Significant		
Al ³⁺ (µg/10cm ² /82 days)								
Surface probe (~ 5 cm) – Early			Surface probe (~ 5 cm) – Late			Deep probe (~ 12 cm)		
Effect test: Treatment			Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob >F	DF	F Ratio	Prob >F	DF	F Ratio	Prob > F
2	11.55	0.0002*	2	3.67	0.0362*	2	4.85	0.014*
Effect details			Effect details			Effect details		
Natural	A		Natural	A		Natural	A	
Live	A		Live	A	B	Stockpiled	A	B
Stockpiled		B	Stockpiled		B	Live		B
Effect test: Collar			Effect test: Collar			Effect test: Collar		
Not Significant			Not Significant			Not Significant		

Effect test: Treatment* Collar	Effect test: Treatment* Collar	Effect test: Treatment* Collar
Not Significant	Not Significant	Not Significant
Cd²⁺ (µg/10cm²/82 days)		
Surface probe (~ 5 cm) – Early	Surface probe (~ 5 cm) – Late	Deep probe (~ 12 cm)
Effect test: Treatment	Effect test: Treatment	Effect test: Treatment
Not Significant	Not Significant	Not Significant
Effect test: Collar	Effect test: Collar	Effect test: Collar
Not Significant	Not Significant	Not Significant
Effect test: Treatment* Collar	Effect test: Treatment* Collar	Effect test: Treatment* Collar
Not Significant	Not Significant	Not Significant

Table A.T7. Spearman's correlation coefficients showing the relationships between environmental factors with nutrient availability in probes in ~12 cm depth. P-values are given in brackets and significant ($P < 0.05$) relationships noted by *.

	Nutrient $\mu\text{g}/10\text{cm}^2/82$ days	Probes in ~ 12 cm depth			Nutrient $\mu\text{g}/10\text{cm}^2/82$ days	Probes in ~ 12 cm depth			Nutrient $\mu\text{g}/10\text{cm}^2/82$ days	Probes in ~ 12 cm depth	
		Spearman ρ	Prob> ρ			Spearman ρ	Prob> ρ			Spearman ρ	Prob> ρ
Average volumetric water Content (%)	Total N	0.77	<0.0001*	Average temperature (°C)	Total N	-0.31	0.0465*	Average water table depth (cm)	Total N	0.42	0.0111*
	NO3-N	0.22	0.16		NO3-N	-0.24	0.13		NO3-N	0.19	0.28
	NH4-N	0.74	<0.0001*		NH4-N	-0.27	0.09		NH4-N	0.42	0.0100*
	Ca	-0.63	<0.0001*		Ca	-0.18	0.24		Ca	-0.41	0.0136*
	Mg	0.05	0.76		Mg	0.34	0.0291*		Mg	-0.33	0.05
	K	0.65	<0.0001*		K	0.31	0.0435*		K	0.59	0.0001*
	P	0.53	0.0003*		P	-0.48	0.0015*		P	0.15	0.40
	Fe	0.86	<0.0001*		Fe	0.10	0.52		Fe	0.28	0.09
	Mn	0.62	<0.0001*		Mn	-0.11	0.48		Mn	-0.22	0.20
	Cu	-0.67	<0.0001*		Cu	0.09	0.58		Cu	-0.15	0.40
	Zn	0.46	0.0021*		Zn	-0.10	0.54		Zn	0.64	<0.0001*
	B	-0.37	0.0172*		B	-0.35	0.0252*		B	-0.53	0.0008*
	S	-0.73	<0.0001*		S	0.48	0.0012*		S	-0.30	0.08
	Pb	0.66	<0.0001*		Pb	0.06	0.73		Pb	0.45	0.0055*
	Al	-0.57	<0.0001*		Al	-0.33	0.0314*		Al	-0.10	0.55
	Cd				Cd	0.08	0.61		Cd	-0.15	0.37

Table A.T8. Spearman's correlation coefficients showing the relationships between CO₂ and CH₄ effluxes with nutrient availability in deep probes. P-values are given in brackets and significant (P < 0.05) relationships noted by *.

