NET ECOSYSTEM CO_2 EXCHANGE
IN
NATURAL, CUTOVER AND PARTLY RESTORED PEATLANDS

By
KEVIN WARNER, BES

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AUTHOR: Kevin D. Warner, BES. (University of Waterloo)

SUPERVISOR: Assistant Professor J. M. Waddington

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ABSTRACT

Peatlands are an important component of the global carbon cycle, storing 23 g C m^{-2} yr^{-1} to comprise a global carbon pool of approximately 455 Pg. Peat drainage and harvesting results in removal of surface vegetation, thereby reducing gross photosynthesis to zero. Moreover, lowering the water table increases carbon oxidation. Consequently, peatland drainage and mining can reduce or eliminate the carbon sink function of the peatland. In the first part of this study, net ecosystem CO_2 exchange was studied in a natural (NATURAL), two-year (YOUNG) and seven-year (OLD) post cutover peatland near Ste. Marguerite Marie, Quebec during the summer of 1998. Although the NATURAL site was a source of CO_2 during the study season, CO_2 emissions were 270 to 300% higher in the cutover sites (138, 363, and 399 g CO_2-C m^{-2}; NATURAL, YOUNG and OLD, respectively). Active restoration practices and natural re-vegetation of peatlands have the potential to return these ecosystems to net carbon sinks by increasing net ecosystem production (NEP) and therefore decreasing CO_2 emissions to the atmosphere. Net ecosystem CO_2 exchange in a natural (NATURAL) peatland and a partly restored peatland (REST) near Ste. Marguerite Marie, Quebec, was compared with a naturally re-vegetated peatland (RVEG) near Riviere-du-Loup, Quebec. Ecophysiological parameters indicate that the REST site was more than twice as productive as the natural LAWNS and three times as the RVEG site (GP_{max}=18.0, 8.3, and 6.5 g CO_2 m^{-2} d^{-1}, respectively). These results indicate that active restoration improves carbon sequestration over natural re-vegetation but that the net carbon sink function at both sites has not been restored. The presence of *Sphagnum* cover at the RVEG site resulted in a significant decrease in net ecosystem respiration (NER),

indicating the potential for decreasing soil respiration at restored cutover sites through increasing the volumetric soil moisture content.

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Many thanks to the Beastie Boys for their inspirational motivation getting me through a long field season, remembering that you don't have to be the best, just tenacious and original to succeed.

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CHAPTER 1

INTRODUCTION

1.0 INTRODUCTION

Globally northern peatlands cover approximately 346 x 10^6 ha (Gorham, 1991).

Historically, these peatlands have stored large quantities of carbon (455 Pg C),

approximately one third of the world's soil carbon, and currently represent a net sink

(0.076 Pg C yr^{-1}) of atmospheric carbon (Gorham 1991). Drainage of peatlands for

agriculture, forestry, horticulture, and fuel and the subsequent abandonment of these

cutover peatlands leaves an environment not conducive to natural regeneration of

Sphagnum mosses (Rochefort *et al*, 1995; Price, 1996; Bugnon *et al*, 1997; Ferland and

Rochefort, 1997; Quinty and Rochefort, 1997). Consequently, the ability for peatlands to

act as net carbon sinks has been reduced as net ecosystem production (NEP) was reduced

to zero. Moreover, because drainage promotes aerobic decomposition of the peat through

enhanced heterotrophic soil respiration, these peatlands become a net source of CO_2

(Billings *et al*, 1982; Moore and Knowles, 1989; Gorham, 1991; Nykänen *et al*, 1995;

Silvola, 1996; Silvola *et al*, 1996; Nykänen *et al*, 1997). Armentano and Menges (1986)

indicate that the net carbon sink has been reduced by 21 to 33% in Finland and Russia, by

50% in Western Europe, and completely lost in Central Europe due to drainage and

cutover operations. Considering the total area of drained and cutover peatlands

worldwide is 11.6 x 10^6 ha or 3.3% of the total peatlands (Gorham 1991), there is a

potential for peatland drainage to significantly alter the global carbon cycle.

Recent studies in peatland restoration have found moderate success with the use of various mulch covers (Price *et al*, 1998; Rochefort *et al*, 1995), creating surface microtopography (Ferland and Rochefort, 1997; Quinty and Rochefort, 1997), and shelter techniques (Bugnon *et al*, 1997). Moreover, blocking drainage ditches (LaRose, *et al*, 1997) can aid in partly restoring the natural hydrology and some re-colonization of native peatland species (Campeau and Rochefort, 1996). By restoring the *Sphagnum* moss, there is the potential to return these ecosystems to a net carbon sink.

Consequently, there is a need to gain a greater understanding of the processes controlling net CO_2 ecosystem exchange and carbon storage within natural, cutover and restored peatlands. The general goals of this thesis are to determine the magnitude of carbon oxidation in abandoned cutover peatlands and the effect that peatland restoration has on the net CO_2 ecosystem exchange in these ecosystems.

1.1 *Peatland Ecosystems*

Wetlands in Canada are classified as "land that has the water table at, near, or above the land surface or which is saturated for a long enough period to promote wetland or aquatic processes as indicated by hydric soil, hydrophytic vegetation, and various kinds of biological activity that are adapted to the wet environment" (Tarnocai, 1980). In Canada, peatlands are classified as wetlands when organic soils exceed a depth of 40 cm (National Wetlands Working Group, 1988). Canada contains approximately 110 to 130 million hectares of peatlands (35% of the world's total), with the second largest peat resources in the world (Mitsch and Gosselink, 1993; Gorham, 1991). Globally, peatlands

are most commonly found in mid-latitudinal areas (between 45 and 75°N), with the largest extent of Canadian peatlands located in the Hudson Bay Lowlands and throughout the boreal forest region.

In Canada, peatlands are classified into bogs, fens, and sometimes swamps (National Wetlands Working Group, 1988). Bogs are ombrotrophic, generally raised wetlands that are isolated from regional groundwater but with a water table at or near the surface whereas fens are characterized by a high water table near or above the surface, forming in depressions therefore receiving minerotrophic water inputs from adjacent upland areas (National Wetlands Working Group, 1988). Swamps have seasonally standing or gently moving waters with both subsurface and overland flow occurring at different periods throughout the year, resulting in a nutrient rich ecosystem (National Wetlands Working Group, 1988).

Although not the most productive ecosystems when compared to forests (Ruimy *et al*, 1995; Frolking *et al*, 1997), peatlands are an important net global atmospheric CO_2 sink (Gorham, 1995). Due to their permanently saturated conditions, the decomposition rates in peatlands are low, allowing for a net accumulation of organic carbon in the formation of peat. Gorham (1991) estimates the total long term atmospheric carbon sink in natural peatlands to be 0.1 Pg yr^{-1} or 29 g C m^{-2} yr^{-1}. This has been estimated to have decreased over the last few centuries to 23 g C m^{-2} yr^{-1} (Gorham, 1991). Peatland formation occurs through two processes: *i*) terrestrialization, where peatland encroaches on adjacent upland areas, and *ii*) padudification where adjacent open water areas are slowly covered (Mitsch and Gosselink, 1993). These processes can often lead to two

distinct layers in a peatland: the acrotelm and the catotelm (Clymo, 1984). The acrotelm is the uppermost layer that is seasonally unsaturated, high porosity, relatively high hydraulic conductivity, low bulk density, and rich in organic material (labile carbon). This is the active layer where carbon uptake and most decomposition occurs owing to more labile carbon and aerobic conditions. Surficial bulk density in *Sphagnum* peat has been found to be approximately 0.02 g cm^{-3}, holding 95 to 100% water at saturation (Boelter, 1964). The catotelm is a layer which is permanently saturated, with a high bulk density, low hydraulic conductivity and porosity. Moreover, this zone has a higher mineral content and lower organic content than the acrotelm as much of the organic carbon has been mineralized (Clymo, 1984). Below 25 cm, bulk density is approximately 0.25 g cm^{-3} in decomposed peat, holding 80 to 90% water by volume at saturation, with many small pores which are not easily drained at low suction (Boelter, 1964). Approximately 10% of plant material may reach the catotelm as the peatland grows where it undergoes anaerobic decomposition and methanogenesis (Clymo, 1984). Isotopic gas analysis from samples in the catotelm found that gases are younger than the adjacent peat in the catotelm, and are from downward transport of labile carbon in pore water DOC (Charman *et al*, 1994). Only very slow decay occurs in the catotelm as decomposition is inhibited by oxygen deficit (Mulqueen, 1986). Oxygen penetration in saturated peat is much slower because of a much lower rate of diffusion of gases in water compared to air (Svensson and Sundh, 1992).

