EFFECT OF PEAT FUEL EXTRACTION AND PEATLAND RECLAMATION ON VEGETATION AND GREENHOUSE GAS EXCHANGE

By

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ABSTRACT

Peatlands are an important component of the global carbon cycle, acting as a significant global net sink of carbon dioxide (CO_2) and are the largest natural source of atmospheric methane (CH_4). Peat is also an important resource and has been used as a fuel grade source of energy and for horticultural purposes in various countries for hundreds of years. The Ontario Government recently developed a target to eliminate coal-fired electricity generation by 2014 which has spurred discussions of replacing coal with alternative biomass fuel sources such as peat in northern Ontario power plants such as the Atikokan Generating Station. The aim of this thesis is to determine the impact of extracting peat from a northern Ontario peatland, in order to assess and quantify any potential ecological impacts in relation to (i) vegetation characteristics and functioning and (ii) greenhouse gas (GHG) emissions.

In April, 2008, a paired-catchment experiment was initiated at a northwestern Ontario peatland near Thunder Bay. Peat was extracted from a 12 m by 12 m segment of the experimental site and subsequently reclaimed using the peat block / acrotelm transplant method. The upper 30 cm thick section of the surface vegetation within the experimental plot was removed and set aside briefly prior to peat fuel extraction. Following the removal of approximately 1.5 m in depth of peat, the surface vegetation peat blocks were transplanted back onto the surface of the remaining catotelm in the plot.

Vegetation and greenhouse gas exchange (GHG) measurements were conducted at a nearby control site, the experimental site and a ditch site located alongside the experimental site outflow extraction ditch during the growing seasons of 2008 and 2009 to assess the impacts of extraction on GHG dynamics.

Biodiversity between the control and experimental site did not differ largely in the sampling years, with the ditch hollows displaying the lowest overall levels of plant biodiversity (0.72 Shannon-Weiner Biodiversity Index) in 2009. Despite bare peat being the greatest proportion of late growing season ground cover at the experimental site in both 2008 and 2009, and despite the lack of trees, keystone *Sphagnum* mosses made up the largest category of living vegetation (30 - 50 %), and had the overall greatest rate of seasonal moss production of all sites. The ditch site had the largest proportion of dead vegetation (77 %), with significantly lower (p < 0.05) production rates of both *S. fuscum* and *S. magellanicum* than *S. fuscum* production rates at the experimental site.

In the first year following extraction GHG fluxes of CO_2 and CH_4 were significantly higher (p < 0.05) at the experimental site than at the control site. By the following year, although CH_4 fluxes increased significantly, fluxes of CO_2 decreased nearly four times from 7.5 to 1.9 g CO_2 m⁻² d⁻¹, and were no longer significantly different from control site hollows.

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Overall the peat block / acrotelm transplant method of peat extraction for the use of peat as a biomass fuel allowed for the rehabilitation of mosses at the experimental site, and prevented any significant overall changes to species cover and biodiversity. As a result of the successful re-establishment of vegetation at the experimental site, overall efflux of CO_2 was significantly reduced only one year after extraction. Despite increased overall fluxes of CH_4 between sampling years it is predicted that GHG fluxes will continue to decrease with time since extraction and rehabilitation.

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CHAPTER 1: INTRODUCTION

1.1 Peatland Structure and Function

Globally, peatlands comprise nearly 400 million ha, or approximately 3% of total land areas (Maltby and Immirzi, 1993). In Canada, peatlands cover approximately 113 million hectares, comprising 76% of the 148 million hectares of wetlands across Canada (Daigle and Gautreau-Daigle, 2001). Peatlands are peat-forming wetlands in which saturated conditions result in slow organic matter decomposition rates and peat accumulation (Clymo, 1984). These peatlands may be further subdivided broadly hydrologically into fens or bogs; bogs are ombrotrophic ecosystems that receive water and nutrients solely from atmospheric sources (*i.e.*, precipitation) and are not influenced by external hydrological sources. Conversely, fens receive groundwater and/or surface water inputs, and are thus influenced by their external hydrological surroundings (Moore, 2002). Northern peatlands are commonly thought to possess a diplotelmic (dual-layered) soil structure (Ingram, 1978). The upper layer is often termed the acrotelm, and consists of poorly decomposed organic matter, and possesses high hydraulic conductivity and porosity (Clymo, 1984; Gleeson et al., 2006). The lower layer, often termed the catotelm, consists of comparatively highly decomposed organic matter, and hence smaller pore sizes, which ultimately leads to high water retention and low hydraulic conductivity. Through their formation under waterlogged and anaerobic conditions, these wetlands begin to accumulate dead organic matter in the form of peat (Clymo, 1984). Peat accumulation is primarily controlled by rates of decay where the biomass formed on the upper portion of the acrotelm decays selectively, with approximately 10 – 20% passing into the catotelm (Makila and Saarnisto, 2008). Furthermore, peatlands generally display a heterogeneous, and sometimes patterned microtopographical structure of pools, hollows, lawns and hummocks. This microtopographic structure is controlled by water table position relative to the peat surface, so that these structures differ in hydrological properties as well as plant species (Malmer, 1986).

Overall, peatlands support many complex ecological functions including climate and catchment hydrology regulation (Gleeson *et al.*, 2006), as well as provision of wildlife habitats and various social and cultural benefits (Daigle and Gautreau-Daigle, 2001). Direct uses of peatlands and wetlands as identified by the Ontario Ministry of Natural Resources (OMNR) include recreational activities such as birding, wildlife viewing and fishing, the commercial harvesting of various resources such as nuts, berries and grains, as well as peat acquisition and logging (Gleeson *et al.*, 2006). Additional indirect benefits of these ecosystems include nutrient retention, water filtration, shoreline protection, and micro-climatic stabilisation. Peatlands are also sources of archaeological information via their preservation of human artefacts for thousands of years, while also providing archives for the reconstruction of long-term plant and animal community changes, with forested peatlands acting as a seed source following fire or clear-cutting (Rochefort, 2000).

1.2 Peatland Vegetation: Ecology and Importance

Peatland vegetation and ecological conditions are greatly determined by the peatland spatial organisation of hummocks and hollows. Varying in elevation by only tens of centimetres, hummocks and hollows exhibit microscale differences in microclimatology, and most importantly, water table levels (Bubier *et al.*, 2006). As a result, very small-scale variation in plant community over the surface of a peatland is common, with different plant communities displaying very different inherent and associated environmental characteristics (Kelly and Schouten, 2002).

In addition, different classifications of peatlands (*i.e.*, fens or bogs) display their own characteristic dominant vegetation, which may be subdivided into ecological types such as open, shrub, treed bogs and fens, and forested swamps. From an ecological perspective, bogs are relatively drier (*i.e.*, they possess a thick acrotelm) with a large percentage of their area dominated by hummocks (Vitt, 2000). Bogs may also be relatively wooded, dominated by shrubs, or entirely treeless. They also often lack a sedge component, but contain abundant ericaceous shrubs, and are dominated by *Sphagnum* moss, or feather moss and lichens. In contrast, fens are generally wetter with a thinner acrotelm and a higher percentage of area covered by lawns and carpets with pools sometimes present (Vitt, 2000). Fens are commonly dominated by sedges, with poor fens dominated by *Sphagnum* mosses and sometimes ericaceous shrubs; and moderately to extremely rich fens dominated by brown mosses, whilst largely lacking ericaceous shrubs (Vitt, 2000).

Peatlands in general often include many rare plant species (Rochefort, 2000), such as many rare species of orchids. In addition, vegetation occurring in peatlands includes many plants that are generally not found in mineral soil ecosystems. Some of these include *Sphagnum* spp. (mosses), *Sarracenia* spp. (pitcher plants), *Ultricularia* spp. (bladderworts), and *Drosera* spp. (sundews), the last three of which are carnivorous species and are able to capture insects as a prime nutrient source. This allows such species to survive in the acidic and generally nutrient poor habitats of ombrotrophic bogs (Daigle and Gautreau-Daigle, 2001). As a result of these conditions, many of the plant species occurring in peatlands have specified ecological niches, and are thus rarely found outside of these ecosystems. Other species such as *Picea mariana* (black spruce) are found in several ecosystems, but may be found in peatlands as they are well adapted to the prominent conditions.

Mosses often dominate 90 – 100% of the ground layer in peatlands, and have been found to influence peatland functioning in four major ways including nutrient sequestration, water-holding abilities, decomposition, and acidification (Vitt, 2000). In relation to nutrient sequestration, nitrogen added to a peatland has been shown to be quickly sequestered by the moss layer, with subsequent movement to vascular plants controlled by its release. Furthermore, mosses are considerably drought-tolerant as a result of their inactivity in moisture-limited conditions, with peatland mosses having developed morphological adaptations to retain moisture and remain active for as long as possible, thus allowing for greater growth (Vitt, 2000). *Sphagnum* mosses play a significant role in

regulating peatland hydrology through their water retention capability (Proctor, 2000; van Breemen, 1995; Waddington *et al.*, 2010). This water retention characteristic may be attributed to their cellular structure and large number of small pores (Gleeson *et al.*, 2006). In addition, bryophytes by definition lack internal vascular structure, so they must retain water from external surfaces, generally in the form of direct precipitation or dew, or by capillary rise, and thus require relatively stable levels of moisture availability (Stern, 2003; Proctor, 2000; McNeil and Waddington, 2003). Bryophytes are thus well adapted to take up water and nutrients over the entirety of their shoots, while also efficiently absorbing and intercepting solutes in rainwater, cloud and mist droplets, as well as airborne dust (Proctor, 2000). *Sphagnum* growth and photosynthetic rates are therefore primarily dependent on water table and available moisture levels, as well as community composition (McNeil and Waddington, 2003; Grosvernier, 1997; Rydin *et al.*, 1985).

Decomposition of moss species in a peatland plays a primary role in the overall carbon accumulation, in that *Sphagnum* species are the principal contributors to peat accumulation (McNeil and Waddington, 2003). In non-peat forming wetlands, rapidly decomposing vascular plants dominate, thus preventing a significant overall accumulation of carbon (Vitt, 2000). Lastly, mosses have been shown to produce the acidic characteristics of peatlands observed. *Sphagnum* exhibits an ability to exchange cations, in that hydrogen ions produced by uronic acid are exchanged for base cations contained in pore waters, thus producing an overall lower pH observed in peatlands

(Clymo, 1963). These geochemical conditions have been identified as important to peat accumulation because they inhibit the growth of bacteria and fungi and, therefore, decomposition (Stern, 2003). As a result, moss species may be considered an integral component to ecosystem functioning, and thus restoration projects often focus on their re-establishment in order to restore functions such as water filtration, carbon storage, acidity, and wildlife habitat (Ferland and Rochefort, 1997).

Thus, due to the unique ecological and vegetation characteristics of peatlands, the OMNR has identified that prior to peat exploitation, thorough species inventories should be undertaken in order to ensure that this habitat for rare species is not lost following extraction (Gleeson *et al.*, 2006).

1.3 Peatland Greenhouse Gas Exchange

During the Holocene, peatland uptake of CO_2 through photosynthesis, and release of CO_2 and methane (CH₄) through respiration, have strongly influenced atmospheric GHG concentration and, through the resultant radiative forcing, global climate (Frolking *et al.*, 2006, Korhola *et al.*, 2010).

1.3.1 Carbon dioxide

As significant terms in the global carbon gas budget, peatlands both absorb and emit the carbon they store (Gleeson *et al.*, 2006, Frolking *et al.*, 2006, Korhola *et al.*, 2010). Most peatlands function as long-term sinks of CO_2 (Roulet, 2000) due to rates of fixation

through photosynthesis exceeding those of aerobic decomposition (Bubier et al., 2003). Since the melting of northern ice sheets since the last glacial maximum, the development of northern peatlands has significantly lowered atmospheric CO₂, as seen from historical atmospheric records (Blunier et al., 1995). As a result, peatlands are an important component of the global carbon cycle, with northern peatlands accounting for fully one third of all global soil carbon, or 455 Pg of carbon (Riutta et al., 2007; Gorham, 1991). Approximately 98.5% of the total carbon in peatlands is stored as peat, with the remaining 1.5% occurring in above ground vegetation (Gorham, 1991). Average peatland carbon accumulation rates range approximately from 10 to 30 g C m^{-2} y^{-1} (Bubier et al., 2003). Net ecosystem exchange (NEE) is driven primarily by photosynthetic uptake of CO₂, and its emission from plant and soil respiration (Bubier et al., 2003). The accumulation of carbon occurs slowly and is reliant upon high water levels, anoxic conditions, and the stability of peat-forming plant litter (Riutta et al., 2007; Blodau, 2002). The primary controls on the carbon cycle generally include soil temperature, plant community structure, position of redox boundaries associated with the water table, ecosystem productivity, and the chemical composition of plant tissues and peat (Blodau, 2002; Strack and Waddington, 2008). As a result of these controls, CO₂ exchange rates in peatlands vary both geographically and seasonally. Considerations of the dynamics of carbon accumulation and peatland-atmosphere CO₂ exchange become increasingly important with the harvesting of peat for horticultural purposes and energy production. With the removal of peat, peatlands are often converted to sources of CO₂ and DOC, ultimately reducing or removing altogether their ability to accumulate carbon (Waddington *et al.*, 2002).