Nutrient	Flow through system CO ₂ (μmol m ⁻² s ⁻¹)		Static chamber system CO ₂ (μmol m ⁻² s ⁻¹)		Static chamber system CH ₄ (nmol m ⁻² s ⁻¹)	
	Spearman ρ	Prob> ρ	Spearman ρ	Prob> ρ	Spearman ρ	Prob> ρ
Total N	0.67	0.0003*	0.69	<0.0001*	0.63	0.0004*
NO ₃ -N	0.37	0.08	0.41	0.0316*	0.12	0.53
NH ₄ -N	0.65	0.0005*	0.63	0.0003*	0.63	0.0003*
Ca	-0.52	0.0095*	-0.55	0.0024*	-0.51	0.0056*
Mg	-0.23	0.28	-0.29	0.13	0.11	0.58
K	0.82	<0.0001*	0.87	<0.0001*	0.48	0.0098*
P	0.21	0.32	0.22	0.26	0.49	0.0079*
Fe	0.61	0.0014*	0.57	0.0015*	0.68	<0.0001*
Mn	0.12	0.56	0.13	0.51	0.56	0.0018*
Cu	-0.07	0.73	-0.16	0.42	-0.61	0.0006*
Zn	0.74	<0.0001*	0.75	<0.0001*	0.26	0.18
B	-0.68	0.0003*	-0.70	<0.0001*	-0.25	0.19
S	-0.49	0.0148*	-0.57	0.0016*	-0.62	0.0004*
Pb	0.62	0.0011*	0.63	0.0003*	0.51	0.0058*
Al	-0.17	0.44	-0.24	0.22	-0.50	0.0063*
Cd	-0.17	0.43	-0.22	0.27	-0.20	0.30

Table A.T9. Tukey HSD test results from statistical analyses of mineralized C respired from treatments during the incubation period, based on analysis of treatments (live, stockpiled, and natural), soil sample depths (0-5, 5-10 cm), and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Results have been presented only for significant changes. Levels not connected by same letter are significantly different.

Mineralized C (mg C-CO ₂ g ⁻¹ dry soil d ⁻⁹⁷)					Mineralized C (mg C-CO ₂ g ⁻¹ C d ⁻⁹⁷)							
Effect test: Treatment					Effect test: Treatment							
DF		F Ratio		Prob > F	DF		F Ratio		Prob > F			
		10.2		0.0007*	2		7.5		0.0033*			
Effect details					Effect details							
Live				A				Live	A			
Stockpiled					B			Stockpiled		B		
Natural					B			Natural		B		
Effect test: Sample depth					Effect test: Treatment depth							
Not Significant					Not Significant							
Effect test: Treatment * Sample depth					Effect test: Treatment * Treatment depth							
DF		F Ratio		Prob > F	DF		F Ratio		Prob > F			
2		0.05		0.9	2		0.1		0.8			
Effect details					Effect details							
Not Significant					Not Significant							

Table A.T10. Tukey HSD test results from statistical analyses of C, N and C:N ratio of incubated treatment samples, based on analysis of treatments (live, stockpiled, and natural), soil sample depths (0-5, 5-10 cm), and their interactions. Asterisks denote significant differences ($\alpha = 0.05$). Results have been presented only for significant changes. Levels not connected by same letter are significantly different.

C(%)			N (%)		
Effect test: Treatment			Effect test: Treatment		
DF	F Ratio	Prob > F	DF	F Ratio	Prob > F
2	9.8	0.0009*	2	12.8	0.0002*
Effect details			Effect details		
Live		A	Live		A
Stockpiled		B	Stockpiled		B
Natural		A	Natural		A
Effect test: Sample depth			Effect test: Treatment depth		
Not Significant			Not Significant		
Effect test: Treatment * Sample depth			Effect test: Treatment * Treatment depth		
DF	F Ratio	Prob > F	DF	F Ratio	Prob > F
2	0.02	0.9	2	0.02	0.9
Effect details			Effect details		
Not Significant			Not Significant		

C:N Ratio		
Effect test: Treatment		
DF	F Ratio	Prob > F
2	23.4	<0.0001*
Effect details		
Live	A	
Stockpiled	B	
Natural	A	
Effect test: Sample depth		
Not Significant		
Effect test: Treatment * Sample depth		
DF	F Ratio	Prob > F
2	0.08	0.9
Effect details		
Not Significant		

Table A.T11. Spearman's correlation coefficients showing the relationships between C_{min} from laboratory incubated samples and soil and environmental variables within in treatment groups and overall. P-values have been shown in parenthesis and * symbols denote significant relationships ($\alpha = 0.05$).