Carbon dioxide (CO_2) is the most abundant trace greenhouse gas, representing approximately 90% of the carbon cycled in a peatland (Gorham, 1991). Consequently,

peatlands, which have been found to be both significant sinks and sources for CO_2 (Hamilton *et al*, 1994; Kim and Verma, 1992; Moore and Knowles, 1987; Oberbauer *et al*, 1991; Waddington and Roulet, 1996; Bellisario *et al*, 1998; Whiting *et al*, 1991), are an important component of the global carbon cycle.

1.2 *Peatland carbon balance*

The carbon balance of a peatland can be represented as:

$$\Delta C = CO_{2(NEP)} - CO_{2(NER)} - CH_4 - DOC \tag{1}$$

CO_2 is fixed by plants through the process of photosynthesis to form organic carbon with net ecosystem production (NEP) measured as the difference between total uptake of atmospheric CO_2 and the loss of CO_2 via plant, root and soil respiration (NER). The rate of NEP is dependent on the level of photosynthetically active radiation (PAR) (Bellisario *et al*, 1998; Frolking *et al*, 1998; Ruimy *et al*, 1995; Waddington and Roulet, 1996), and will vary throughout the growing season with changes in leaf area and other ecophysiological parameters. Photosynthesis is defined by the following reaction,

$$CO_2 + H_2O \xrightarrow{\text{sunlight}} CH_2O + O_2 \tag{2}$$

where plants bind carbon from CO_2 to an organic form (CH_2O), using sunlight and H_2O, and releasing O_2. During decomposition, activity by microbial populations release CO_2

to the atmosphere, which is measured as soil respiration. Respiration is dependent on the degree of saturation of the peat, amount of labile material of the organic carbon and soil temperature. Valentine *et al* (1994) found that decomposition increased with increasing labile carbon and Hobbie (1994) found that soil and litter respiration increased with soil temperature. Moreover, the rate of decay of carbon is affected by the species composition (Johnson and Damman, 1991) likely due to the differences in C:N ratios between different species of *Sphagnum*. Plant and root respiration occur with the oxidation of CH_2O by the plant, with root respiration occurring in the soil substrate and plant respiration occurring in the leaves through the stomata. The total CO_2 flux or net ecosystem exchange (NEE), is measured as the difference between NEP and NER.

Daily NEE in peatlands has been found to range between -5 to 3 g CO_2-C m^{-2} d^{-1} (with a negative flux representing a peatland ecosystem uptake from the atmosphere), depending on the peatland type and conditions (Bellisario *et al.*, 1997; Ruimy *et al.*, 1995; Whiting, 1994; Gorham, 1991).

Peatlands are a large source of CH_4 to the atmosphere, releasing approximately 4.0 g C m^{-2} yr^{-1} globally (Gorham, 1991), with daily emission rates ranging from <1 to 135 mg C m^{-2} yr^{-1} in northern peatlands, depending on site characteristics and peatland type (Roulet *et al*, 1992; Hamilton *et al*, 1994; Roulet *et al*, 1994; Waddington and Roulet, 1996). Peatlands emit approximately 40% to 50% of the total CH_4 released to the atmosphere (Whiting and Chanton, 1993). The production of CH_4 through methanogenesis occurs under reduced conditions in peatlands. The line between the aerobic and anaerobic zones, determined by the top of the capillary fringe, is the border

between zones of predominantly CH_4 production and predominantly oxidation. However is should be noted that aerobic groundwaters can supply oxygen at depth. CH_4 production is dependent on the position of the water table, soil temperature and the labile carbon of the organic material in the methanogenic zone (Svensson and Sundh, 1993; Moore *et al*, 1994). There is a correlation between NEP and CH_4 production, with approximately 3% of the carbon fixed by peatlands (NEP), emitted back to the atmosphere as CH_4 (Whiting and Chanton, 1993). CH_4 is the second most abundant greenhouse gas, but is ~ 20 times more effective as a greenhouse gas than CO_2. Gorham (1991) estimates the global DOC export for peatlands at 20 g C m^{-2} yr^{-1}, however, Waddington and Roulet (1997, 1999) and Moore (1987a) suggest it can be much lower (1.1 to 6 g C m^{-2} yr^{-1}) in patterned boreal peatlands. Peatlands provide a large input of allochthonous material to downstream aquatic ecosystems. DOC is comprised of various organic compounds such as humic and fulvic acids. Whereas DOC concentrations in rainwater is usually <2 mg L^{-1} (Dalva and Moore, 1991), DOC in peatland soil and stream water can often exceed 50 mg L^{-1} (Dalva and Moore, 1991). Concentrations are usually much lower in stream water (2 to 16 mg L^{-1}) (Fiebig *et al*, 1991; Koprivnjak and Moore, 1992) than peat soil water (2 to 40 mg L^{-1}) (Moore, 1987a; Waddington and Roulet, 1997).

Although CO_2 does not represent the complete carbon balance, over 90% of the carbon exchanged in peatlands is in this form, and therefore is often used as a measure of contemporary peatland carbon storage (Gorham, 1995).

1.3 *Impact of peatland drainage and harvesting on hydrology and carbon cycling*

Harvesting peat has increased over the last century due to the suitability of peat as a cheap fuel source and for agriculture and horticulture (Armentano and Menges, 1986). Extensive drainage networks are installed to lower the water table, then one of the following three main techniques are used to harvest the peat: 1) *machine block cutting* which removes peat in strips resulting in a series of baulks and trenches; 2) *extrusion* which removes peat to a metre's depth; and 3) *surface milling* which removes peat by vacuuming which creates a relatively flat topography (Money, 1995). Bogs are the main type of peatland harvested because *Sphagnum* peat has the desired water holding capabilities demanded by horticulture and most peat mining in Canada is for this purpose (Price, pers. comm., 1999).

The combined effects of peatland drainage and harvesting lowers the position of the water table from just below the surface to greater than 60 cm below the surface (Mawby, 1995, Vompersky and Sirin, 1997). Due to the increased bulk density of the exposed catotelm, the specific yield of the peat is greatly reduced from 0.2 to 0.05 (Price, 1997). Drainage enhances lateral seepage of water and when coupled with a reduced specific yield in the upper peat profile, will increase any variation in water storage, resulting in a highly fluctuating water table (Boelter, 1964; Bragg, 1995).

Volumetric moisture contents in cutover peatlands are often higher than in *Sphagnum* dominated hummocks in natural bogs but less than natural *Sphagnum* lawns (Price, 1996). Hayward and Clymo (1982) found that *Sphagnum* moss cannot generate matric suction in soil with a soil tension more negative than -100 mb. Soil tension was

found to be greater (more negative) than -100 mb 82% of the time in a drained peatland, whereas it was between 0 and -100 mb 97% of the time in a mulch-covered, rewetted peatland (Price, 1997). Moreover, Price (1997) found that the greatest loss of moisture in a cutover peatland was in the upper 30 cm according to soil tension profiles, therefore inhibiting *Sphagnum* re-colonization as the soil retained moisture at a higher tension than the plant could generate.

This research indicates that rewetting techniques need to be implemented to increase soil moisture content by creating excess surface water before restoration operations can commence. Blocking drainage ditches and excavating reservoir ditches have been found to increase local water storage after precipitation and snow melt (Schouwenaars, 1993; Heathwaite, 1994; Bragg, 1995; Heathwaite, 1995; Mawby, 1995; Okruszko, 1995; Schouwenaars, 1995; LaRose *et al*, 1997), resulting in a higher and more stable water table. Volumetric moisture content was also found to increase from 55% in a control site, where ditches were blocked, to 62% in an experimental site with reservoir ditches (LaRose *et al*, 1997).