1.3.2 Methane

Despite their ability to sequester large amounts of CO_2 from the atmosphere, peatlands are additionally important to GHG cycling through their emission of methane (Gorham, 1991; Frolking et al., 2002). CH₄ emissions affect the oxidative capacity of the troposphere, and influence the radiative forcing of the Earth's energy balance (IPCC, 2007). Natural CH₄ emissions comprise between 15 - 40% of the total global estimate, with wetlands constituting the majority of these emissions (Christensen et al., 2004; Segers, 1998). CH₄ is more potent as a GHG, because it is approximately 25 times more effective in radiative forcing than CO₂, for a 100-year global warming potential (GWP) (IPCC, 2007). The production of CH₄ comes primarily from microbiological respiration, which is predominantly controlled by the prevalence of anoxic conditions, the availability of labile organic matter, and the absence of alternative electron acceptors such as NO₃, Fe³⁺, and SO₄²⁻ (Segers, 1998). Thus, CH₄ production occurs when organic matter is degraded anaerobically by bacteria, with the final stage of degradation involving methane-producing bacteria ("methanogens") (Segers, 1998). Methane is produced at the interface between the acrotelm and catotelm. Dissolved methane may be transported through the soil matrix and vegetation by diffusive and advective processes, which ultimately determines the rate of methane release from the soil. Emissions of CH₄ are thus primarily controlled by water-table position (which determines the degree of anaerobicity in the peat profile), and peat temperature (which regulates the rate of microbial activity) (Bubier *et al.*, 1995; Segers, 1998). Average CH₄ emissions in northern peatlands most commonly vary between 5 and 80 mg m⁻² d⁻¹ (Blodau, 2002).

1.4 Peat as an Energy Source

Along with the increase in price for fuels such as oil and natural gas, the Ontario Government has developed a target to eliminate coal-fired electricity generation by 2014 (Tang, 2009). This has spurred interest in replacing coal with alternative fuel sources in OPG power plants such as the Atikokan Generating Station near Thunder Bay, Ontario, which has been provisionally scheduled for closure (Gleeson et al., 2006). As a result, peat has been suggested as a potential alternative fuel source. The Environmental Commissioner of Ontario's 2004-05 Annual Report concluded that regulation of peat resources and harvesting activities is lacking, and further recommended that OMNR and other provincial ministries "develop appropriate regulations to ensure that peat harvesting is carried out with minimal ecosystem disturbance, and that appropriate rehabilitation is undertaken" (as quoted by Gleeson et al., 2006). Peat is currently not being used as a fuel source in Ontario and Canada (Daigle and Gautreau-Daigle, 2001); thus, the importance of the ecosystem impact assessment of peat extraction and site rehabilitation has increased greatly in recent years with new energy budgets and provincial policies and their considerations of the use of peat for energy. In determining and assessing the qualities of peat as a potential alternate fuel source, various considerations need to be made including the amount of peat available, the quality of peat as an energy source (i.e.,

net calorific value), the sustainability of extraction in relation to peat accumulation rates, additional considerations of burn emissions, and the costs and considerations for converting existing generating stations into peat-burning facilities.

Representing an early stage in the formation of coal, peat may eventually develop successively into lignite, charcoal, anthracite, and graphite, eventually becoming coal through exposure to sufficient heat and pressure with burial under other sediments (Gleeson *et al.*, 2006). As a result, peat has a relatively low net calorific energy value in comparison to other fuel sources. At its highest net calorific value, peat in the form of pellets can provide 16.2 MJ/kg of energy, with coal in its various forms (lignite to anthracite) providing between 21 and 32 MJ/kg, and fuel oil and natural gas providing 43 and 49 MJ/kg of energy respectively (Gleeson *et al.*, 2006).

It has been estimated that over 70 million tonnes of peat accumulate in Canada each year, though current applications for horticultural use utilise over one million tonnes annually (Daigle and Gautreau-Daigle, 2001). As Canada has limited experience with the use of peat as an energy source, researchers often consider experiences from other regions such as Finland and Ireland, where peat is an important local energy source, with 7.5 and 4.7 million tonnes of peat respectively produced for energy in 1999 (Gleeson *et al.*, 2006).

Canada opened its first (and hitherto only) peat-fired power station, Dapp GS in Alberta in 1998. This station, which closed in 2000, was a 17 MW facility and required 25 tonnes

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of peat per hour as fuel (Gleeson *et al.*, 2006). Furthermore, in Ireland, a country with a long history of peat use for fuel, milled peat accounts for 13% of the country's power needs, with approximately 3 million tonnes of milled fuel peat per annum provided to power stations for electricity generation by the Irish company Bord na Mona (Gleeson *et al.*, 2006).

Furthermore, common by-products associated with the burning of any biofuel, such as peat, include fly ash and dust, with a 100 MW generation station producing between 11,000 and 100,000 tonnes of ash per year (Gleeson *et al.*, 2006). Similarly to coal, peat can contain many trace elements, most of which are retained in the ash and emitted to the atmosphere. Few studies have provided specific estimates of these trace emissions for peat combustion, and further studies are presently still underway. Lastly, since peat produces emissions of varying components to coal, modifications to fuel handling systems and generating stations in general are required for burning peat. According to a 2006 peat fuel report produced by the OMNR, these estimates had not yet been made (Gleeson *et al.*, 2006).

1.5 Peat Extraction Techniques and Rehabilitation Methods

Various extraction techniques exist for the removal of peat from a peatland. Traditional peat extraction techniques include block cutting and vacuum harvesting, both of which have been shown to damage peatland ecohydrology and convert them into persistent sources of atmospheric CO_2 (Waddington *et al.*, 2009; Waddington *et al.*, 2002). The

block cutting method involves the removal of blocks of peat from large trenches approximately 200 m long and 10 m wide, ultimately leaving a landscape of alternating baulks and trenches (Waddington *et al.*, 2009; Price *et al.*, 2003). These peat blocks_are then left in piles to dry before being removed from the peatland. Although this technique is generally no longer commercially employed, many abandoned block-cut peatlands still remain in eastern North America, the UK, the Republic of Ireland, and continental Europe (Price *et al.*, 2003). Similarly, the vacuum harvesting technique, also known as milled peat harvesting, is the most common method of peat extraction employed in Canada (Price *et al.*, 2003), and involves an initial drainage of the peatland, followed by the removal of surface vegetation and milling of upper layers of peat. The peat fragments are then collected by large vacuum extraction vehicles. During drying, the peat decomposes, releasing significant amounts of CO_2 (Waddington *et al.*, 2009).

Recently, a Canadian peat horticultural company developed a new extraction technique known as the acrotelm-transplant approach, also referred to as the peat block reclamation method (Waddington *et al.*, 2009). Prior to the typical mechanical extraction of peat, 30 cm-thick peat blocks consisting of the surface vegetation are removed from the extraction plot and set aside. Following the peat removal of approximately 1-3 m in depth, the conserved acrotelm blocks are transplanted back onto the surface of the remaining catotelm in the plot. The result is a trench topography in which the surrounding natural peatland is higher than the plot of extraction (Waddington *et al.*, 2009). As the process is repeated over time, the overall elevation of the peatland decreases. By transplanting the

acrotelm back onto the cutover surface, the peatland is thus 'rehabilitated' into a vegetation supporting ecosystem.

Following peat extraction, CO₂ release from peat can increase by 400 % (Waddington et al., 2002). Rehabilitation techniques should therefore be employed in order to allow peatland characteristics to return to "near natural" conditions, to restore their ability to accumulate carbon. Examinations of mined peatlands indicate that these regions do not tend to recover their hydrological function, because the dominant peat-forming genus, Sphagnum, does not readily re-establish itself on degraded peat surfaces (Price et al, 2003; Cagampan and Waddington, 2008). This ultimately alters carbon accumulation and CO₂ flux patterns from the peatland (Waddington et al, 2002). In addition, peat extraction absent of site rehabilitation often results in a lowering of the water table due to: drainage; increases in runoff; increases in peak flows; and increases in baseflow relative to natural conditions (Price et al., 2003). If peatlands harvested using the block cut or vacuum harvest methods are restored, CO₂ emissions may be significantly reduced (Ferland and Rochefort, 1997; Tuittila et al., 2004). Such restoration techniques may include the rewetting of these peatlands to restore the water table, damming of ditches and other drainage trenches, maintaining humid conditions for Sphagnum growth, as well as various techniques for re-establishing Sphagnum fragments on bare peat surfaces (Price et al., 1998; Ferland and Rochefort, 1997; Tomassen et al., 2003).

Thus, the peatland restoration component of the acrotelm-transplant method leads to far lower carbon emissions, and has an overall lower GWP, than block cutting and vacuum harvesting (Waddington *et al.*, 2009). Even with post-harvest restoration, peatlands subject to block cutting and vacuum harvesting emit much greater amounts of GHGs than those where the acrotelm-transplant method is employed, because the duration of peatland abandonment prior to restoration is eliminated (Waddington *et al.*, 2009).

1.6 Objectives

As a result of the Ontario Government's target to eliminate coal-fired electricity generation by 2014 (Oliveira, 2009), there is present interest in replacing coal with alternative fuel sources, such as peat, in OPG power plants such as the Atikokan Generating Station near Thunder Bay, Ontario (Gleeson *et al.*, 2006).

As a result of a peatland's ability to sequester and accumulate carbon, as well as the unique ecological compositions and characteristics of these sites, determining the potential impacts of large-scale extraction of peat is essential for the development of remediation techniques and overall preservation of these ecosystems. In the past, studies have shown that without appropriate and immediate restoration measures, these peatlands lose a large part of their ecological characteristics, lose their overall ability to accumulate carbon, and often become persistent sources of carbon to the atmosphere, thus contributing to climate forcing and global warming (Ferland and Rochefort, 1997; Waddington *et al.*, 2002).

This thesis aims to examine the impacts on vegetation characteristics and GHG exchange, following peat extraction for the purpose of utilization as an alternative energy source, with application of the acrotelm transplant method. While other peat extraction methods often remove and discard the vegetated portion of the peatland (the acrotelm), this method retains the acrotelm and transplants it back onto the surface following extraction (Cagampan and Waddington, 2008). It is thus hypothesized that following extraction, with use of this method, vegetation functioning and characteristics will be restored so that species coverage, growth and biodiversity will not differ from a natural site, while water table levels at the acrotelm surface will be maintained. In addition, with maintenance of a near-natural water table level and re-establishment of the acrotelm, it is hypothesized that CO_2 fluxes will increase following extraction with the disturbance of the peat structure, while fluxes of CH₄ will increase with a potential increase in water table level.

More specifically, the objectives of this thesis are to:

- 1. Determine the likely changes to plant-community composition and structure following extraction and acrotelm transplant. Addressing this objective will involve:
 - a. Assessing vegetation coverage and species composition at a natural (unharvested) site;
 - b. Assessing vegetation coverage and species composition at an experimental peat extraction site, as well as all aspects of the extraction plot including the plot itself and the nearby drainage ditch;

- c. Determining the dominant controls of any differences observed.
- 2. Determine the extent of the changes to GHG cycling attributable to peat extraction and acrotelm transplant at a harvested peatland. Addressing this aim will involve:
 - a. Measurements of spatial and temporal CO₂ and CH₄ fluxes from a natural site and an experimental peat extraction plot;
 - b. Determination of the factors controlling the observed fluxes and any differences observed.

CHAPTER 2: METHODOLOGY

2.1 Site Description and Experimental Design

The study was conducted in a poor fen in northwestern Ontario, 40 km east of the town of Upsala (48° 53' N, 89° 59' W) which is representative of peatlands in the region close to the Atikokan Generating Station (AGS). The peatland is a low-boreal wetland, located at the transition between the continental and humid boreal forest. Long-term average annual precipitation is ~ 679 mm, with a long-term average annual temperature of 0.7° C (Environment Canada, 2010).

Vegetation at the peatland is generally characteristic of an open poor fen ecosystem. Ground species consist of mosses, shrubs and sedges, with mosses consisting primarily of *Sphagnum fuscum* on hummocks and *Sphagnum magellanicum* in both hummocks and hollows. Shrubs at the site are dominated by *Chamaedaphne calyculata*, and low ericaceous shrubs including *Andromeda glaucophylla* and *Vaccinium oxycoccus*. Sedges consist primarily of *Carex limosa*, while tree species are dominated by *Picea mariana* and *Larix laricina* with a density of ~ 500 stems ha⁻¹.

Three sites were designated within the peatland: i) a natural (control) site; ii) an extraction (experimental) site; and iii) a near-ditch site. No peat extraction occurred at the control site (i), thereby allowing examination of the peatland in its natural state. In April 2008, peat was extracted from a 12 m x 12 m plot (experimental site (ii)) at a depth of 1.5

m, according to the acrotelm transplant technique (see Cagampan and Waddington, 2008 for details). Slow drainage maintained higher water-table levels at the experimental site than would normally be expected for this technique, thus creating a floating peat-mat structure (Figure 2.1). The experimental site is located ~ 70 m from the site access logging road and ~ 80 m east of the control site (Figure 2.2). The site outflow ditch, installed with a V-notch weir to measure stage and discharge, leads from the experimental plot, and drains water in a southerly direction away from the experimental site, in order to avoid excessive water-logging. The ditch site (iii) is located adjacent to this outflow ditch. Measurements of GHG fluxes, hydrology, and vegetation characteristics took place over two growing seasons, from May to August in 2008 and 2009.