FACTORS		N (%)	C (%)	C:N Ratio	Bulk Density (g cm ⁻³)	Porosity (V _v V _T ⁻¹)
Live peat	C_{min} ($\frac{mg\ C - CO_2}{g\ dry\ soil\ d_w}$)	0.1579 (0.6241)	-0.5804 (0.0479*)	-0.2448 (0.4433)	0.6314 (0.0277*)	-0.6244 (0.0300*)
	C_{min} ($\frac{mg\ C - CO_2}{g\ C\ d_w}$)	0.1474 (0.6476)	-0.6014 (0.0386*)	-0.2378 (0.4568)	0.5961 (0.0408*)	-0.5891 (0.0439*)
Stockpiled mineral/ peat	C_{min} ($\frac{mg\ C - CO_2}{g\ dry\ soil\ d_w}$)	0.3140 (0.3202)	0.2627 (0.4094)	-0.4098 (0.1858)	0.3915 (0.2082)	-0.3656 (0.2426)
	C_{min} ($\frac{mg\ C - CO_2}{g\ C\ d_w}$)	-0.1891 (0.5560)	-0.2657 (0.4038)	0.0559 (0.8629)	0.5211 (0.0823)	-0.6211 (0.0311*)
Natural Peatland	C_{min} ($\frac{mg\ C - CO_2}{g\ dry\ soil\ d_w}$)	0.6325 (0.3675)	0.6325 (0.3675)	-0.8000 (0.2000)	0.6000 (0.4000)	-0.3162 (0.6838)
	C_{min} ($\frac{mg\ C - CO_2}{g\ C\ d_w}$)	0.6325 (0.3675)	0.6325 (0.3675)	-0.8000 (0.2000)	0.6000 (0.4000)	-0.3162 (0.6838)
All treatment groups	C_{min} ($\frac{mg\ C - CO_2}{g\ dry\ soil\ d_w}$)	-0.1833 (0.3503)	0.0645 (0.7445)	0.2685 (0.1671)	-0.2086 (0.2868)	0.2071 (0.2904)
	C_{min} ($\frac{mg\ C - CO_2}{g\ C\ d_w}$)	-0.2251 (0.2495)	-0.1021 (0.6051)	0.2419 (0.2149)	-0.1159 (0.5569)	0.0995 (0.6143)

Table A.T12. Comparison between C:N ratio, C and N content, soil porosity, bulk density, and carbon mineralization rates in incubated soil treatments.