As mentioned earlier, during drainage operations the water table is lowered and stored carbon becomes aerated permitting soil microbial populations to decompose the substrate at a faster rate (Mulqueen, 1986; Lieffers, 1988; Malmer, 1992; Prevost *et al*, 1997). Vompersky and Sirin (1997) found the upper moisture limit in effectively drained peatlands is where air occupies at least 10% of the peat volume. Field and lab studies indicate an increase in CO_2 losses with a lower water table position (Billings *et al*, 1982; Gorham, 1991; Freeman et al, 1993; Moore and Dalva, 1993; Nykänen *et al*, 1995;

Silvola *et al*, 1996; Nykänen *et al*, 1997). CO_2 emissions increased in several drained peatlands with effective drainage by 100% to 400% due to the oxidation of the peat and the removal of carbon fixing vegetation through the harvesting of the acrotelm (Gorham, 1991; Nykänen *et al*, 1995; Silvola *et al*, 1996; Nykänen *et al*, 1997). For example, in an undisturbed Finnish peatland, the net release of CO_2 was 2.4 to 3.6 g CO_2 m^{-2} d^{-1}, which increased to 7.2 to 9.6 g CO_2 m^{-2} d^{-1} when the water table was lowered from 0-10 cm to 40-60 cm (Silvola, 1986). Gorham (1991) estimates that oxidation of peat due to long term drainage operations results in a net flux of CO_2 to the atmosphere of 8.5 Tg yr^{-1}, with the combustion of fuel peat adding an additional 26 Tg yr^{-1}. Drainage of peatlands and the combustion of fuel peat accounts for 35% of the long term rate of carbon fixation and 45% of the current rate (Gorham, 1991). This trend is expected to perpetuate the already rising concentrations of atmospheric CO_2 with respect to global climate change (Armentano and Menges, 1986).

The rate of oxidation has been shown to decrease with post drainage and harvest time as labile carbon decreases with time (Schothorst, 1977). Conversely, Prevost *et al* (1997) found that decomposition rates in the third year of post-drainage were higher than in preceding years. In lab experiments, Silvola and Ahlholm (1989) found CO_2 production was highest in well humified peat when at optimum soil moisture content and temperature. The temporal variation in rates of oxidation could be a result of changing peat properties (e.g. specific yield, bulk density, and porosity). The degree of decomposition determines the porosity and pore size distribution (Boelter, 1969; Hillman, 1997), which controls the water retention, hydraulic conductivity, and water yield of the

peat. A higher degree of decomposition lowers a peat soil's potential to retain a high moisture content, with volumetric soil moisture dependent on both the depth of the water table and the structure of the peat mass (Okruszko, 1995). Prevost *et al* (1997) and Price (1996) both found a decrease in volumetric moisture content in drained peatlands. Vompersky and Sirin (1997) found that the capillary properties of peat increased post-drainage due to the decrease in pore size. Decreased soil moisture content in the upper soil profile results in large diurnal fluctuations in peat temperature in drained peatlands, as the heat capacity in the upper soil profile is reduced with the loss of water. Soil temperature has been found to increase with drainage (Lieffers, 1988; Prevost *et al*, 1997; Hokka *et al* 1997). Oxidation should increase as the rate of decomposition increases due to the aeration of the peat and an increase in peat temperature.

Schothorst (1977) found that oxidation increased the bulk density and relative mineral content of peat to the extent that these variables can be used to calculate the total oxidation of a peatland. Oxidation of organic peat due to microbial activity has been found to be dependent on both site and climate characteristics. Bridgeham and Richardson (1992) and Nadelhoffer *et al* (1991) have both found that low labile carbon limits oxidation. Approximately 45% of the total oxidation over a 10 year period was found to occur within the first two years by Schothorst (1977) and over 50% of recorded subsidence occurred in the first year (Hillman, 1997). Moreover, a carbon accumulation model by Gilmer *et al* (1998) found that carbon emissions stabilize after 10 to 15 years post drainage, with the CO_2 efflux decreasing with a decrease in the concentration in labile carbon. This is contrasted by findings by Stewart and Wheatley (1990) who found

that the highest rates of decomposition in an eroding peat surface were from peat horizons with the most humified strata. While the loss of CO_2 through oxidation, resulting in increased atmospheric concentrations of carbon, have been studied in drained sites (Schothorst, 1977; Gorham, 1991; Billings *et al*, 1982; Nykänen *et al*, 1997), to date there are no studies on the effect of peatland harvesting on NEE and carbon storage. Moreover, the rate of peat oxidation when there is a large decrease in labile carbon with cutover operations, needs to be determined to study if there is a relationship between labile carbon and the rate of oxidation.

Drainage of peatlands also causes a reduction in CH_4 production to decrease as the water table depth lowers. CH_4 is oxidized in the aerobic zone (Ball *et al*, 1997) therefore the rate of CH_4 oxidation should increase as the methanotrophic zone increases in thickness. This results in a reduction in CH_4 emissions (Moore and Knowles, 1987; Lien *et al*, 1992; Freeman *et al*, 1993), with the potential for drained peatlands to become sinks for atmospheric CH_4 (Silvola, 1986). Martikainen *et al* (1992) found the emission of CH_4 declined by half in an ombrotrophic bog and became a net sink in a minerotrophic fen post-drainage. However, emissions from a drainage ditch in an upland boreal forest were found to be 126 times greater than in the surrounding forest, signifying an important source of CH_4.

Moore (1987b) found that there was no significant difference in concentrations of DOC from pristine and drained peatlands in Quebec. Concentrations of DOC ranged from 23 to 34 mg L^{-1} in pristine sites, rising to 55 mg L^{-1} during drainage and decreased to 35 to 43 mg L^{-1} post drainage (Moore, 1987b). However, Ferland and Rochefort

(1997) found a substantial change in the pore water chemistry of post-harvested sites, with increased concentrations in NH_4-N, NO_3-N, SO_4, P, Na, K, Mg, Cl, and Ca, indicating the potential for newly exposed peat surfaces to be more minerotrophic than the previous natural peatland (Wind-Mulder *et al*, 1996).

1.4 *Peatland restoration*

Restoration of a harvested peatland is comprised of three phases; 1) *re-wetting* - creating wet surface conditions; 2) *renaturation* - establishment of peatland vegetation and 3) *regeneration* - development of the self-regulating capacity of a peatland (Heathwaite, 1994).

As mentioned earlier, peatland drainage results in a lower water table leading to a thicker zone of aeration, while harvesting removes the productive acrotelm, exposing the catotelm, thereby creating an environment not conducive to natural re-vegetation. The newly exposed catotelm, which has a higher bulk density and lower hydraulic conductivity than the acrotelm, hinders the regrowth of vegetation, especially *Sphagnum* mosses, due to drier surface conditions and a high rate of decomposition. Ferland and Rochefort (1997) found that the peat surface dries too quickly even when the water table is close to surface for *Sphagnum* regrowth. The increase in moisture tension in the harvested peatland was the major factor hindering regrowth of *Sphagnum* because the water tension in the drained peat was greater than the moss could generate to hold water in its cells (Hayward and Clymo, 1982). *Sphagnum* cannot regulate their water supply

like vascular plants, and will desiccate under prolonged dry periods. This poses a problem for restoring the peatland to its previous function as a net carbon sink.

Restoration techniques of *Sphagnum* on harvested peatlands have been greatly improved through the works by Price (1997), Ferland and Rochefort (1997), Quinty and Rochefort (1997), Campeau and Rochefort (1996), and Lavoie and Rochefort (1996). They developed the implementation of rewetting and shelter techniques, such as ditch blocking, creating reservoir ditches, mulch covers and creating surficial microtopography. Once the sites have been rewetted through blocking drainage ditches, diaspores of *Sphagnum* moss may be seeded on the fields. To prevent desiccation throughout the growing season, shelter techniques are used to decrease water loss through evaporation from the upper peat profile. Quinty and Rochefort (1997) have shown that various coverings, especially straw mulch, increases the rate of survival of the *Sphagnum* moss, and the thickness of the *Sphagnum* moss layer. The use of fertilizers and specific plant parts (including capitula or capitula and stems) also increases the cover of *Sphagnum* moss. The return of the carbon fixing *Sphagnum* moss layer on cutover peatlands may offset the increased losses of CO_2 due to aerobic respiration, however there are no studies that have determined NEE in restored peatlands.