2.2 Meteorological Variables

We set up a meteorological station at the control site in 2008 in order to measure air temperature, photosynthetic active radiation (PAR), hummock and hollow volumetric moisture content (VMC), and hummock and hollow soil temperature. Soil temperatures were measured using Type T (copper-constantan) thermocouples (at depths of 2, 5, 10, 20, and 30 cm for the hummock and 2, 5, 10, and 20 cm for the hollow). VMC was measured using 10 cm Campbell Scientific CS616 water content reflectrometer moisture probes inserted into a hummock and hollow at a 45° angle from the horizontal. PAR was measured at 1 m above the ground surface using a LI-COR LI190SB Quantum Sensor.

All instruments were connected to a Campbell Scientific CR10X data logger, and logged once every 20 min during the 2008 sampling season, and every hour during the 2009 sampling season. An additional meteorological station was set up at the experimental site in May of 2009 to measure water level at the outflow weir (mm); precipitation (mm); hummock and hollow VMC (%); hummock and hollow soil/surface temperature (°C); and shaded air temperature (°C). Soil temperatures at the experimental site were also measured using Type T (copper-constantan) thermocouples, which were inserted at depths of 10, 20, and 50 cm for the hummock and 10, 20, and 30 cm for the hollow. Moisture levels were measured using Campbell Scientific CS616 water content reflectrometers, installed in the same way as those at the control site. Air temperature was measured using a Campbell Scientific CS107 temperature probe. A Campbell Scientific RS-232 pressure transducer was submerged in the ditch and installed on the outflow weir in order to record changes in water level. A Texas Instruments rain gauge (TE525MM) was fixed to a stake at a height of ~ 0.5 m above the ground to record precipitation. All automated measurements at the experimental site were recorded to a Campbell Scientific CR10X data logger at a logging interval of 20 min, from May to August 2009.

Long-term average annual precipitation and air temperature data were obtained from Environment Canada records from the Upsala, Ontario Meteorological Station for 1948 to 2009, located approximately 30 km west of the study site. Data were compiled and ranked using Microsoft Excel's percent rank function, in order to determine where the 2008 and 2009 precipitation and temperature sampling months (May to August) ranked in the long term averages for this region.

2.3 Hydrological Measurements

2.3.1 Water Table and Discharge

Water-table levels within the three sites were determined from manual measurements conducted at a series of stakes and PVC wells (inner diameter = 5.1 cm) (Figure 2.2). At the control site, in addition to the three hummock and three hollow wells installed next to the gas flux collars, five wells were installed surrounding the site. At the experimental site, two stakes were installed at the location of outflow from the plot, in addition to the six collar stakes, for above-surface water level measurements. Thirteen additional groundwater wells were installed on the north, east, and west sides of the experimental extraction pit, perpendicular from each edge of the plot, at a ~ 2 m spacing interval. Three stakes were installed inside the outflow ditch, with three additional wells located just alongside the ditch, along with the two hummock and two hollow wells located adjacent to the gas flux collars. In 2008, water-table elevations were measured in all wells a minimum of three times per week. In 2009, water tables were measured three times per week in the wells located at the gas collars (see section on greenhouse gas methodology for details), during each GHG sampling event, while the remaining wells were measured once per week.

Discharge (dimensions of $L^3 T^{-1}$) from the experimental site was measured in the drainage ditch three times per week during the sampling periods, based on the length of time required to fill a bucket of known volume at the V-notch weir.

2.4 Vegetation and Ecology

Vegetation sampling was conducted at several plots spanning approximately 150 m^2 at each of the three sites including determination of biodiversity, proportional functional species cover, leaf area index, moss growth, and aboveground biomass. Each of these metrics is defined and described below.

2.4.1 Plant Biodiversity

In 2008, plant-species percent-cover was estimated within the CO_2 and CH_4 sampling collars (see section on greenhouse gas methodology for details) (60 cm x 60 cm) at the control and experimental sites. In 2009, percent cover at twelve 30 cm x 30 cm vegetation quadrats, situated every meter along 12 m transects with one transect per site, were sampled in addition to the GHG collars. Control and ditch site transects were each located in a representative location at the sites, with quadrats selected in a subjective sampling strategy, so that approximately equal numbers of hummocks and hollows were selected. Due to the lack of hollows located in close proximity to the ditch site transect, all quadrats were located on hummocks. At the experimental site, the quadrat transect was situated outside the experimental extraction plot, running in an approximately NW direction from the NE end of the plot. Measurements were conducted bi-weekly for the

control and experimental sites from May to August in 2008, and monthly at all three sites from May to August in 2009. Percent cover for each plant species was estimated at each collar and quadrat, with estimates of bare ground cover included in 2009.

Biodiversity levels for 2009 were calculated for each collar and quadrat using the Shannon-Weiner Index of biodiversity (dimensionless):

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$
[1]

Where H' is the index of biodiversity (with higher values indicating greater levels of biodiversity), p_i is the relative distribution of abundances amongst species (evenness of the species in a given area) (Rousseau and Van Hecke, 1999), calculated as the proportion of individuals of a given species to the total number of individuals in the community, (n_i / N) (dimensionless). Furthermore, species richness, S, (dimensionless) was determined by counting the total number of species present at each sampling location.

2.4.2 Proportional Species Cover

Percent cover data from the GHG collars and vegetation quadrats were also used to determine species proportional covers at each of the sites. Proportional covers of individual species were grouped in species groups defined as: mosses, graminoids, shrubs, carnivorous plants, wild flowers, horsetails, and trees, as found in Johnson *et al.*

1995, as well as by site. Proportions were calculated for all species groupings at all sites. Proportional covers of individual species at each site were calculated as follows:

$$P_C = \frac{\sum A_s}{T_s}$$
[2]

Where P_c is the proportional species cover for a given species grouping at a given site; A_s is the percent cover of a given species at a given site; and T_s is the sum total of all species percent covers at the given site.

2.4.3 Leaf Area Index

In order to determine leaf area index of the control, experimental, and ditch sites, five 7.5 \times 7.5 cm subplots were set up inside each collar using wooden dowels (Strack *et al.*, 2006). The species present in each quadrat were noted, and the number of green leaves for each of the species present in the quadrats were counted. A set of three random plants for each species was selected for leaf-size measurements, in accordance with the survey of species present in the quadrats. The individuals selected were located outside of, but within 5 m of the collar surveyed. The length and width of one side of the leaves of these plants were measured at each sampling period, in order to determine leaf sizes.

Measurements of leaf area index were taken once per month from May to August in 2008 and 2009. Species leaf area was then calculated by summing the total number of leaves of each species in a collar, and this was multiplied by the average leaf area of the species sampled. The overall leaf area index was then determined using the following equation:

$$LAI_{C} = \frac{\overline{x}_{i}}{A}$$
[3]

Where LAI_C is the leaf area index of a given species in a collar, \bar{x}_i is the average of each species leaf area in the collar, and A is the sample area, which in this case was equal to 281.25 cm². In order to obtain continuous seasonal values of leaf area, values were extrapolated between measured values at the individual collars. As such, a linear change in leaf area between dates was assumed, and values were calculated accordingly.

2.4.4 Functional Group Leaf Area

Functional group leaf area measurements were conducted by selecting species of conifer trees, sedges, and ericaceous shrubs at the control, experimental, and ditch sites. Taxonomic species for the functional groups selected included the conifer *Larix laricina* (Tamarack), the sedges *Carex limosa* (Mud Sedge) and *Scirpus hudsonianus* (Hudson Bay Bulrush), and the shrub *Chamaedaphne calyculata* (Leatherleaf). Individuals of species were selected and tagged randomly along all site transects and collars, with two conifers, two sedges, and three shrub individuals randomly selected and tagged in 2009. Length and width of a random sample of leaves and needles were measured in order to determine average leaf area for each sample individual. The approximate total number of leaves/needle bunches were then counted in order to calculate the overall leaf area of each sample. In addition, for the Tamaracks, the average number of needles per needle grouping was counted for a random selection of needle groupings. In mid August, total counts and leaf areas of an average selection of new growth needles were also measured

and recorded. VMC was also measured in mid-August in order to obtain an average moisture measurement for each individual. Measurements of functional group leaf areas were conducted once per month from May to August in 2009.

2.4.5 Moss Growth

The cranked-wire method of Clymo (1970) was used in order to assess Sphagnum growth rates. Cranked wires were constructed of 30 cm long fine, (2 mm; 12 gauge), stainless steel rods and inserted into the moss surface so that the protruding end may be measured. with the crank in the wire preventing general movement of the wire with moss growth. Throughout the growing season, the decreasing distance from the Sphagnum surface to the top of the cranked wire is thus used to determine overall length of growth (McNeil and Waddington, 2003). Ten continuous Sphagnum colonies with 85% or greater coverage of Sphagnum species (five for S. fuscum and five for S. magellanicum, the two dominant Sphagnum species at the site) were randomly selected at the control, experimental, and ditch sites, and were used for growth measurements using cranked wires during the 2009 sampling period. Five cranked wires were placed in each species colony (*i.e.*, n = 25 cranked wires per species, per site), and their initial heights were recorded. The wires were then measured monthly from June to August in 2009. Three small destructive samples each of S. fuscum and S. magellanicum were extracted from each of the sites at the end of the season, in order to determine average bulk density values of the mosses at each of the sites. Samples were frozen to avoid compression when cutting, cut into known volumes of small peat blocks, weighed, and placed in an oven at
$^{\circ}$ C for 24 hour increments, until the weighed masses were no longer changing between 24 hr increments. Dry bulk density was then calculated by dividing the dry moss weight by the sample's original volume to obtain a density in g cm⁻³. The overall annual production of the mosses was then calculated by multiplying the total growth (initial cranked wire length minus final) by the dry bulk density of the appropriate species at the site in question.

2.4.6 Aboveground Biomass

In order to determine the aboveground biomass of individual species at each of the three sites, destructive samples were extracted from GHG collars, in that half of each collar was cut to a depth of 5 cm, bagged, and labelled. Destructive samples were extracted prior to plant senescence upon completion of the 2009 sampling season, once all other sampling had been completed so as not to disturb the site. Samples were then processed in the lab to separate each collar sample into species groupings, as defined earlier. All vegetation was clipped for aboveground biomass, while mosses were cut to depths of 5 cm for consistency between collars. Samples were dried at a temperature of 70°C for 24-hour periods, before being weighed in order to obtain the final dry weight of samples.

Dried weights were grouped and summed by species groupings using Johnson *et al.*, 1995. All summed biomass weights were then divided by the total extracted sample area of 30 x 15 cm, in order to determine a mass per unit area of total biomass for each species grouping.

2.5 Greenhouse Gas Exchange

2.5.1 Carbon Dioxide Exchange

The closed-chamber method (Carroll and Crill, 1997) was used to estimate CO₂ fluxes between the peatland and the atmosphere at the control, experimental, and ditch sites. 60 cm x 60 cm steel chamber platforms ("collars") were inserted approximately 10-15 cm into the ground at permanent sampling locations at each of the sites (six at the control site, six at the experimental site, and four at the ditch site). Collars were located randomly for the control and ditch sites with equal numbers of hummocks and hollows selected. Placement of collars in the experimental site was intended to capture the hummockhollow microtopography which existed prior to extraction, informed by plant species assemblages present in transplanted segments. This selection may nonetheless be considered to be essentially random, since the exact microtopographical structure and botanical composition of transplanted vegetation were impossible to estimate accurately.

Net ecosystem CO_2 exchange (NEE) was measured using a clear, square acrylic chamber (area = 1,200 cm², volume = 120 L) placed and sealed over the collars at each of the sampling locations. Concentrations of CO_2 were determined using PP-Systems EGM-1 and EGM-2 portable infrared gas analysers (IRGAs) over a period of three minutes, with values recorded at 30 s intervals. Measurements were taken for treatments including the clear chamber, and progressive shading covers to mimic various light conditions (1shroud cover, 2-shroud covers), and subsequently determine relationships between NEE and levels of PAR. A mid-season calibration was conducted in order to calculate shroud percentage transmittance as compared to full light conditions. An opaque shroud treatment was applied to determine net ecosystem respiration (ER). Gross ecosystem production (GEP) was determined as the difference between measured NEE and ER... values from the plot over the same sampling period. Immediately following CO_2 flux measurements, the area above the gas flux collar was aired out by opening the chamber for a minimum of 30 s prior to the next measurement, allowing the measured gases to dissipate and disperse. Measurements of PAR (PAR-1, PP-Systems, Amesbury, MA) and air temperature within the chamber were taken coincidentally with the CO_2 flux measurements, using the same 30 s time interval, while an internal chamber fan was used in order to achieve homogeneous air humidity and temperatures within the chamber.