	0-5 cm								5-10 cm							
	Cell	N (%)	C (%)	C:N	Bulk Density (g cm ⁻³)	Porosity (V _v V _T ⁻¹)	mg C-CO ₂ per g dry soil	mg C-CO ₂ per g C	Cell	N (%)	C (%)	C:N	Bulk Density (g cm ⁻³)	Porosity (V _v V _T ⁻¹)	mg C-CO ₂ per g dry soil	mg C-CO ₂ per g C
live Peat	19	0.93	51.9	55.6	0.1	0.9	112.3	216.4	19	1.01	51.1	50.8	0.1	0.9	51.9	101.7
	12	0.76	52.3	68.8	0.1	1.0	49.4	94.4	12	0.84	54.6	64.6	0.1	1.0	22.7	41.6
	2	0.79	48.8	61.8	0.1	0.9	175.6	359.9	2	0.74	49.5	66.9	0.1	0.9	102.5	207.1
	18	0.87	48.6	56.2	0.1	0.9	218.5	449.6	18	0.84	50.3	59.8	0.1	0.9	176.1	348.1
	6	0.93	48.4	52	0.1	0.9	237.1	489.9	6	0.89	52	58.3	0.1	0.9	255.7	491.7
	11	0.46	48	104.6	0.1	0.9	214.6	447.1	11	0.43	49.7	114.7	0.1	0.9	127.4	256.3
Stockpiled mineral /peat	4	1	26.5	26.5	0.2	0.8	35.8	135.2	4	1.03	31.8	30.8	0.3	0.7	24.0	75.6
	10	1.19	35.3	29.7	0.2	0.8	59.0	167.1	10	1.51	42.1	27.8	0.2	0.8	31.6	75.1
	8	1.73	47.2	27.3	0.2	0.8	28.2	59.7	8	2	49.8	25	0.2	0.8	27.0	54.3
	17	1.7	42.8	25.1	0.3	0.7	68.4	159.8	17	2.07	50.6	24.5	0.2	0.8	43.5	85.9
	7	1.25	35.8	28.6	0.3	0.8	151.0	421.7	7	0.65	23.7	36.7	0.2	0.8	17.5	73.7
	28	2	48	24	0.3	0.8	68.4	142.6	28	1.69	44.2	26.2	0.2	0.8	35.9	70.9
Natural peatland	A	1.06	51.3	48.5	0.1	0.9	47.3	92.3	A	0.77	50.9	65.7	0.1	0.9	17.0	33.5
	B	0.93	53.4	57.1	0.1	0.9	51.0	95.5	B	0.83	53.4	64.3	0.1	0.9	35.3	66.0

Table A.T13. Results from statistical analyses of CO₂ emission measurements by flow-through system and static chamber methods (a) and CH₄ (b) for the whole season based on treatments (live, stockpiled and natural), collar types (CT and RE), soil placement season (winter and summer) and soil depths (15, 50 and 100 cm). Asterisks denote significant differences ($\alpha = 0.05$). Note that the interactions are not presented. Also results have been presented only for significant changes. Levels not connected by same letter are significantly different.

Same letter are significantly different.

(a): CO ₂ measurements													
CO ₂ measured by static chamber method							CO ₂ measured by flow-through method						
Effect test: Treatment							Effect test: Treatment						
DF		F Ratio		Prob > F			DF		F Ratio		Prob > F		
1		388.7		<0.0001*			1		593.6		<0.0001*		
Effect details							Effect details						
Live				A			Live				A		
Stockpiled					B		Stockpiled					B	
Effect test: Treatment depth							Effect test: Treatment depth						
DF		F Ratio		Prob > F			DF		F Ratio		Prob > F		
2		3.5		0.0311*			2		8.5		0.0002*		
Effect details							Effect details						
15 cm				A			15 cm				A		
100 cm					B		50 cm				A	B	
50 cm					B		100 cm					B	
Effect test: Placement season							Effect test: Placement season						
DF		F Ratio		Prob > F			DF		F Ratio		Prob > F		
1		7.7		0.0056*			1		20.7		<0.0001*		
Effect details							Effect details						
Summer				A			Summer				A		
Winter					B		Winter					B	
Effect test: Collar type							Effect test: Collar type						
DF		F Ratio		Prob > F			DF		F Ratio		Prob > F		
3		2.3		0.0690			3		0.46		0.7068		
Effect details							Effect details						
Live, 15 cm				A			Live, 15 cm				A		
Live, 50 cm					B		Live, 50 cm				A		
Live, 100 cm					B		Live, 100 cm					B	

(b): CH ₄ measured by static chamber method						
Effect test: Treatment						
DF		F Ratio		Prob > F		
1		98.9		<0.0001*		
Effect details						
Live				A		
Stockpiled				B		
Effect test: Treatment depth						
DF		F Ratio		Prob > F		
2		13.4		<0.0001*		
Effect details						
15 cm				A		
100 cm				B		
50 cm				B		
Effect test: Placement season						
DF		F Ratio		Prob > F		
1		72.4		<0.0001*		
Effect details						
Summer				A		
Winter				B		
Effect test: Collar type						
DF		F Ratio		Prob > F		
3		37.2		<0.0001*		
Effect details						
Live, 15 cm				A		
Live, 50 cm				B		
Live, 100 cm				B

Figure A.F1. Total average CO₂ emissions (μmol m⁻² s⁻¹) from the collars measured by flow-through system on DOY 151 -182. Error bars represent +1 standard deviation.