1.5 *Objectives*

Previous research indicates that peatlands can be altered from net atmospheric CO_2 sinks to sources by enhancing peat respiration through drainage and reducing NEP by removing carbon fixing vegetation and harvesting the acrotelm. Oxidation of peat,

due to increased microbial activity, contributes to increasing peat bulk density and decreasing specific yield and volumetric moisture content. Because the labile carbon of the peat decreases with time, the rate of decomposition should also decrease over time, thereby decreasing the rate of peat oxidation. This is supported by research that indicates most of the peat oxidation exists in the first few post harvest years (Schothorst, 1977). Regrowth of *Sphagnum* due to rewetting and shelter techniques should increase the rate of CO_2 fixation, although it might also result in increased CO_2 emissions, due to an increase in labile organic material. Nevertheless, there is a possibility to revert these ecosystems back to net atmospheric CO_2 sinks. The objective of this thesis, therefore, is to gain a better understanding of the processes controlling net ecosystem exchange (NEE) in natural, drained/cutover and restored peatlands. Specifically this thesis will determine:

1. The difference in net ecosystem CO_2 exchange between a natural peatland and a 2-year and 7-year post harvest peatland;

2. the difference in net ecosystem CO_2 exchange between a natural peatland, a restored and a naturally re-vegetated *Sphagnum* peatland; and

3. the biogeochemical processes controlling the temporal variability in net ecosystem CO_2 exchange at natural, cutover/drained, and restored peatlands.

CHAPTER 2

THE EFFECT OF DRAINAGE ON PEAT OXIDATION IN A CUTOVER PEATLAND.

2.0 INTRODUCTION

Peatlands play an important role in the global carbon cycle, sequestering approximately 455 Pg of carbon and represent one-third of the world's soil carbon. Peatlands have been found to emit CO_2 and CH_4 to the atmosphere (Crill *et al*, 1988; Klinger *et al*, 1994; Moore *et al*, 1998; Schiller and Hastie, 1996; Oberbauer *et al*, 1991; Waddington *et al*, 1996, Waddington and Roulet, 1996), exporting DOC (McKnight *et al*, 1985; Molot and Dillon, 1996; Urban *et al*, 1989; Waddington and Roulet, 1996) and sequestering atmospheric CO_2 (Bubier *et al*, 1998; Bellisario *et al*, 1998; Klinger *et al*, 1994; Whiting *et al*, 1991). Gorham (1991) estimates the long term storage of peatlands to be 23 g CO_2 m^{-2} yr^{-1}. Much research has concentrated on determining the role of peatlands in the global carbon cycle (Armentano and Menges, 1986; Crill, 1991; Gorham, 1991; Moore *et al*, 1998; Oberbauer, 1991; Waddington and Roulet, 1996). Much of this research has had a special emphasis on boreal and subarctic regions where the majority of peatlands reside, and to the potential effects of climate change on these ecosystems (Moore *et al*, 1998; Waddington *et al*, 1998). However, little attention has been paid to determining how drainage and cutover operations affect net ecosystem CO_2 exchange (NEE), and therefore carbon storage, in these peatlands.

Drainage and cutover operations lower the water table in peatlands and remove the upper peat layer, leaving an environment devoid of vegetation and difficult for *Sphagnum* species to re-colonize (Bugnon *et al*, 1997; Campeau and Rochefort, 1996; Ferland and Rochefort, 1996; LaRose *et al*, 1997; Quinty and Rochefort, 1997). Drainage increases the depth of the aerobic zone, increasing soil respiration by intensifying decomposition (Freeman *et al*, 1993; Nykänen *et al*, 1995; Nykänen *et al*, 1997; Silvola *et al*, 1996; Stewart and Wheatley, 1990). Net ecosystem production (NEP) is reduced to zero as any vegetation is removed from the surface through harvesting. Previous research indicates that peatlands can be altered from net atmospheric CO_2 sinks to sources by enhancing peat respiration through drainage and reducing NEP by removing carbon fixing vegetation by harvesting the acrotelm (Silvola, 1986; Moore and Knowles, 1987; Gorham, 1991; Nykänen *et al*, 1995; Silvola *et al*, 1996; Nykänen *et al*, 1997). While plant and root respiration is eliminated from NER and the labile carbon of the organic material is decreased, the lack of any NEP results in NEE changing from a sink or a weak source, to an increase as a source by 100 to 400% (Gorham, 1991; Moore and Dalva, 1993; Nykänen *et al*, 1995; Silvola *et al*, 1996;). To compound these problems, peat shrinkage and oxidation increase the bulk density of the peat substrate, increasing the magnitude of water table fluctuations (Boelter, 1964; Bragg, 1995; Price, 1997), and therefore increasing the potential for peat oxidation under dry conditions.

Schothorst (1977) has shown that oxidation can contribute up to 85% of total peatland subsidence, with the greatest losses occurring in the first three years, while Volk (1973) measured 58 to 73%, and Shih *et al* (1978) found it contributed about 70 to 90%.

Oxidation of peat, due to increased microbial activity, increases peat bulk density and decreases specific yield and volumetric moisture content (Boelter, 1965). Without the addition of labile carbon, the labile carbon of the peat decreases with time, thereby decreasing the rate of peat oxidation. Although shrinkage and compression, the other two components of peatland subsidence, can be offset by rewetting cutover sites, oxidation cannot be reversed. Therefore it is believed that it is important to restore peatland functions quickly in order to properly restore disturbed peatlands to their natural state.

The objectives of this paper, therefore, is to gain a better understanding of the processes controlling NEE in cutover peatlands. Specifically this paper will determine:

1. The difference in net ecosystem CO_2 exchange between a natural peatland and a two-year and seven-year post cutover peatland; and

2. the biogeochemical processes controlling the spatial and temporal variability in net ecosystem CO_2 exchange at natural and cutover peatlands.

2.1 MATERIALS AND METHODS

2.11 *Study area*

This study was conducted at the Sainte-Marguerite-Marie peatland in the Lac Saint Jean region in central Québec (48°47'N, 72°10'W) (Figure 2.1). The average annual temperature is 2.2 °C (Environment Canada, 1998). The mean annual precipitation is 908.5 mm, of which 65.7% falls as rain (Environment Canada, 1998).
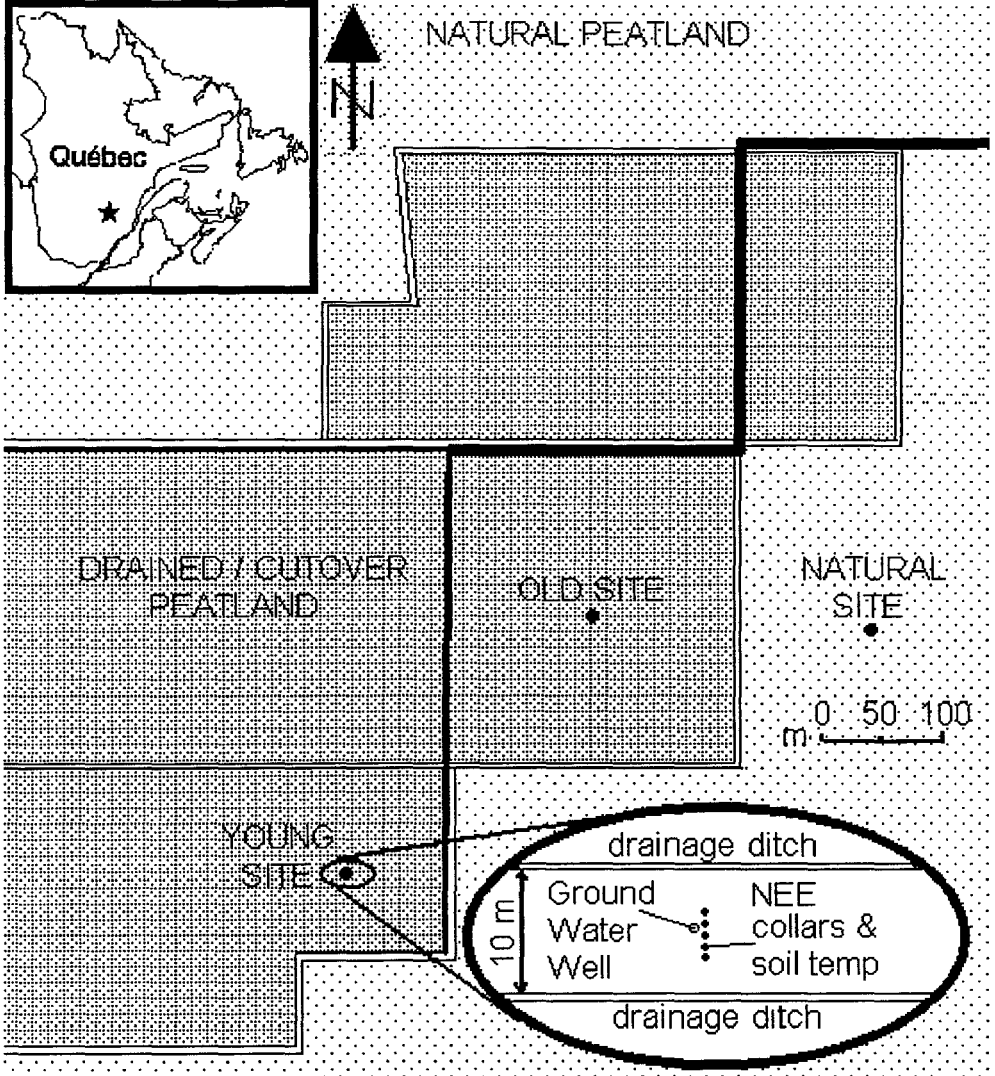
Figure 2.1: Location of Lac Sainte Jean (LSJ) research site in Quebec (delineated by star in top insert map), and location study sites in LSJ. Setup of NEE monitoring site illustrated in bottom insert map. Light stipled area refers to natural peatland while dark stipled area refers to drained/cutover peatland.