Additional measurements of water table level, volumetric moisture content (VMC), and peat temperature were recorded in order to establish empirical relationships with measured CO₂ fluxes. Water table levels were determined using manual point measurements from 150 cm-long PVC wells located adjacent to each collar. VMC of underlying peat at each collar was measured using a Campbell Scientific Hydrosense TDR, inserted at an angle of 45° to the horizontal to a depth of 10 cm. Peat temperature profiles (°C) were obtained using a K-type handheld temperature probe for depths at 2, 5, 10 and 20 cm. CO_2 flux measurements were taken twice a week from May to August at the Control and Experimental sites in 2008 and 2009, and bi-weekly at the ditch site in 2009. The CO_2 flux rate (g CO_2 m⁻² d⁻¹) was calculated as the slope of the linear relationship between headspace CO_2 concentration and time. Slopes with r² values of less than 0.70 were excluded from calculations except for the case of zero flux.

Overall CO₂ fluxes were analysed using a repeated measures ANOVA and Tukey's *post hoc* comparison of means (SPSS Statistics v17.0) for NEE, ER, and GEP. In order to allow for an appropriate number of degrees of freedom, data sampling days in 2008 and 2009 were split into beginning (27 May – 03 Jul) and end of season (07 Jul – 13 Aug) sections. Due to the fewer overall sampling terms at the ditch site, analysis including this site was conducted over the entire season and included five sampling days, ranging from 01 Jun to 29 Jul 2009.

Analyses of inter-annual data were conducted for individual sites using a paired t-test. Sites were not cross analysed, such that individual sites were only compared with each other between sampling years.

Ecophysiological parameters of CO_2 fluxes were characterized at each sampling collar at the control and experimental site, and by microtopographical feature (*i.e.* hummock and hollow) at the ditch site. Parameters were characterized by creating light-response curves, which model the relationship between GEP, PAR, and environmental variables that have

been shown to control variations in GEP fluxes, including leaf area, water table, and soil moisture.

Data were split into two main sections for control and experimental sites, beginning of sampling season (late May to early July; "spring") and end of season (early July to mid August; "summer"), in order to improve curve prediction for GEP and environmental variables. The ditch site was examined over the entire sampling season, due to the fewer sampling dates and thus available data for curve prediction.

Firstly, measured GEP fluxes were modelled at sample collars for light dependence, where GEP and PAR are fitted using a least-squares regression to a rectangular hyperbola (Cagampan, 2006).

$$GEP = \frac{(\alpha * PAR * GP_{MAX})}{(\alpha * PAR) + GP_{MAX}}$$
[4]

Where *PAR* is measured PAR (μ mol m⁻² s⁻¹), *GP_{MAX}* is the empirically derived gross photosynthetic exchange of CO₂ and α is the initial slope of *GEP* versus *PAR*.

Additional curves were modelled to incorporate the environmental variables of leaf area, water table, and soil moisture. The equation model incorporating leaf area is that of Wilson *et al.* (2006) and is defined as:

$$GEP = \frac{Q * PAR * GP_{max}}{Q * PAR + GP_{max}} * (LAI/(LAI + kLAI))$$
^[5]

Where *PAR* is the measured PAR, GP_{max} is the empirically derived gross photosynthetic exchange of CO₂, *LAI* is the measured leaf area, *kLAI* is the empirically derived leaf area, and *Q* is an unknown co-efficient.

Variations in PAR and water-table level were incorporated in an additional model from Chivers *et al.* (2009), defined as:

$$GEP = GP_{max} * \frac{PAR}{k + PAR} * exp \left[-0.5 * \frac{(WT - uP)^2}{tP^2} \right]$$
[6]

Where *PAR* is the measured PAR, GP_{max} is the empirically derived gross photosynthetic exchange of CO₂, Q is an unknown co-efficient, WT is the measured water table level, uP is the optimal water table for photosynthesis, and tP is a measure of the amplitude of the water table.

Modelled VMC curves were derived using the equation used for LAI, with LAI replaced with *VMC* and *kVMC* values:

$$GEP = \frac{Q * PAR * GP_{max}}{Q * PAR + GP_{max}} * (VMC/(VMC + kVMC))$$
^[7]

Where *VMC* is the measured volumetric moisture content of the soil, and *kVMC* is the empirically-derived volumetric soil-moisture content.

Lastly, a combination of LAI and WT variables were incorporated into one model in order to maximize the GEP curve estimation power using the following equation, as derived using the Wilson *et al.*, 2006 model for LAI.

$$GEP = \frac{Q * PAR * GP_{max}}{Q * PAR + GP_{max}} * \left(\frac{LAI}{LAI + kLAI}\right) * \left(\frac{WT}{WT + kWT}\right)$$
[8]

--.

Where *kWT* is the empirically derived WT level.

Environmental variable relations with ER were also modelled, although data were not split into seasonal sections, as fewer data points existed for ER data (dark shroud only). Environmental variables modelled include water table, soil temperature, and soil moisture content.

The dependence of ER on water table level was modelled using the equation from Chivers *et al.* (2009):

$$ER = ER_{max} * exp\left[-0.5 * \frac{(WT - uR)^2}{tR^2}\right]$$
[9]

Where ER_{max} is the maximum rate of ER when water table position is optimal for plant respiration, uR is the optimal water table for respiration, and tR is a measure of the width of water table amplitude.

ER and soil temperature dependencies were modelled using both a temperature model from Chivers *et al.* (2009) and the GAMMA model from Khomik *et al.* (2009).

The Chivers et al. (2009) model is defined as:

$$ER = A x Q_{10}^{\frac{T}{10}}$$
 [10]

Where A is the rate of ecosystem respiration at 0°C, Q_{10} is the temperature dependence of ecosystem respiration, and T is soil temperature beneath the soil surface, for temperature depths of 2, 5, and 10 cm.

Furthermore, the alternate GAMMA model for ER and temperature (Khomik *et al.*, 2009) is defined as:

$$ER = T^{\alpha} e^{\beta_0 + \beta_1 (T)}$$
^[11]

Where T is the soil temperature + 40 °C, and α , β_0 and β_1 are unknown co-efficients to be estimated.

Model curve r^2 , corresponding p-value, and all estimated values for GEP and ER models were then summarised into tables, after which the model with the best average fit for a given site (i.e. control hummock, control hollow, and experimental site) was selected. As such, the GEP model derived from Wilson *et al.* (2006) defined in equation number eight was selected for the control site hummocks and hollows, while the Chivers *et al.* (2009) model defined in equation number six was selected for the experimental site. For ER, the Chivers *et al.* (2009) model for temperature in equation ten was selected for the control site hummocks and hollows, while the Chivers *et al.* (2009) model for water table in equation nine was selected for the experimental site. Derived model parameters were then averaged from site collars accordingly, and these average parameters were then used to model total seasonal NEE fluxes. Environmental variables (LAI; PAR; soil temperature and water-table level), corresponding to the selected model, were then extrapolated at the respective sites for the season, in order to determine continuous measurements of the variable in question. Values were extrapolated with the assumption of a linear change between actual measured values. Using the average model results and extrapolated values, GEP and ER values were then calculated for the entire season using continuous PAR field measurements from the control site data logger, in order to determine the netsum CO_2 flux over the sampling seasons.

For both GEP and modelled ER relations, model parameters were excluded from averages if model fits with the environmental variables examined were insignificant (i.e. p > 0.05), or if predicted values exceeded reasonable bounds for the environmental variable in question (i.e. if predicted *GEP_{max}* was greater than 35 g CO₂ m⁻² d⁻¹). The ditch site was not modelled for 2009 data due to a lack of significant model data results.

2.5.2 Methane Fluxes

Fluxes of CH₄ from the three sites were measured using closed-chamber sampling over the two sampling seasons. Opaque, square-section, stainless steel chambers with attached plastic tubing were placed and sealed at each of the collar sample locations. After being pumped and cleared of any residual gas content, 10 mL syringes were connected to the tubing to extract initial air samples (representing ambient air), and after 10, 20, and 30 min, at each of the sample locations. Air temperatures inside the chambers were regulated and recorded at 5, 10, 15, and 30 minutes. Small fans in the chambers ensured wellmixed air, and so homogeneous internal atmospheric humidity and temperature. Watertable level, VMC, and soil temperatures at 2, 5, 10, 15, 20, 25, and 30 cm were also recorded at each collar during each sampling period. Air samples were stored in a cooler and shipped to McMaster University's Ecohydrology Lab within several hours, before analysis with a Varian gas chromatograph (GC). CH₄ sampling was conducted once a week for the experimental and control sites from May to August in 2008 and 2009, and bi-weekly for the ditch site from May to August in 2009 only.

Surface inverted funnel bubble traps (see Strack *et al.*, 2005) were installed at the Control and Experimental sites to measure methane ebullition. Traps consisted of 20 cm diameter funnels connected to \sim 30 cm lengths of Tygon tubing attached to a syringe with a stopcock. Funnels were inserted just below the peat surface to the water table level. An extra syringe was used to draw water into the funnel until the syringe on the funnel was full of water and did not contain any air bubbles. Five funnel traps were installed at both the control and experimental site, although by the end of the summer, four traps remained in working order at the control site, with three at the experimental site. Gas traps were measured by recording the volume of gas in the syringe three times per week. When the gas volume reached a minimum of 5 mL, samples were extracted in an additional syringe and sent to the lab for concentration analysis on the GC.

Net CH₄ fluxes were analysed using a one-way ANOVA and Tukey *post hoc* comparison of means (SPSS Statistics v 17.0) in order to determine the effect of site on measured CH₄ fluxes.

Inter-annual comparisons were conducted using a paired t-test. Sites were not crossanalysed, such that individual sites were only compared with each other between sampling years.

Additional relations of CH₄ and environmental variables including soil temperature, soil moisture, and water-table level were determined using a linear regression. Variables were grouped according to site, and binned. Bin sizes were either 1° or 2°C bin ranges for soil temperatures, with a maximum allowance of two bin averages having two or fewer averaged variables, in which case the bin ranges were adjusted. Water table and VMC data were not binned due to the large variation in sample values.

2.5.3 Global Warming Potential Estimates

Estimates of global warming potential (GWP) were determined by calculating average carbon equivalences from field measurements, in that CH_4 fluxes are equivalent to 26 times the measured CO_2 fluxes, such that

$$GWP_s = \overline{NEE} + 26 * (\overline{\uparrow \downarrow CH_4})$$
[11]

Where GWP_s is the site global warming potential, \overline{NEE} is the average measured site net . ecosystem exchange, and $\overrightarrow{1}CH_4$ is the average measured site CH₄ flux.

GWP estimates were calculated for each site in each year, using the average measured seasonal fluxes from each of the sites.

CHAPTER 3: RESULTS

3.1 Environmental Variables

3.1.1 Temperature and Precipitation

The 2008 and 2009 study seasons were generally both cooler and wetter than the long term normals (1948-2009) for the months of May to August at the Upsala, Ontario meteorological station (Environment Canada, 2010). Average daily temperatures during May 2008, May 2009, and July 2009 fell below the 25th percentile (Figure 3.1). The minimum average daily temperatures during the sampling periods occurred during late May in both 2008 and 2009 (2.7°C and -0.2°C respectively) while the maximum daily average temperatures occurred on August 6th 2008 (24.3°C) and August 13th 2009 (22.7°C).

Overall, five of the eight study months exhibited higher levels of precipitation than average long term normals from the Upsala, Ontario meteorological station. In 2008, precipitation in the month of May was above the 80th precipitation percentile, with June and July occurring above the 90th percentile. In 2009, August was above the 90th percentile, while May 2009 experienced 252.4 mm of rain, the highest precipitation for that month for the previous 60 years. Total 2008 precipitation was 509.5 mm, with 2009 precipitation totalling 482.4 mm.

3.1.2 Water table position

Water table position in hollows at the control site remained between 0 and + 10 cm relative to the peat surface over the sampling seasons in both 2008 and 2009, while the average water table level in hummocks at the control site remained between -20 and -30 cm relative to the peat surface. Following peat-extraction at the experimental site in 2008, the water table at the experimental site rose gradually from May to August, fluctuating between approximately +10 and + 20 cm relative to the peat surface (Figure 3.2a). In the 2009 sampling season, minor fluctuations in water table level occurred at the experimental site, with no increasing trend in water table observed.

2009 water table levels exhibited the greatest fluctuations at the ditch site, with hollows dropping from approximately -10 cm to -40 cm at the driest period (Figure 3.2b). Ditch hummocks displayed a similarly drastic drop in water table level from -40 cm to -80 cm at the driest period. The water table levels at the ditch hollows and hummocks recovered slightly with several large precipitation events in late July, reaching -15 cm and -60 cm respectively.

3.2 Vegetation Biomass, Diversity, and Cover

3.2.1 Plant Biodiversity

Plant biodiversity increased at the control, experimental, and ditch sites throughout the 2009 sampling season (Table 3.1); the highest levels during 2009 occurred at the end of July. Average levels of species diversity during this final sampling period did not differ

largely between the control and experimental sites. The highest levels of plant biodiversity were observed at the control and ditch hummocks (1.63 and 1.50 respectively), followed by the experimental collars (1.47) and the control hollows (1.32). The ditch hollows displayed the lowest overall levels of plant biodiversity (0.72) in 2009.

The highest levels of species evenness were observed in hollows at the ditch and control sites (0.22 and 0.14 respectively), followed by hummocks at the ditch site (0.11); hummocks at the control site and collars at the experimental site displayed low levels of evenness, with values of 0.103 and 0.10, respectively.