Figure A.F2. Total average CO₂ emissions (μmol m⁻² s⁻¹) from the collars measured by flow-through system on DOY 188 - 230. Error bars represent +1 standard deviation.

Figure A.F3. Total average CO₂ emissions (μmol m⁻² s⁻¹) from the collars measured by static system on DOY 151 - 182. Error bars represent +1 standard deviation.

Figure A.F4. Total average CO₂ emissions (μmol m⁻² s⁻¹) from the collars measured by static system on DOY 188 - 230. Error bars represent +1 standard deviation.

Figure A.F5. Total average methane emissions ($\text{nmol m}^{-2} \text{s}^{-1}$) from the collars measured by static system on DOY 151 - 182. Error bars represent +1 standard deviation.

Figure A.F6. Total average methane emissions ($\text{nmol m}^{-2} \text{s}^{-1}$) from the collars measured by static system on DOY 188 - 230. Error bars represent +1 standard deviation.

Figures A.F7 – A.F19: Nutrient capture by PRS™ probes from surface (~5 cm) and deep (~12 cm) of live peat, stockpiled mineral/peat and natural peatland in control, root exclusion and between collars. Probe supply rate for surface probes is in microgram per 10 cm² per 41 ± 1 days (Early: June 28 - July 7, Late: July 7 - August 17, 2010) and for deep probes is in microgram per 10 cm² per 82 ± 1 days (between June 28 and August 17, 2010). Error bars represent +1 standard deviation.

Figure A.F7

* Method detection limit for Ca²⁺ is 2 µg per 10cm² per days of burial.

Figure A.F8.

* Method detection limit for Mg^{2+} is $4 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F9.

* Method detection limit for K^+ is $4 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F10.

Figure A.F11.

Figure A.F12.

* Method detection limit for Mn^{2+} is $0.2 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F13.

* Method detection limit for Cu^{2+} is $0.2 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F14.

* Method detection limit for Zn^{2+} is $0.2 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F15.

* Method detection limit for B ion is $0.2 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F16.

* Method detection limit for $\text{SO}_4^{2-}\text{-S}$ is $2\ \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F17.

* Method detection limit for Pb^{2+} is $0.2\ \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F18.

* Method detection limit for Al^{3+} is $0.4 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F19.

* Method detection limit for Cd^{2+} is $0.2 \mu\text{g}$ per 10cm^2 per days of burial.

Figure A.F20. Microbial respirations ($\mu\text{g C- CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 0-5 cm soil depth of live peat samples incubated for 97 days.

Figure A.F21. Microbial respirations ($\mu\text{g C- CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 0-5 cm soil depth of stockpiled samples incubated for 97 days.

Figure A.F22. Microbial respirations ($\mu\text{g C- CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 0-5 cm soil depth of natural peatland samples incubated for 97 days.

Figure A.F23. Microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 5-10 cm soil depth of live peat samples incubated for 97 days.

Figure A.F24. Microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 5-10 cm soil depth of stockpiled samples incubated for 97 days.

Figure A.F25. Microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 5-10 cm soil depth of natural peatland samples incubated for 97 days.

Figure A.F26. Average microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 0-5 cm soil depth during 97 days of incubation. Blue, red and green lines represent the emissions from live, stockpiled and natural plots respectively. Error bars represent ± 1 standard deviation.

Figure A.F27. Average microbial respiration ($\mu\text{g C-CO}_2 \text{ g}^{-1} \text{ dry soil d}^{-1}$) from 5-10 cm soil depth during 97 days of incubation. Blue, red and green lines represent the emissions from live, stockpiled and natural plots respectively. Error bars represent ± 1 standard deviation.