The Sainte-Marguerite-Marie peatland is situated on a terrace of deltaic sands (Price, 1997) in the Lac Saint Jean lowlands, forming a 4315-ha bog-poor fen complex which is classified as a plateau bog (National Wetlands Working Group, 1988). A portion of the Sainte-Marguerite-Marie peatland has been drained and the acrotelm harvested, with operations commencing in 1990. This study presents data from a natural portion of the bog and contrasts it with data from various cutover sections.

Net ecosystem CO_2 exchange was measured at three sites: a natural site (NATURAL), a two-year post cutover site (YOUNG), and a seven-year post harvested (OLD) between 5 May and 27 August of 1998. The NATURAL site was located east of the OLD site and has been shown to not be impacted by the surrounding cutover peatland (Price, pers. comm., 1998). The natural bog consisted of lawns and small hummocks, approximately 0.3 m above lawn. Both the hummocks and lawns were interspersed with *Sphagnum magellanicum,* and *S. angustifolium,* with *Polytrichum strictum, Ledum groenlandicum, Kalmaia angustifolia, and Larix laricina* present in the hummocks and *Vaccinium oxycoccos,* and *Andromeda glaucophylla* occurring in the lawns (S. Campeau, pers. comm.). The bog was fairly open with some sparse covering of *Picea mariana* and *Larix laricina.* The depth of peat to the mineral substrate in the NATURAL site was approximately 2.7 m. Due to the presence of an iron hardpan near the mineral substrate (Price, 1996), the bog is isolated from regional groundwater flow. The YOUNG site was drained in the fall of 1996 and cutover in 1997. The depth of peat had been reduced to approximately 1.8 m. The OLD site was drained in the fall of 1990 and cutover in 1991,

with the depth of peat approximately 1.70 m. Differences in peat thickness at each of the sites was due to peat extraction and subsidence.

2.12 *Methodology*

Measurements of net ecosystem CO_2 exchange (NEE) were made with a climate-controlled chamber and a PP systems EGM-1 infrared gas analyzer (IRGA) assembly placed and sealed over PVC collars set into the organic substrate. At the NATURAL site, collars were placed approximately 100 m from the edge of the cutover area to prevent the water table position from being affected by the drainage ditch. Collars at the YOUNG and OLD site were clustered approximately 2 to 5 m from the ditch egde where the water table position was flat. Net ecosystem CO_2 respiration (NER) was measured using an opaque chamber or a clear chamber covered with an opaque shroud. The chambers covered a surface area of 0.05 m^2 with a volume of 20 L. CO_2 concentrations were measured for a five minute duration at one minute increments. Fans inside each chamber ensured well mixed air during the sampling period. The clear chamber utilized a cooling device, consisting of a manual pump circulating cold water through a copper piping radiator inside the chamber, to ensure air temperature inside the chamber remained within 1°C of ambient air temperature. Instantaneous measurements of soil temperature and water table position were taken to establish an empirical relationship between these variables and NER. Net ecosystem CO_2 production (NEP) was measured using a clear chamber, with a series of neutral density shrouds to measure NEP under low light

conditions. A relationship between photosynthetically active radiation (PAR) and NEP was developed.

The following environmental variables were recorded every minute, then averaged for hourly measurements using Campbell Scientific CR10 data loggers. Peat temperature was measured continuously at various depths (0, 2, 5, 10, 25, 50, and 100 cm) at both the NATURAL and OLD sites. Volumetric soil moisture content was measured at both the NATURAL and OLD sites by the time domain reflectromitry method using Campbell Scientific water content reflectrometer probes at 5, 20 and 100 cm. Each probe was calibrated in peat soils taken from each study site. Photosynthetically active radiation (PAR) was monitored continuously at the NATURAL site only using a LI-COR quantum light sensor. Water table position was measured in a well using a water level recorder at both the NATURAL and OLD sites. Manual measurements of the following environmental variables were recorded at each of the three sites; soil temperature (air temperature and at depths of 2, 5, 10, 30, and 100 cm) and water table position were recorded concurrently with NER. Empirical relationships between environmental variables and NER were used to model continuous NER. Continuous soil temperatures and water table position at the YOUNG site were modeled using a correlation of instantaneous measurements at the YOUNG site and continuous measurements at the OLD site (5 cm soil temperature; r^2 =0.96, and water table; r^2 =0.98). Total rainfall was measured using a tipping bucket rain gauge 0.5 m above the surface at the OLD site.

Changes in bulk density and surficial (0 to 3 cm) soil moisture were determined gravimetrically, with weekly samples taken with a hand cutter which sampled the upper 3 cm of the peat surface. Dried peat samples were burned at 550°C for 30 minutes to determine the organic content by lost on ignition.

A time series model using continuous measurements of PAR, soil temperature (5 cm), water table position or volumetric soil moisture content at 5 cm depth was developed using the relationships of NEP with PAR and NER with soil temperature, water table position or soil moisture (at 5 cm depth). NEE was modeled using:

$$\text{NEE} = \frac{mT_{5cm}}{msT_{5cm}} \times \frac{\alpha * PAR * GP_{max}}{\alpha * PAR + GP_{max}} \times NER \tag{3}$$

where, the first term is a weighting term using a 7 day running mean of the daily peat temperature (mT_{5cm}) divided by the seasonal average (msT_{5cm}) (from Trumbore et al, 1999) to express a factor of seasonality on the measurements. In the second term, ecophysiological parameters (alpha (α) and maximum gross production (GP_{max})) were established from the coefficients of a best fit line of a quadratic curve (NEP vs PAR) using software created by Dundee Scientific, UK. NEP was modeled with a non-rectangular hyberbola using continuous PAR measurements. In the third term, multi-linear regression was used to determine a relationship between NER and soil temperature at 5 cm depth (T 5cm), and water table position (WT) or volumetric soil moisture content at 5 cm depth (θv 5cm).

$$NER = x(T\ 5cm) + y(WT) + z(\theta v\ 5cm) + constant \qquad (4)$$

Volumetric soil moisture content was found to have a stronger relationship with NER than water table position in the OLD site therefore was used in the model. Water table position was used in the other sites. Respiration was modeled using a time series multi-regression for each of the three sites. LAWN and HUMMOCK sites were modeled separately then weighted together (50% cover of each) to determine the total NER for the NATURAL site. The correlation between NER and soil temperature at 5 cm depth and water table position was much greater in the cutover sites than the NATURAL site.

2.2 RESULTS

2.21 Environmental Variables

The precipitation events were small and uniform throughout most of the study period with two dry periods occurring at the end of May and mid-August (Figure 2.2a). The precipitation was 78% of the normal thirty year mean for this period, indicating that the study season was drier than normal. Precipitation during May, July and August was low having 54%, 79% and 33% of the normal rainfall for these months, respectively. June precipitation exceeded the normal precipitation by 161% (Environment Canada, 1998). The two short drought periods resulted in lower water table positions at each of the sites.