3.2.2 Functional Species Cover

The greatest proportion of late growing season ground cover observed at the experimental site in both the 2008 and 2009 growing seasons consisted of no live vegetation (Figure 3.3). Wild flower and tree species did not occur at all in the experimental site; tree species were unable to survive the transplant and lost all of their leaves by late July 2008. The surface cover at the experimental site collars therefore consisted mainly of a combination of standing water, bare peat surface, and dead vegetation. Despite this, mosses including *Sphagnum fuscum, Sphagnum magellanicum, Sphagnum angustifolium*, and *Polytrichum strictum* made up the largest category of living vegetation at the experimental site in 2008 (approximately 50 %) and 2009 (approximately 30 %). In both sampling seasons hummocks at the control site were dominated by moss species, while control site hollows were dominated by sedges. The largest proportions of shrub species

were present at control and ditch site hummocks in each of the growing seasons. Sampled only during the 2009 growing season, the ditch hollows exhibited the greatest overall proportion (77 %) of no live vegetation of all sampling sites, consisting mainly of bare surface and dead vegetation.

3.2.3 Moss Growth

Average seasonal moss production, sampled only in 2009, was greatest at the experimental site for both *Sphagnum fuscum* (118 g m⁻²) and *Sphagnum magellanicum* (134 g m⁻²), although these rates are not significantly higher (p < 0.05) than those recorded for control site *Sphagnum* species (Figure 3.4). The lowest rates of seasonal production occurred at the ditch site for both *S. fuscum* and *S. magellanicum*, with *S. magellanicum* displaying the lowest overall production rate of 24 g m⁻². Production rates of both *S. fuscum* and *S. magellanicum* at the ditch site were significantly lower (p < 0.05) than *S. fuscum* production rates at the experimental site. At the control site, *S. fuscum* is the slower growing moss species, with overall average annual production rates for both control site *S. fuscum* and *S. magellanicum* indicating intermediate growth rates.

3.2.4 Leaf Area Index

Average site leaf area index increased for most species throughout the growing seasons in both 2008 and 2009. The experimental site had the lowest leaf areas of all sites in both 2008 and 2009, with nearly 50% less average leaf area for all functional groups combined (Figure 3.5). In both years, shrubs make up the greatest leaf area of all functional species types at the experimental site. Sedges make up the largest average leaf area at the control and ditch hollows, with wild flowers, shrubs, and some sedges making up the largest leaf area at the control and ditch hummocks. In both sampling seasons, the control hollows have the greatest average leaf area of all sites. In 2009, the ditch hollows experienced the greatest increase in sedge leaf area between late June and late July (from 0.09 to 0.27 m² m⁻²).

3.2.5 Functional Group Leaf Area

2009 average late summer leaf areas of functional groups (conifers, shrubs, and sedges) were highest at the control site, with sedges displaying the highest average leaf area of all functional species, at all sites (Figure 3.6). Late summer shrub leaf areas did not vary significantly between sites. No live conifer species were present at the experimental site, with the ditch site displaying the highest conifer leaf areas overall, although conifer leaf areas were not significantly different between sites.

Average conifer new growth lengths were highest at the ditch site (8.1 cm), followed by the control site (3.4 cm), with the experimental site conifers lacking any new growth at all.

Functional Group Soil Moisture

Individual functional species soil moisture levels were higher on average at the experimental site (61.6 - 97.5 % VMC), with experimental sedge moisture levels

exhibiting similar values to the control sedges (p = 0.997). Ditch hollows displayed exceptionally low soil moisture levels in late summer for all individual species locations (3.6 – 4.7 % VMC). All measured upper end VMC values (> 60 %) were not significantly different from each other, but were significantly different from all lower end VMC values (< 20 %).

3.2.6 Aboveground Biomass

Biomass total dry weight per unit area (g m⁻²) was highest for moss species at all sites apart from the control site hollows, where graminoids constituted the greatest biomass (Table 3.2). The control and ditch site hummocks had the greatest overall biomass per unit area (12,676 and 7944 g m⁻² respectively), with the control and ditch site hollows exhibiting the lowest overall biomass (151 and 776 g m⁻² respectively). At the experimental site, mosses represent the greatest overall biomass of all species (4885 g m⁻²), followed by shrubs (374 g m⁻²) and graminoids (46 g m⁻²).

3.3 Greenhouse Gas Exchange Following Peat Extraction and Acrotelm Transplant

3.3.1 Carbon Dioxide

Net Ecosystem Exchange

Following the extraction of peat in 2008, overall beginning of season NEE was not significantly different between sites in 2008, however, end of season NEE was significantly higher at the experimental site than both control hummocks and hollows (Figure 3.7). In both sampling years, on average control hummocks were overall sinks of

CO₂, while control hollows were sources. Between the 2008 and 2009 sampling season, NEE at the experimental site experienced a significant reduction in positive sourcing to the atmosphere. In 2009 the experimental site was nearly four-times less of a source of CO₂ than in 2008 (decrease from 7.5 to 1.9 g CO₂ m⁻² d⁻¹), and was no longer significantly different at the end of the season from control hollow NEE fluxes (p <0.001). However, at the beginning of the growing season in 2009, control site hummocks were a significantly greater sink of CO₂ than both control hollows and the experimental site, and are a significantly greater sink of CO₂ than the experimental site at the end of the 2008 sampling season. Throughout the 2009 sampling season however, the experimental site was not a significantly greater source of CO₂ than the control site hollows. There were no significant difference between 2008 and 2009 in net CO₂ fluxes at the control hollows (p = 0.67) and hummocks (p = 0.14).

Overall, the ditch site hummocks were not a significantly greater sink of CO_2 than the control or experimental sites throughout the sampling season. The ditch hollows are an overall source of CO_2 (3.3 g CO_2 m⁻² d⁻¹), although they are not significantly greater sources of CO_2 throughout the sampling season.

For 2008 NEE flux data, day of year was a significant factor determining NEE differences at the beginning of the season (p < 0.02, RM ANOVA) (Table 3.3). On the other hand, site is a significant factor determining differences in NEE at the end of the season (p < 0.003). In 2009, differences in NEE were a factor of site only, at the

beginning and end of the season (p < 0.02), although this was not observed at the ditch site.

Gross Ecosystem Productivity and Ecosystem Respiration

Average overall fluxes of ecosystem respiration and gross ecosystem production reveal that in 2008, the control site hummocks were significantly more productive than the control hollows and the experimental site throughout the sampling season, with control hummocks displaying the most negative GEP values (-11 g CO_2 m⁻² d⁻¹) (Figure 3.8a). Overall site ER in 2008 was not significantly different between any of the sites, although the experimental site has slightly higher average seasonal ER than the control site hummocks and hollows (8.1, 7.2, and 3.4 g CO_2 m⁻² d⁻¹ respectively) (Figure 3.8a). At the beginning of the sampling season in 2009 (27 May - 03 Jul), the control site hummocks were still a significantly greater sink of CO₂ (5.7 g CO₂ m⁻² d⁻¹) than control hollows (2.1 g CO₂ m⁻² d⁻¹) and the experimental site (2.7 g CO₂ m⁻² d⁻¹). At the end of the season however, productivity was not significantly different between the three sites (Figure 3.8b). As a result of the reduced GEP at the end of the season at the control hummocks, they were an overall lower sink of CO₂ than in 2008, although similar ER values were recorded, thus resulting in the higher overall NEE at this site. Throughout the 2009 season, the control hummocks and experimental site display similar levels of ER, although the experimental site is a significantly greater source of CO₂ than the control hollows (3.5 and 1.5 g CO_2 m⁻² d⁻¹ respectively). Recorded ER values decreased at the experimental site between 2008 and 2009 (Figure 3.8b).

The highest ER and GEP values in the 2009 sampling season were recorded at the ditch hummocks and hollows (5.5 and 6.9 g CO_2 m⁻² d⁻¹ respectively), with the ditch hummocks displaying significantly higher ER fluxes than both the control. and experimental site, and ditch hollows displaying significantly higher ER fluxes than the control hollows. However, the ditch site hummocks were significantly more productive than the control hollows and experimental site.

In 2008, day of year was the only factor determining differences in ER fluxes at the end of the season (p < 0.03, RM ANOVA) (Table 3.3). Day and site were factors for GEP differences, with day being significant at the beginning of the season (p < 0.02), and site being significant for both the beginning and end of the season (p < 0.01).

In 2009, day of year was a significant factor for ER at the beginning and the end of the season, and for tests involving the ditch site, while site was only significant for the ditch site tests (p < 0.001). Tests for GEP indicate that day of year was a significant factor only for the ditch site, while site was a significant factor at the beginning of the season, as well as for the ditch site tests.

Total Seasonal CO₂ Exchange

Modelled relations for GEP and environmental variables indicate that water table and leaf area were the dominant factors controlling measured GEP fluxes at the control site, while water table was the main controlling factor for GEP at the experimental site in 2008 and 2009, as well as at the ditch site in 2009 (Table 3.4a). In 2008 and 2009, estimated

environmental parameters from one control hollow and hummock were not included in the final estimated parameter averages due to unreasonable parameter estimation. Similarly, at the experimental site, one collar in both 2008 and 2009 was not included in the final environmental parameter averages. All other estimated model curve values for the selected control site and experimental site models were significant to p < 0.04 at individual collars.

Modelled relations for ER and environmental variables indicate that soil temperatures at varying depths including 2, 5, and 10 cm below the surface, were the dominant factors controlling ER at the control site (both hummocks and hollows) in both 2008 and 2009 sampling seasons, while water-table level was the main controlling factor at the experimental site in both years. Measured ER values at ditch site hollows were mainly controlled by water-table levels, while modelled results for the ditch site hummocks were not significant for any of the variables considered (p > 0.3). Modelled curve relations in 2009 at all collars at the control site, and three collars at the experimental site display significant decreasing respiration relations with increasing soil temperatures.

Modelled data for ER were not included for the control site hollow collar C2 in 2008, as a result of insignificant relations with modelled variables (p > 0.2). Average environmental parameter values for control site hollows were therefore obtained from two collars in 2008. At the experimental site, modelled ER data were not included in environmental parameter averages from three collars in 2008 (collars E1, E2, and E3) and three collars

in 2009 (collars E2, E4, and E6), due to insignificant relations with modelled environmental variables (p > 0.1). As a result, the total flux values for the experimental site consisted of an average of three collars in both 2008 and 2009. Due to insignificant modelled ER and environmental variable relation results at the ditch site hummocks, total seasonal flux values were not calculated at the ditch site.

Resulting modelled NEE values indicate that the experimental site was a source of CO_2 during the 2008 and 2009 growing seasons, while the control site hummocks were a small source in 2008, but a sink in 2009, and the control site hollows were a sink in 2008, but a source in 2009 (Table 3.5). The experimental site was a much larger source of carbon than the control site in both years, with fluxes decreasing minimally between the sampling seasons (218.1 g m⁻² to 207.9 g m⁻²).

3.3.2 Methane

Methane Ecosystem Fluxes

Following peat extraction in 2008, experimental site CH₄ production was significantly higher (p < 0.005) in comparison to the control site, averaging 161.7 mg CH₄ m⁻² d⁻¹ (Figure 3.9). During both sampling seasons, the control site hummocks and hollows were small sources of CH₄ (19.0 and 34.1 mg CH₄ m⁻² d⁻¹), with fluxes increasing insignificantly between 2008 and 2009.

In 2009, ditch hummocks and hollows were small sinks of CH₄, averaging -8.5 mg CH₄ m⁻² d⁻¹ at the hummocks, and -5.3 mg CH₄ m⁻² d⁻¹ at the hollows. Between the sampling seasons, the experimental site increased significantly in CH₄ production (p < 0.005), averaging 385 mg CH₄ m⁻² d⁻¹ for the season, a two-fold increase in production. As a result, experimental site CH₄ production remained significantly higher than control values in 2009.

 CH_4 ebullition concentrations at the experimental site ranged between 1 and 14 mg CH_4 m⁻² d⁻¹, while ebullition concentrations at the control site ranges between 0.1 and 6 mg CH_4 m⁻² d⁻¹, less than half of those occurring at the experimental site.

Methane and Environmental Variable Relations

Methane flux relations with environmental variables (soil moisture and temperature, air temperature, and water table level), were greatest for soil temperatures of 10 cm or greater in depth at all sites (Table 3.6). In 2008 and 2009, control site hummocks displayed significant relations with soil temperature at a depth of 15 cm (P < 0.05). The control hollows displayed the overall most significant relation with soil temperature at 10 cm (2008) and 15 cm (2009) ($r^2 > 0.7$; p < 0.01). None of the sites in 2008 and 2009 indicated any significant relation with hydrologic variables including water table level and soil moisture.

Ditch site hummocks and hollows did not display any significant relations with any of the environmental variables considered (p > 0.2).

3.3.3 Global Warming Potential of a Peat Extraction Site Following Acrotelm Transplant

Global warming potential estimates indicate that in 2008, the control site was an overall carbon sink (-1.0 g CO₂-e m⁻² d⁻¹), as a result of the greater overall sink of CO₂ despite the site being an overall source of CH₄ (Table 3.7). At the experimental site however, 2008 GWP indicates a strong positive forcing to the atmosphere (11.7 g CO₂-e m⁻² d⁻¹), as a result of the overall sourcing of both CO₂ and CH₄ at this site.

In 2009, the overall GWP of the control site increased, resulting in the control site becoming a weak source of carbon (0.2 g CO₂-e m⁻² d⁻¹), with the slight increase in both CO₂ and CH₄ positive fluxes from that site. Despite a large decrease in average CO₂ flux from 2008 to 2009, GWP at the experimental site increased slightly (11.9 g CO₂-e m⁻² d⁻¹), as a result of large increases in CH₄ flux (4.2 to 10.0 g CO₂-e m⁻² d⁻¹). Due to this increase in CH₄ flux at the experimental site, the experimental site remained a much larger carbon source than the control site two years post extraction.

In 2009 the ditch site was a weak source of CO_2 , though less so than the experimental site, but a weak sink of CH_4 as well. As a result, the ditch site was an overall weak source of CO_2 -e, equal to only 0.09 g CO_2 -e m⁻² d⁻¹ greater than the control site.

CHAPTER 4: DISCUSSION AND CONCLUSIONS

4.1 Rehabilitation of Vegetation...at a Peat Extraction Site Following Acrotelm Transplant

4.1.1 Biodiversity and Vegetation Cover

Although peatlands generally exhibit low levels of biodiversity in comparison to other wetland ecosystems (Moore, 2002), peat extraction and the fragmentation of peatlands have been shown to have significant long term impacts on regional biodiversity (Rochefort, 2000). Following the extraction of peat and transplant of the acrotelm at the study fen, differences in overall biodiversity between the control site and experimental (extraction) site were negligible. However, biodiversity at the ditch hollows suggest that under a water-table drawdown scenario, plant biodiversity decreases and evenness increases, because the primarily hydrophilic species that dominate these microforms are unable to thrive under these conditions, and only one or two species persist there.

Species cover and leaf area index are often utilised to determine the success of the reestablishment of vegetation on a reclaimed or harvested peatland. For example studies on the restoration of mined peatlands have indicated that the spontaneous re-colonisation of plants on a peatland generally does not occur without the availability of viable seeds and suitable propagules, (Cobbaert *et al.*, 2004) and the growth of *Sphagnum* requires stable, permanently wet and ombrotrophic conditions (McNeil and Waddington, 2003; Price *et al.*, 2003). The acrotelm-transplant reclamation approach, however, does not depend on, and is not limited by these factors, due to the direct transplant of acrotelm surfaces and maintenance of a high water table. As a result, inferences of re-establishment may be made simply from measurements of species cover and leaf area at the extraction site. Two years after peat extraction, the experimental site displayed the lowest leaf area of all sites, and the largest proportion of the experimental site consisted of a combination of standing water and bare peat surface. Low leaf areas and larger proportions of open water were likely due to the under-representation of graminoids in terms of proportional cover and total biomass at the experimental site. This was likely due in turn to the lack of a hummock-hollow microtopography at the experimental site, as a result of the difficulties in transplanting hollows. Thus, more open water exists where hollow microforms may persevere in natural conditions.

In addition, the experimental site lacks the presence of both wild flowers and trees. Most wildflowers at the study peatland are perennial plants that grow from bulbs, rhizomes, or fleshy roots (Johnson *et al.*, 1995). Thus, it is possible that these plants were unable to survive the initial transplant following extraction. Furthermore, immediately following the transplant, several trees were present in the experimental plot. However, due to the lack of an established subsurface peat layer and the floating peat mat structure of the experimental site, trees appear unable to survive and re-establish the root structures necessary for a successful transplant.

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Proportional species cover for hummocks at the ditch site was similar to that of the natural peatland site, although there was a slightly greater proportion of dead vegetation present. Due to the ability of *Sphagnum fuscum* to thrive even in dry conditions, the lowered water table and reduced soil moisture at this site do not appear to impede its survival. Moreover, plants with deeper root systems such as shrubs and trees were able to survive at this site despite the lowered water table, thereby explaining the higher overall leaf areas observed, and the greater overall new growth of *Larix laricina* at this site.

Conversely, ditch site hollows have been most affected by the lowered water table, evidenced by the large proportion of this site that is covered by dead vegetation; the generally hollow-dominating graminoid species represent a small proportion of the species cover, because they are not highly adapted to the dry conditions present in the ditch-site hollows. As a result, the exceptionally low soil moisture levels in late summer for all measured species at the ditch hollows resulted in the lowest overall sedge leaf areas of all sites, and the highest conifer leaf areas.

4.1.2 Mosses: Re-establishment and Growth

Despite lower leaf areas and larger proportions of open water, proportional species cover and total vegetation biomass at the experimental site was dominated by moss species. As a key-stone species in the ecology and functioning of peatlands, *Sphagnum* mosses do not readily colonise mined peatland sites (Ferland and Rochefort, 1997; Rochefort, 2000; Price *et al.*, 1998). Often, peatland restoration may not even occur without the reintroduction of these moss species, and thus peatland functioning cannot be restored (Ferland and Rochefort, 1997; Price *et al.*, 1998). Successful re-establishment of mosses may require the introduction of active *Sphagnum* diaspores, along with the re-wetting of the site in order to improve hydrological conditions (Chirino *et al.*, 2006).

As water table and moisture levels were not limiting at the experimental site, the rate of *Sphagnum fuscum* and *Sphagnum magellanicum* growth there was actually higher than at all other sites, including the natural control site. Several studies have identified that *Sphagnum* growth may be promoted where residual peat is buoyant (Lamers *et al.*, 1999; Smolders *et al.*, 2002; Tomassen *et al.*, 2003), thus allowing for a permanently waterlogged, but not flooded situation, as is present at the experimental site.

Adapted to dry conditions, *Spagnum fuscum* (Chirino *et al.*, 2006) had greater growth rates than *S. magellanicum* at the dry ditch site. However, under the moist conditions at the experimental site, *S. fuscum* growth rates were still able to exceed those of the fast-growing *S. magellanicum*, for which natural production rates were exhibited at the control site. This occurrence may be the result of processes of competition or facilitation, which have been shown to affect the growth of transplanted species (Chirino *et al.*, 2006). Furthermore, following the transplant of *Sphagnum* species onto bare peat surfaces at an abandoned peatland, Chirino *et al.* (2006) found that hummock *Sphagnum* species (*i.e.*, *S. fuscum*) tend to perform better following transplant than hollow species

(*i.e.*, *S. magellanicum*), although the current author knows of no previous studies into the establishment of moss for wet-extraction scenarios.

In addition, studies have noted the importance of the role of companion species in the successful re-establishment of *Sphagnum* and other plants at a restoration site (Ferland and Rochefort, 1997; McNeil and Waddington, 2003; Tuitilla *et al.*, 2004). In reintroducing *Sphagnum* species at a cutover bog site, Tuitilla *et al.* (2004) noted that *Sphagnum* regeneration and rates of photosynthesis benefit greatly from the presence of vascular plants. As such, it was further noted that the carbon balance of surfaces with transplanted *Sphagnum* is more sensitive to changes in water table than those dominated by cottongrass (Tuitilla *et al.*, 1999). McNeil and Waddington (2003) similarly found that in areas where *Sphagnum* was growing with companion species, growth and productivity were nearly double that of *Sphagnum* cushions growing without companion species.

4.2 Impacts of Peat Extraction on GHG Exchange

4.2.1 Changes to CO₂ Fluxes with Peat Extraction and the Controlling Factors

As discussed earlier, peat harvesting often leaves a peatland ecosystem in a semipermanently altered state, with the peatland becoming a persistent source of GHGs and losing its overall ability to accumulate and store carbon (Waddington *et al.*, 2009; Waddington *et al.*, 2002; Price *et al.*, 2003). Ecosystems reaching this altered state are often drained or cutover peatlands subjected to (dry) vacuum-harvesting or block-cut extraction techniques, combined with a lack of immediate restoration. Moreover, these ecosystems often lack a healthy vegetation surface and hydrological conditions that would allow for the re-establishment of *Sphagnum* (Waddington *et al.*, 2002). It has been estimated that peat-cutting techniques and extensive drainage practices result in an average annual flux of 8.5 Tg, with the combustion of peat fuel adding an additional 26 Tg (Gorham, 1991). These increased fluxes of CO_2 are due to a thickened oxic layer caused by drainage, resulting in increased aerobic decomposition of organic matter (Tuitilla *et al.*, 1999).

On the other hand, the novel acrotelm-transplant technique prevents the oxidation of remnant peat layers with the presence of a near-natural water-table level and the transplant of the vegetative acrotelm layer (Cagampan and Waddington, 2008). Strack and Waddington (2007) identify that the timescales involved for vegetation response to changes in water-table levels is largely unknown, and shifts in the water-table may often result in a shift to community composition, and therefore CO_2 fluxes. I thus hypothesised that with the extraction of peat and immediate rehabilitation using the acrotelm-transplant method, fluxes of CO_2 from the experimental site would be higher than at a nearby natural (control) site with initial extraction and disturbance of the peat layers, but would decrease over time.

Average fluxes at the experimental site were significantly higher than at the control site in 2008, but decreased four-fold between the 2008 and 2009 sampling seasons, with average fluxes becoming statistically no different from those observed at the control site hollows in 2009. Thus, with peat extraction and vegetation transplant at the experimental site, the resulting rise in water-table level relative to the peat surface appears to have allowed for relatively stable rates of productivity over the two years, thus indicating that the acrotelm-transplant method has allowed for a re-establishment of a near natural peatland functioning, in relation to productivity level.

While increased water-table levels are generally expected to decrease fluxes of CO_2 with the saturation of surface soils and the limitation of diffusion of oxygen into the peat (Chivers *et al.*, 2009), some research has shown that ER can be reduced in flooded areas with the decreased thickness of the acrotelm, while GEP, and possibly also ER, may increase due to changes in plant-community composition and substrate quality (Chivers *et al.*, 2009; Turetsky *et al.*, 2002). In the case of the experimental site in this study, the increased NEE, as a result of increasing ER following the extraction of peat and saturated moisture conditions, were likely due to the disturbance of substrates and changes to the plant community structure following transplant.

In contrast, at the ditch site (which emulates a water-table draw-down scenario), despite the fact that NEE fluxes were not significantly higher than at the control or experimental sites, the ditch hummocks and hollows displayed significantly higher respiration rates, namely the ditch hollows, although production was significantly higher at the ditch hummocks as well. This suggests that while increased oxidation is likely occurring at the ditch site, production at hummocks is also likely enhanced, thus indicating the importance of the presence and types of vegetation for GHG fluxes, as potentially indicative of the higher overall LAI at the ditch hummocks. Although several studies have demonstrated lowered GEP during water-table drawdown (Waddington et al., 2002; Gorham, 1991), others have found that lowered water table can in fact increase overall GEP, due to increases in the productivity of woody plants and the nutrient availability for plant uptake (Gorham, 1991; Thormann and Bayley, 1997; Weltzin et al., 2000). Furthermore, net photosynthetic rates may be greater amongst deciduous species – such as *Larix laricina* (Tamarack) which have a larger presence at the ditch site –than common hummock evergreen species (C. calyculata, L. Groenlandicum, and K. angustifolium) which are dominant at the control hummocks, thus potentially resulting in the higher photosynthetic rates at the ditch hummocks (see Moore et al., 2002). Also, the ditch hummocks have a large evergreen leaf area, in common with natural hummocks, which have been found to be more resilient to changes in moisture (Bubier et al., 2003). In addition, the high respiration rates at the ditch site hollows, where the dominant plants are sedges and graminoids, were likely due to the large drop in water-table levels alongside the outflow ditch, and thus reflect an increase in soil oxidation, and plant senescence and browning with moisture stress, as was similarly observed in Bubier et al. (2003).

Numerous studies (e.g., Bubier *et al.*, 2003; Rochefort *et al.* 1990; Vitt, 1990; Waddington and Roulet, 1996) have indicated that hollows display lower productivity in both fens and bogs. Indeed, hollows at the control and ditch sites acted as sources of CO_2 in both sampling years, while the hummocks were CO_2 sinks.

Although extensive research has been conducted into the environmental controls on CO_2 fluxes in terms of leaf area, vegetation biomass, water table and moisture controls, as well as soil and air temperature (e.g., Chivers et al., 2009; Khomik et al., 2009; Strack and Waddington, 2007; Bubier et al., 1993), overall changes to these controls on a seasonal basis are not as widely discussed. Results from this study suggest that controls on observed CO₂ fluxes differ on small temporal scales, *i.e.*, from spring to late summer. For example, differences in CO_2 fluxes between sites become significant later in the growing season, when leaf area was at near-peak levels, temperatures were higher, and PAR levels reached maximum rates on sunny days. Prior to this (in the spring and early summer), very little difference in observed net CO₂ flux occurred immediately following extraction in 2008. In 2009 when the experimental site fluxes were not significantly different from the control site hollows' at the beginning or end of the season, it is apparent that even at maximal LAI potential at the hollows, the experimental site was still nearly as productive. As a result, the experimental site is thus exhibiting a near-natural CO_2 flux rate in 2009.