The water table at the NATURAL site (Figure 2.2b) remained near the surface throughout the study period with small variations in depth and a mean seasonal water

table position of –11.8 cm. During this period the water tables in the YOUNG (Figure 2.2c) and OLD (Figure 2.2d) sites were significantly lower, with greater magnitude in fluctuations after rain events, and mean water table positions of –30.6 and –35.1 cm, respectively. The water table was only above the surface at the NATURAL site during the beginning of the study period with a maximum height of 1.5 cm above the surface, which occurred directly after snowmelt when the peat was completely saturated. The presence of a frost table in the cutover sites at the beginning of the study period prolonged saturation at these sites. The water table was never above the surface in the cutover portions of the bog, but had a minimum depth below the peat surface of –2.5 cm at the OLD site. The water table reached its greatest depth in mid August during the second drought period with maximum depth of –40.9 and –68.9 cm in the NATURAL and OLD sites, respectively. The water table remained above –10 cm in the NATURAL site 55% of the study season but only 3% of the time in the OLD site. Similarly, the water table was above -30 cm for 92% of the study season in the NATURAL site and only 35% in the OLD (Figure 2.3).

Air temperature was within 1.0°C of the 30 year mean for the most of the summer with the exception of May, where it was 3.3°C higher than normal (Environment Canada, 1993). The 5 cm depth soil temperature was significantly higher at the cutover sites than at the NATURAL site, with a mean 5 cm depth soil temperature of 14.1, 14.7, and 16.6 °C at the NATURAL, YOUNG, and OLD sites, respectively.
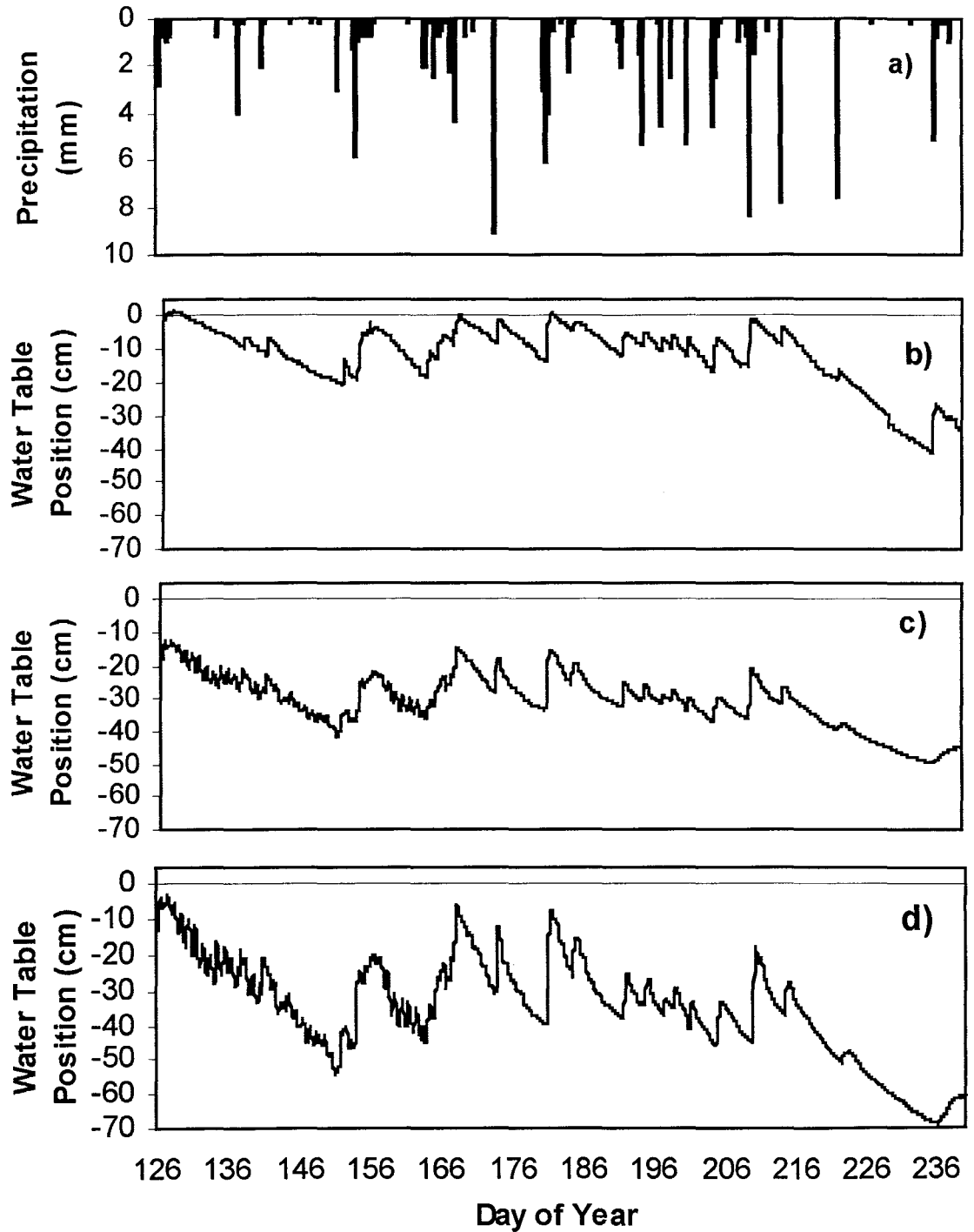
Figure 2.2: Precipitation (a) (mm) and water table positions (cm) at the NATURAL (b), YOUNG (c) and OLD (d) sites. Signal noise due to equipment error resulted in an exaggeration in water table fluctuations between julian day 126 and 166.
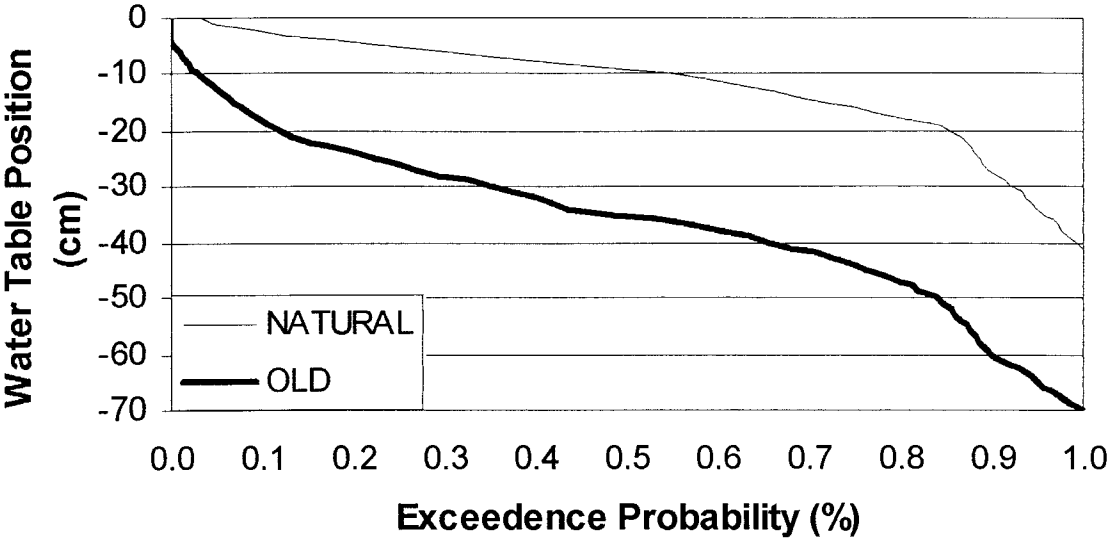
Figure 2.3: Frequency of water table positions at the NATURAL and OLD sites.

The mean daily 5 cm soil temperature in both the natural and cutover sites followed similar patterns to that of the air temperature, with the 5 cm soil temperature in the OLD site the closest to the air temperature in the magnitude of fluctuations (Figure 2.4). During the first half of the study period, the air temperature exceeded the soil temperature in the OLD site while the opposite was found during the latter half of the season. Soil temperature reached a maximum of 23.0 and 31.5 °C, and a minimum of 3.0 and 0.5 °C in the NATURAL and OLD sites, respectively. This is compared to air temperature which ranges from -2.8 °C to 31.3 °C over the study season.