GEP is largely controlled by PAR, LAI, water table and moisture levels (e.g., Chivers *et al*, 2009; Tuittila *et al.*, 2004; Blodau, 2002). In both sampling years at the majority of control and experimental site collars, and ditch site collars in 2009, GEP fluxes were predominantly controlled by water-table with 50 % of the variation in GEP explained by water-table level. Similarly, Tuitilla *et al.* (2004) and Lucchese *et al.* (2009) noted that although several environmental factors affect observed CO₂ fluxes, moisture conditions

are the most important factor controlling CO_2 fluxes as well as rates of carbon accumulation on restored peatland sites, likely due to the importance of water table and moisture on *Sphagnum* productivity (McNeil and Waddington, 2003; Strack *et al.*, 2009; Weltzin *et al.*, 2000). Especially in ecosystems where *Sphagnum* species are dominant, water-table and moisture considerations become most important in determining rates of photosynthesis and productivity (Tuitilla *et al.*, 2004). Water table is the dominant factor controlling the observed rates of GEP at the experimental site because *Sphagnum* growth rates there were higher than those at the control site, and productivity levels and percent cover were comparable between the control and experimental sites in the second year post-extraction.

Furthermore, in examining the modelled environmental relations with ER at the control site, the strongest relations were found to occur between soil temperatures at varying depths, with 10 cm being the maximum depth at which strong relations are observed. On the other hand, at the experimental and ditch sites, water-table level explained the greatest portion of variations in ER. As such, it is apparent that at sites where water-table levels are either higher or lower than those at natural sites, hydrological function plays a more important role in determining ER fluxes, while at natural sites (*i.e.*, the control site), temperature is the dominant factor.

Once again, despite significant relations between most modelled variables and ER, model r^2 values were low for all sites, ranging between 0.1 and 0.4. In addition, in 2009, all of

the collars exhibiting significant relationships with soil temperature at the control site, and three of four collars at the experimental site, displayed unusual, weak (average $r^2 =$ 0.3) <u>negative</u> relations with ER. While most studies report increasing ER with increasing temperatures (Chivers *et al.*, 2009; Crill *et al.*, 1988; Frolking and Crill, 1994), respiration and positive overall NEE fluxes are known to occur in winter, early spring, and fall, when vegetation is inactive and soil respiration is taking place (Carroll and Crill, 1997; Frolking *et al.*, 1998). As such, it is possible that in the spring, when temperatures are low and vegetation is only beginning to become active, respiration from soil and vegetation is amplified, and this decreases with mounting temperatures over the season, as plant production increases. However, as this was found to only occur in the 2009 growing season, the exact reason for this relationship is largely unknown. This finding may prove to be interesting in climate change studies with increasing temperature scenarios in northern peatlands.

4.2.2 Changes to CH₄ Fluxes with Peat Extraction and the Controlling Factors

Waterlogged, anoxic conditions in peatlands promote anaerobic decomposition and the production of CH₄, and thus make these ecosystems significant sources of atmospheric CH₄ (Ström *et al.*, 2005; Segers, 1998). CH₄ fluxes are highly variable, on both a temporal and spatial basis (Blodau, 2002), as both methane production and consumption are biological processes dependent on environmental conditions (Segers, 1998). As would therefore be expected, the control site hummocks and hollows were sources of CH₄ in both 2008 and 2009. Average fluxes at the control site were similar to those

reported in previous studies (e.g., Segers, 1998; Blodau, 2002; Strack and Waddington, 2007; Sundh et al., 1995) for natural poor fen sites, with a range of average CH₄ efflux between 20 and 40 mg CH₄ m⁻² d⁻¹. Control hollows emitted greater overall amounts of CH₄ than hummocks as a result of their higher water-table levels in relation to the peat surface, and thus a thicker anaerobic layer available for CH₄ production. Similarly, according to the dependence of CH₄ efflux on anaerobic conditions, the experimental site, possessing the highest water-table level of all sites, also had the largest flux of CH₄ with over 160 mg CH₄ m⁻² d⁻¹ produced on average in 2008, and increasing to nearly 400 mg $CH_4 \text{ m}^{-2} \text{ d}^{-1}$ in 2009. In a review on carbon cycling in peatlands, Blodau (2002) noted that values over $100 - 200 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ are uncommon, and restricted to ponds. In accordance with Blodau's findings (2002), these measured CH₄ values are unusual, but easily explained by the floating peat-mat characteristics and large amount of standing water present in the experimental site. In an examination of carbon emissions following various peat extraction methods, Waddington et al. (2009) similarly found that emissions of CH₄ were highest in areas of saturated and anoxic conditions, such as drainage ditches associated with the block-cut and vacuum-harvest techniques. Furthermore, with the occurrence of floating peat mat formations, Lamers et al. (1999) and others have identified that the ability for these mats of peat to remain buoyant is directly related to the accumulation of CH_4 bubbles in the peat subsurface. As a result, the buoyant properties of the experimental site are likely an indication of the high production of CH₄ at this location (Strack and Waddington, 2008). The rapid CH₄ production may also signify the high quality of the substrates, and higher pH at this site, which Tomassen et al. (2003)
found to be a direct determining factor of CH_4 production rates in a floating peat mat formation.

Furthermore, although the water-table level and soil temperature did not change significantly between sampling years at the experimental site, a significant increase in CH_4 production was observed between 2008 and 2009. As a result, the reason for the large difference between 2008 and 2009 average CH_4 flux at the experimental site is largely unknown. CH_4 is dependent indirectly on soil temperature, vegetation type present, chemical characteristics of the peat, and the position of redox boundaries in direct relation to the water table (Bubier *et al.*, 2003; Whiting and Chanton, 1993; Blodau, 2002). As chemical characteristics and the position of redox boundaries were not directly quantified during this study, these characteristics may provide insight to the measured fluxes.

In addition, in an examination of the variation in CH_4 point flux measurements at the experimental site, it becomes apparent that large CH_4 peak events are occurring on a regular basis at the experimental site. Studies have identified that large volumes of gas may accumulate in peatlands and vent to the atmosphere in discrete events, with over 40 g CH_4 may be recorded in a span of minutes or hours (Baird *et al.*, 2004).

Recent studies have indicated the importance of mechanisms of ebullition of biogenic gas bubbles in the flux of CH₄ to the atmosphere (Fechner-Levy and Hemond, 1996; Baird *et* *al.*, 2004; Kellner *et al.*, 2006). Often, this process of ebullition is suggested to account for the large variability in measured spatial and temporal fluxes (Fechner-Levy and Hemond, 1996). Although measurements of CH_4 ebullition from the study sites were largely unsuccessful due to bubble meter malfunctions, the high variation and large CH_4 fluxes in 2008 and 2009, as well as the inter-annual variation, may be attributable to occasional capture of ebullition events, as observed from ranges of ebullition events captured at the experimental site (between 1 and 14 mg CH_4 m⁻² d⁻¹), as have been similarly reported in previous studies (Baird *et al.*, 2004).

Measurements of CH_4 ebullition have previously been reported to be significantly higher at flooded sites (Waddington and Roulet, 1996; MacDonald *et al.*, 1998; Bubier *et al.*, 1995), which may potentially explain the measured fluxes at the experimental site.

On the other hand, CH₄ flux measured at the ditch site indicated that at both hummocks and hollows, on average, CH₄ is being consumed at rates of between 5 and 10 mg CH₄ m⁻² d⁻¹. As a result of the large oxic layer at the ditch site due to the significant drop in water table, methane production is largely limited and conditions are available for consumption to occur (Segers, 1998). In a water-table drawdown experiment, Strack and Waddington (2007) concluded that the lowering of the water table at an experimental site resulted in overall reduced CH₄ emissions with an increase in the oxic zone.

Along with the importance of microbial activity to CH_4 fluxes from a peatland, considerations of hydrologic and thermal regimes in the peat profile are essential to

understanding spatial and temporal variations in fluxes (Bubier *et al.*, 1995; Segers, 1998). Despite the dependence of CH₄ production and emission on anoxic conditions, there was no clear observed relationship between CH₄ and water-table position at the majority of the site collars examined. Strack and Waddington (2007) similarly found a lack of a clear relationship to hydrologic conditions, noting that this was likely due to the high spatial variability in CH₄ emissions, as found in this study as well. Significant relations were, however, found to occur between soil temperature and CH₄ efflux. In addition, most sites displayed the strongest relations with soil temperatures at 10 to 15 cm below the peat surface, as the majority of CH₄ production occurs at or below the water table, but not above (Segers, 1998). Significant relations with environmental variables were not found at all at the ditch site ($r^2 < 0.1$), likely due to the high variability in measured fluxes at both hummocks and hollows. Fluxes at this site are thus likely most dependent on the microbiological characteristics of methanotrophs.

4.3 Overall Implications of the Extraction of Peat with Acrotelm Transplant

4.3.1 Implications for Vegetation Survival and Growth Following Peat Extraction

Following peat extraction using block cutting, vacuum harvesting, and other similar techniques, a healthy vegetative surface is all but entirely removed from the extraction site (Price *et al.*, 1998; Cobbaert *et al.*, 2004; Tuitilla *et al.*, 2004). Restoration techniques at these cutover and water-table drawdown peatlands are often difficult, because *Sphagnum* fragments need to be transplanted manually (Campeau and Rochefort, 1996), outflow ditches need to be blocked (Price, 1997), and surface microtopography needs to

be altered, often with shading devices, straw mulches, and companion species (Ferland and Rochefort, 1997; Campeau and Rochefort, 1996; Price *et al.*, 1998). It has also been stated that for *Sphagnum* to recolonise cutover peat surfaces successfully, water tables and soil-moisture levels need to be high and stable (Campeau and Rochefort, 1996; Price, 1997; Price and Whitehead, 2004). However, Tuitilla *et al.* (2004) identify that the maintenance of an optimum water-table level over an entire growing season in field conditions is often not easy, or even possible.

Thus, from an ecological restoration perspective, the application of the acrotelmtransplant method for the re-establishment of a healthy acrotelm on a peat harvesting site is ideal. The overall negligible difference in measured biodiversity between the experimental and control sites indicates the overall success of vegetation transplant on biodiversity, with utilisation of this method. In addition, the high growth rates and large proportional cover of mosses at the experimental site is also indicative of the success of transplant for the keystone moss species. The transplant of the acrotelm in its un-altered state also allows for the inclusion of vascular and graminoid species, which have been shown to act as successful companion species to *Sphagnum*, and aid in the regeneration of the C sink function of the peatland (Tuitilla *et al.*, 2004).

Despite the lack of tree species during the second year following transplant, over time with the continued C accumulation and regeneration of a catolem layer, it is expected that trees will re-establish at the experimental site.

The current study suggests that, from an ecohydrological perspective at least, the acrotelm-transplant method may allow for a rapid and successful re-establishment of keystone and peatland-specific species, and thus aid in the overall success of the restoration of the harvested site.

4.3.2 GHG Fluxes and Peat Extraction: Importance and Dependence on Environmental Variables

Alterations to GHG fluxes following peat extraction is one of the environmental concerns surrounding the utilisation of peat as an energy source in Ontario. The exploitation of these resources can increase CO₂ fluxes by 400% (Waddington and Price, 2000), and Cleary *et al.* (2005) estimated that it may take approximately 2000 years for the peatland to recover the lost carbon. Increased losses of GHGs after peat extraction arise from changes in land use, the combustion of fossil fuels, and peat decomposition, and therefore contribute to climate change (Cleary *et al.*, 2005). Despite this, life-cycle analyses of peat extraction using the acrotelm-transplant method indicate that this method has the lowest annual carbon emissions, in comparison to block-cut and vacuum-harvest techniques (Waddington *et al.*, 2009). Short-term impact findings in this study for GWP indicate that positive forcing to the atmosphere is occurring following extraction, with contributions from CO₂ decreasing over time. In terms of CH₄ production, highly saturated floating peat mat conditions, as a worst-case-scenario representation, allow for strong contributions from CH₄, a more significant GHG, to climate forcing. It may therefore be predicted that future GWP from this site will decrease with a lower stabilised water table, accumulation of a catotelm layer, increasing productivity, and reduced respiration.

4.4 Concluding Remarks

In analysis of the short-term impacts of peat extraction for energy utilisation in a northern Ontario peatland, implications for vegetation characteristics and GHG exchange were considered. I therefore conclude that the acrotelm-transplant technique is optimal for a rapid re-establishment of vegetation on an extracted surface, with previous studies noting that the successful regeneration of vegetation indicates the overall efficiency of restoration of this method, in comparison to other techniques (Waddington *et al.*, 2009). In addition, with the general difficulty of maintaining optimal water-table levels in field conditions at a restoration site, it may be noted that the acrotelm-transplant method avoids this problem, and allows for the healthy growth of bryophytes, which are more sensitive to variations in water level, as directly related to peatland carbon balance (Tuitilla *et al.*, 2004).

In addition, GHG and GWP measurements indicate the potential for a decreasing forcing of GHGs on the atmosphere with the AT method, with the expectation that the carbon accumulation functions will be restored over a shorter period of time than that of other extraction techniques. However, it is also important to note that although the present accumulation of carbon in Canadian peatlands greatly exceeds that of exploitation (Price *et al.*, 2003), the potential scale and extent of usage of peat for energy purposes is still largely unknown. Such utilisation has not yet been attempted in Canada, and thus ecosystem impacts on larger areas, as would be required to sustain energy requirements, cannot currently be determined. It is thus recommended that a combination of field studies and modelling techniques be applied in order to determine the potential large scale impacts of peat extraction, and the requirements for a sustainable usage of this resource.