Volumetric soil moisture content varied greatly throughout the peat profile in the cutover portion of the bog, with the lowest moisture content occurring near the surface (5 cm) (Figure 2.5b). This depth also showed the greatest range in soil moisture of 58.9 to 85.4 % with a mean of 70.0 %. There was less variation in soil moisture at 20 cm, with a range of 67.9 to 87.1% and a mean of 78.9%. The gradual decline in volumetric soil moisture at the 5 cm depth as the summer progresses is not seen at the 20 cm depth. As the water table position decreases, the response of volumetric soil moisture to rainfall events decreases in magnitude, especially at the 5 cm depth.

Bulk density ranged between 0.046 and 0.058 g cm^{-3} (mean of 0.052 g cm^{-3}) in the NATURAL site, 0.063 and 0.114 g cm^{-3} (mean of 0.084 g cm^{-3}) at the YOUNG site, 0.098 and 0.141 g cm^{-3} (mean of 0.119 g cm^{-3}) at the OLD site. Bulk density was consistently greater at the OLD site than the YOUNG site throughout the study period.
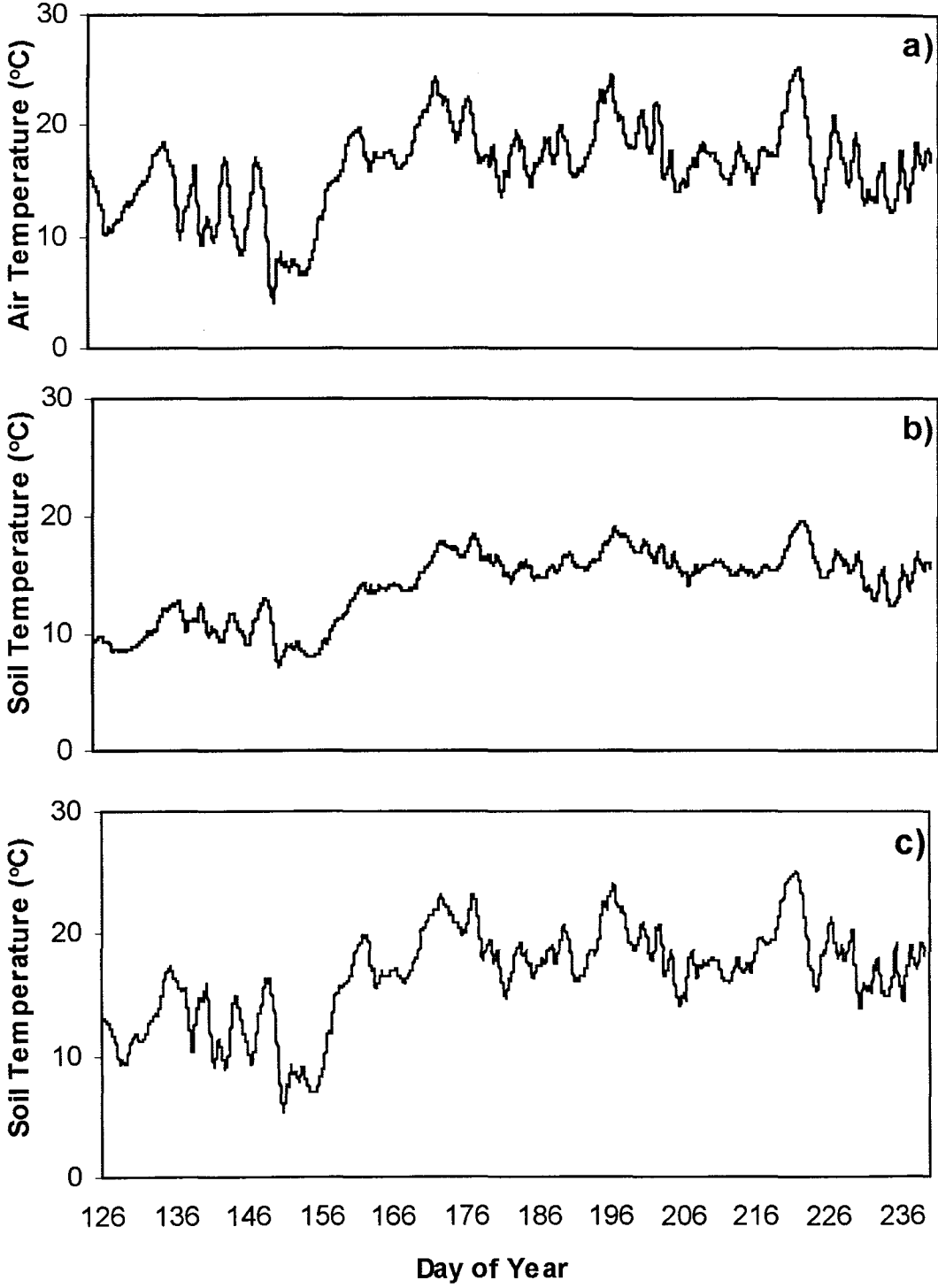
Figure 2.4: (a) Air temperature, and soil temperature (°C) at 5 cm depth at (b) NATURAL (b) and (c) OLD sites.
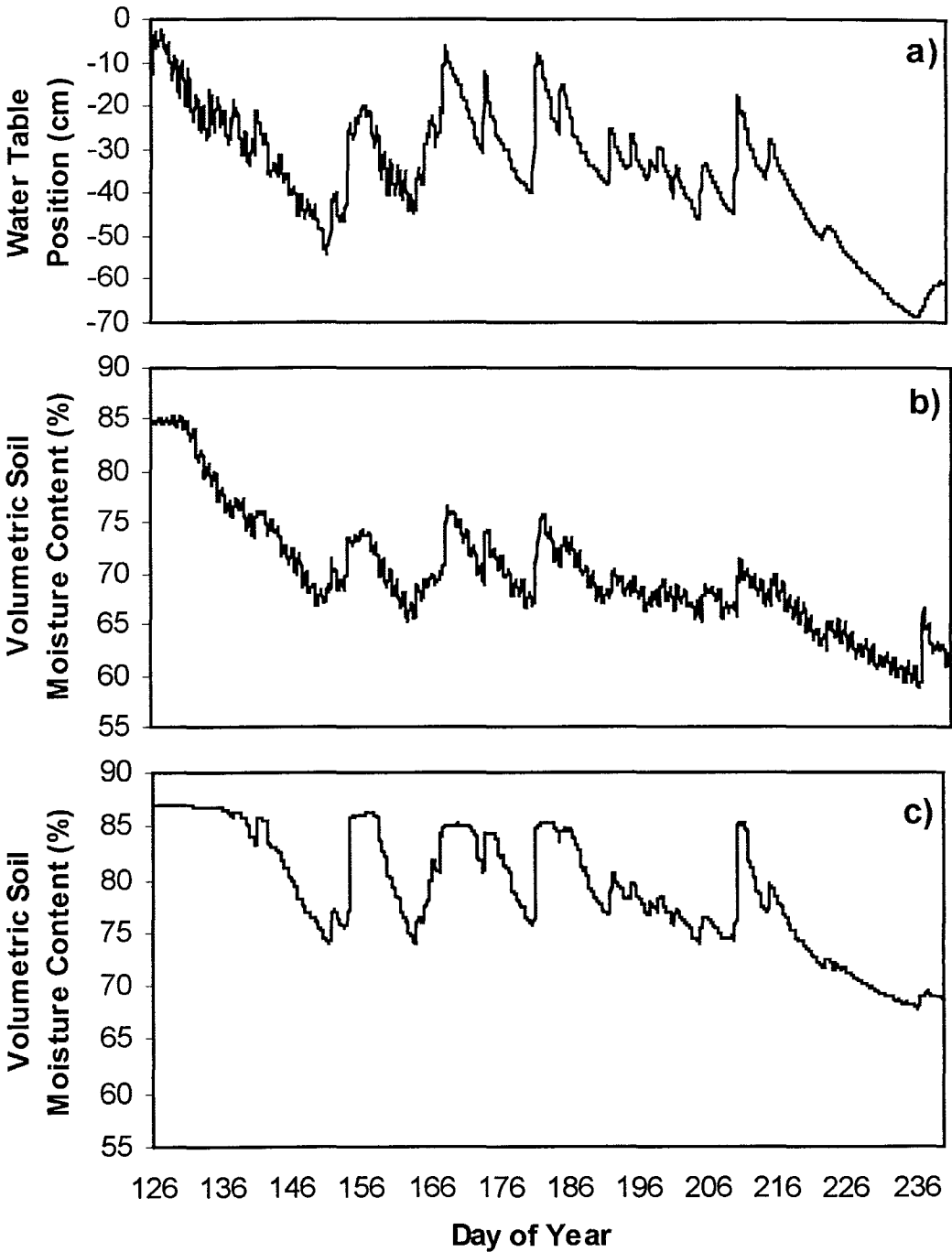
Figure 2.5: (a) Water table position (cm) and volumetric soil moisture content (%) at (b) 5 and (c) 20 cm at OLD site. Signal noise resulted in an exaggeration in water table fluctuation between julian days 126 and 166