In addition, although the potential impacts to the ecological functioning of peatlands have been the focus of this thesis, considerations for the economic and logistical feasibility for the usage of peat as an alternative energy source have not been examined here. Furthermore, impacts of climate change on peatland functioning have also not been mentioned, and need to be considered for the future usage of this resource, because overall impacts of increasing temperatures and changes to precipitation patterns on peatlands are still largely unknown.



Figure 2.1: Peat extraction and acrotelm transplant technique (a-d1) and the technique as conducted at the study peatland near Upsala, Ontario (April 2008) (a-d2, e) (Adapted from Cagampan and Waddington, 2008).



Figure 2.2: Relative locations of all sites and their respective sampling locations at the study peatland (not to scale).



Figure 3.1: Monthly total precipitation and monthly average temperature percentiles for each of the eight study period calendar months in the 2008 and 2009 sampling periods. Precipitation and temperatures are expressed as percent ranks from 0 to 100% of 30-year normals from the Upsala weather station.



Figure 3.2a: 2008 water table levels (cm) in relation to the peat surface (denoted here by the axis through 0) and precipitation (mm) (bars) for control hummocks, hollows, and the experimental site.



Figure 3.2b: 2009 water table levels (cm) (lines) in relation to the peat surface (denoted here by the axis through 0) and precipitation (mm) (bars) for all sites.



Figure 3.3: Late season proportional species cover (0 to 1) at all sites for 2008 (a) and 2009 (b). The "other" category includes carnivorous, wild flower, horsetail, tree, and rose species.



Figure 3.4: Measured late season annual moss production rates (g m⁻²) at the control (C), ditch (D), and experimental (E) sites for *S. fuscum* (F) and *S. magellanicum* (M). Letters denote statistical differences (p < 0.05).



Figure 3.5: Leaf area index $(m^2 m^{-2})$ for all sites in 2008 (a) and 2009 (b) by species category. The "other" category includes horsetails and wild flowers.



Figure 3.6: 2009 Late summer average functional species leaf area index (bars) for control, ditch, and experimental site conifer (black), shrub (grey), and sedge (white) species, and their corresponding average volumetric moisture content (points). Letters denote statistical differences (p < 0.05).



Figure 3.7: Measured mid-day NEE fluxes (g $CO_2 \text{ m}^{-2} \text{ d}^{-1}$) at all sites for 2008 and 2009 sampling seasons. Letters denote statistical differences (p < 0.05). The first (lowercase) letter in brackets represents the beginning of season (27 May – 03 Jul) significance, while the second letter in brackets (uppercase) represents the end of season significance (07 Jul – 13 Aug). Statistical significance at the ditch site was conducted for the entire 2009 season (uppercase letters), and are compared with uppercase letters only. Inter-annual statistical comparisons are not included here.



Figure 3.8 a & b: Average ER and GEP fluxes for 2008 (a) and 2009 (b) for all sites and the corresponding RM Anova statistical significance (p < 0.05). The first (lowercase) letter in brackets represents the beginning of season (27 May – 03 Jul) significance, while the second letter in brackets (uppercase) represents the end of season significance (07 Jul – 13 Aug) for ER (top) and GEP (bottom). Statistical significance at the ditch site was conducted for the entire 2009 season (uppercase letters), and are compared with uppercase letters only. Inter-annual statistical comparisons are not included here.



Figure 3.9: Average methane fluxes (mg CH₄ m⁻² d⁻¹) in 2008 and 2009 for all sites. Letters denote statistical differences (p < 0.05).

TABLES

Table 3.1: Average site Shannon-Weiner biodiversity (H'), evenness (pi), and their corresponding standard deviations (SD), as well as total site species richness (S'). All values are dimensionless.

SITE	H'	SD (H')	pi	SD (pi)	S'
C Holl	1.32	0.27	0.14	0.18	15
C Hum	1.63	0.27	0.10	0.14	13
D Holl	0.72	0.07	0.22	0.32	5
D Hum	1.50	0.19	0.11	0.15	16
E	1.47	0.29	0.10	0.16	14

	Site Biomass (g m ⁻²)									
Species Group	E	C Hum	C Höll	D Hum	D Holl					
Moss	4885	12676	151	7944	776					
Shrub	374	326	9	585	31					
Graminoid	46	85	357	55	131					
Carnivorous	13	30	1	0	0					
Horsetail	2	3	31	0	0					
Wildflower	0	18	0	0	0					

Table 3.2: Biomass dry weights (g m⁻²) for groups of species at each site.

Table 3.3: Repeated Measures ANOVA within subjects effects (Day & Site) from measurements of net ecosystem exchange (NEE), ecosystem respiration (ER), and gross primary production (GEP) across sites in 2008 and 2009, with seasons split into beginning (B) (May 27 – Jul 3) and end (E) (Jul 3 - Aug 13) of season. Tests involving the ditch site were conducted for the entire season (A*) (01 Jun to 29 Jul 2009). Significant values are in bold.

<u> </u>		2008				2009						
		Day			Site			Day			Site	
	df	<u>F</u>	P	df	<u> </u>	<u> </u>	df	<u> </u>	<u> </u>	df	<u> </u>	<u> </u>
NEE												
В	2.86	3.96	0.02	2	2.71	0.12	3.31	1.05	0.39	2	7.99	0.01
E	9	1.52	0.16	2	11.52	0.003	3.30	0.46	0.73	2	8.01	0.02
A*	n/a	n/a	n/a	n/a	n/a	n/a	1.61	1.51	0.25	4	2.47	0.11
ER			<u> </u>			<u> </u>			<u> </u>		<u> </u>	
В	1.78	1.29	0.30	2	0.65	0.54	3.63	6.91	0.01	2	1.61	0.26
Е	2.23	4.17	0.03	2	1.13	0.37	0.38	3.44	0.03	2	4.70	0.059
A*	n/a	n/a	n/a	n/a	n/a	n/a	1.81	6.79	0.01	4	10.51	0.001
GEP		<u> </u>		_,		<u></u> _	<u></u>				· · · · · · · · · · · · · · · · · · ·	<u> </u>
В	3.47	3.55	0.02	2	7.16	0.01	3.26	2.54	0.07	2	9.97	0.007
Ε	3.99	2.24	0.08	2	12.20	0.003	3.18	0.89	0.47	2	2.27	0.19
A*	n/a	n/a	n/a	n/a	n/a	n/a	1.68	4.05	0.04	4	4.51	0.02

AVG

Table 3.4a: GEP model parameters, r^2 , and p values for control (i) and experimental (ii) site collars and the average model equation parameter for control site hummocks and hollows (equation # 8) and the experimental site (equation # 6) for 2008 and 2009 data. Data not incorporated in averages are in bold.

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(i)							
2008 Collars	Q	GP max	kLAI	kWT	N	r^2	p
<u>C1</u>	0.003	2.45	-0.27	13.47	29	0.57	< 0.0001
C2	0.12	2.62	0.03	-5.13	22	0.44	0.0004
C3	0.17	5.35	0.35	16.62	29	0.34	0.001
C4	0.05	31.92	0.01	12.96	22	0.67	< 0.0001
C5	0.16	19.3	0.12	7.01	29	0.42	< 0.0001
C6	33.49	99235	2468.7	-3.81	17	0.81	<0.0001
AVG HOLL	0.09	17.27	0.02	3.92			
AVG HUM	0.16	12.32	0.23	11.81			
2009 Collars							
<u>C1</u>	0.004	9.79	-0.05	11.58	38	0.64	<0.0001
C2	0.01	3.67	0.07	-0.04	20	0.16	0.04
C3	0.003	7.54	0.15	18.07	28	0.4	0.0002
C4	0.01	4.21	0.12	0.14	26	0.31	0.002
C5	0.01	4.07	0.0007	11.24	29	0.36	0.0003
C6	0.0008	819.58	-0.59	0.11	22	0.87	<0.0001
AVG HOLL	0.01	3.94	0.09	0.05			
AVG HUM	0.01	5.80	0.08	14.65			
							_
(ii)							
2008 Collars	k	GP max	uP	tP	N	\mathbf{r}^2	p
El	414.13	9.83	3.42	-0.5	25	0.36	0.001
E2	214.83	6.5	12.61	7.21	19	0.55	0.0001
E3	119.94	11.89	12.7	4.24	25	0.18	0.02
E4	710.11	9.81	31.57	11.75	21	0.87	< 0.0001
E5	224.52	53.03	-179.84	86.65	27	0.33	0.01
E6	72.01	26.64	70.34	27.68	27	0.52	< 0.0001
AVG	306.2	12.93	26.13	10.08			
2009 Collars							
E1	569.93	8.1	2.67	4.06	27	0.48	0.001
E2	626.73	17.41	11.53	13.27	22	0.35	0.002
E3	592.74	136.49	67.93	25.57	24	0.71	<0.0001
E4	418.91	6.76	28.98	2.56	23	0.33	0.002
E5	266.81	9.97	14.24	7.07	34	0.59	< 0.0001
E6	199.02	15.72	14.91	-0.51	23	0.44	0.0003

14.46

5.29

11.59

416.28

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Table 3.4b: ER model parameters, r^2 , and p values for control (i) and experimental (ii) site collars and the average model equation parameter for control site hummocks and hollows (equation # 10) and the experimental site (equation # 9) for 2008 and 2009 data. Data not incorporated in averages are in bold.

(i)	~~.					
2008 Collars	T Depth (cm)	A	010		r^2	
C1	5	3.03	1.69	23	0.13	0.05
C2	5	7.24	0.72	23	0.05	0.16
C3	5	1.33	2.81	22	0.32	0.003
C4	5	1.26	2.16	25	0.1	0.05
C5	5	3.05	1.62	24	0.13	0.04
C6	10	1.11	1.96	24	0.12	0.05
AVG HOLL	5	1.18	2.06			
AVG HUM	5	2.47	2.04			
2009 Collars						
C1	2	11.27	0.52	25	(-)0.37	0.001
C2	10	4.86	0.46	25	(-)0.18	0.02
C3	2	9.72	0.4	24	(-)0.42	0.0003
C4	10	8.67	0.23	24	(-)0.31	0.002
C5	10	7.69	0.57	23	(-)0.26	0.01
C6	2	3.07	0.46	25	(-)0.1	0.05
AVG HOLL	10	5.54	0.38			
AVG HUM	2	9.56	0.5			
_(ii)						
2008 Collars	ER Max	uR	tR	<u> </u>	r²	p
E1	5.25	2.32	3.27	23	0.01	0.31
E2	9.2	7.49	17.11	23	0.08	0.09
E3	23.14	24.09	21.23	23	0.05	0.16
E4	8.7	39.86	10.09	23	0.33	0.002
E5	6.51	26.84	3.9	23	0.11	0.05
E6	8.89	14.06	10.56	22	0.31	0.003
AVERAGE	8.03	26.92	8.18			
2009 Collars						
E1	5.77	-3.59	8.82	24	0.15	0.03
E2	2.99	7.52	9.7	24	0.03	0.21
E3	8.0	14.44	8.62	24	0.27	0.005
E4	2.61	-24.4	113.38	23	0.0004	0.46
E5	4.48	18.81	5.31	25	0.2	0.01
E6	96.50	-41.69	22.22	24	0.05	0.15
AVERAGE	6.09	9.89	7.58			

SITE	2008	2009
	Avg Total Flux g CO ₂ m ⁻²	Avg Total Flux g CO ₂ m ⁻²
C Hum	11.5	-68.2
C Holl	-83.7	142.8
Exp	218.1	207.9

Table 3.5: Average of total NEE seasonal fluxes (g $CO_2 m^2$) for control hummocks, hollows, and the experimental site.

Table 3.6: Most significant summarized CH₄ relations with environmental variables (soil temperature (Ts), moisture, air temperature, and water table level) and corresponding r^2 , N and P values. Bin size indicates the size of the averaged bin ranges for each variable. Ditch site variables were not binned due to the lower overall sample sizes. * indicates insignificant values where p > 0.05.

Year	Site	Variable	\mathbf{r}^2	N	Bin Size	P (one- tailed)
	Con Hum	Ts 15 cm	0.459	7	2 °C	0.047
2008	Con Holl	Ts 10 cm	0.783	6	2 °C	0.010
2000	E	Ts 15 cm	0.567	6	2 °C	0.042
	Con Hum	Ts 15 cm	0.736	8	2 °C	0.003
	Con Holl	Ts 15 cm	0.939	7	2 °C	0.0002
2009	Ε	Ts 15 cm	0.531	7	2 °C	0.032
	Dit Holl	Ts 25 cm	0.037	11	N/A	*0.286
	Dit Hum	Ts 25 cm	0.056	10	N/A	*0.255

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SITE	Average CO ₂ (g CO ₂ m ⁻² d ⁻¹)		Averaş (g CO ₂ -6	ge CH ₄ e m ⁻² d ⁻¹)	GWP (g CO ₂ -e m ⁻² d ⁻¹)	
	2008	2009	2008	2009	2008	2009
Control	-2.4	-1.6	1.4	1.9	-1.0	0.2
Experimental	7.5	1.9	4.2	10.0	11.7	11.9
Ditch	-	0.7	-	-0.4	-	0.3

Table 3.7: Global warming potential estimates for a 26x carbon equivalence from average measured NEE and CH_4 flux values for each site in 2008 and 2009.

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