Assuming a 50% carbon content in organic matter (Nykänen *et al*, 1995), the mean carbon content (0 to 2 cm) at the NATURAL, YOUNG and OLD sites was 44% ±0.01, 44%±0.06, and 38%±0.03, respectively. The carbon content at 5 cm in each of the cutover sites was 40%±0.05, and 39%±0.03, in YOUNG and OLD respectively.

2.22 *Modeled respiration*

Daily NER was significantly greater in the OLD site than in the YOUNG throughout the study season. Two of the collars at the YOUNG site exhibited a large increase in respiration as they dried out, resulting in an average NER significantly greater than that measured in the OLD site. Conversely at the YOUNG site, collars 1, 2, and 3 were significantly lower than those at the OLD site. The disparity in NER between the collars at the YOUNG site shows the spatial variability in NER which can occur in newly cutover sites.

Time series models were developed for net ecosystem respiration using multiple-linear regression of environmental variables (namely 5 cm depth soil temperature, water table position, and 5 cm volumetric soil moisture) with NER (Table 2.1).

Table 2.1: Regression coefficients and corresponding r^2 values from multiple-linear regression. Where NER = x(T 5cm) + y(WT) + z(θv 5cm) + constant. Mean daily values were used at all of the sites except at the LAWN site.

Site	n	r^2	x	y	z	Constant
LAWN	131	0.67	0.33	-0.23		-1.47
HUMMOCK	10	0.54	1.08	0.38		-8.52
YOUNG	12	0.79	0.60	-0.49		-12.04
OLD	10	0.78	0.80	-0.33		-11.20
OLD	10	0.83	0.58		-0.92	68.53

Mean daily modeled NER was greatest in the OLD site, followed by the YOUNG and NATURAL sites (12.8, 11.7, and 9.5 g CO_2 m^{-2}) (Figure 2.6). Each site was significantly different at the 95% confidence interval. NER was highly variable in the cutover sites where the greatest range in moisture conditions occurred. NER ranged between 0 and 26.6 g CO_2 m^{-2} d^{-1} and 0 and 28.0 g CO_2 m^{-2} d^{-1} at the YOUNG and OLD sites, respectively. There was much less variation in modeled NER throughout the study period at the natural site (2.2 to 17.3 g CO_2 m^{-2} d^{-1}).

2.23 *Relationships between environmental variables and net ecosystem respiration*

To illustrate the strong temporal variation in the hydrological controls on NER, mean daily NER at the OLD site was plotted versus time with water table position and volumetric soil moisture content at 5 cm depth (Figure 2.7). Volumetric soil moisture content was found to have a significantly stronger relationship with NER than water table position (r^2 = 0.83 compared with 0.79, for volumetric soil moisture content and water table position, respectively)(see Table 2.1). Price (1997) states that surface wetness is better evaluated by using volumetric soil moisture content and soil water tension than water table position. In the early part of the study season, NER increases as the upper peat layer dries. The wetting event that occurred between julian days (JD) 153 and 156 resulted in a decrease in NER from 23.0 to 1.7 g CO_2 m^{-2} d^{-1} as the water table moved from –43 to –19.9 cm and volumetric soil moisture increased from 69.2% to 74.2%. Within six days, NER increased to 17.2 g CO_2 m^{-2} d^{-1} as the peat dried again.
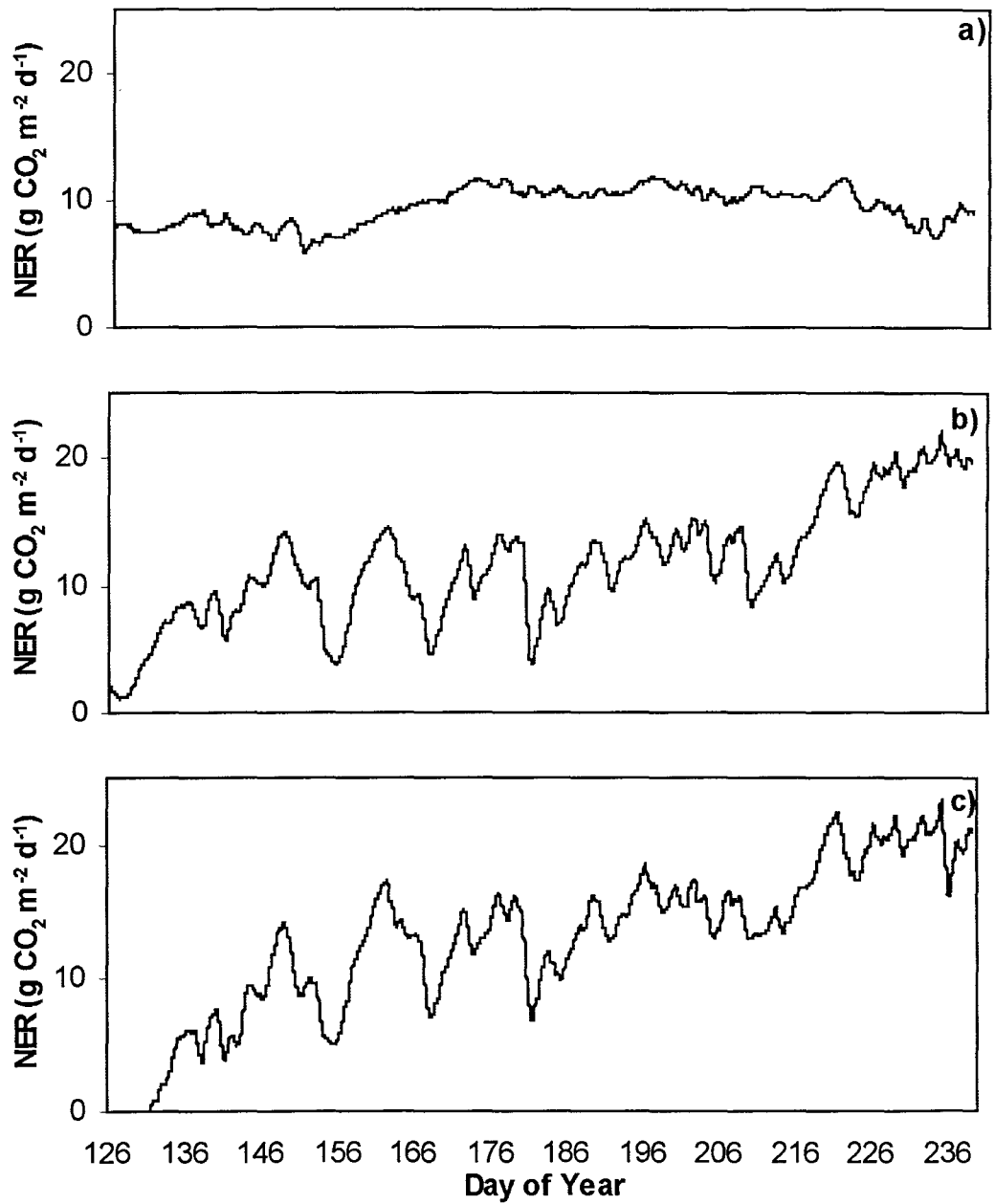
Figure 2.6: Modeled net ecosystem respiration (in g CO_2 m^{-2} d^{-1}) at the (a) NATURAL, (b) YOUNG, and (c) OLD sites
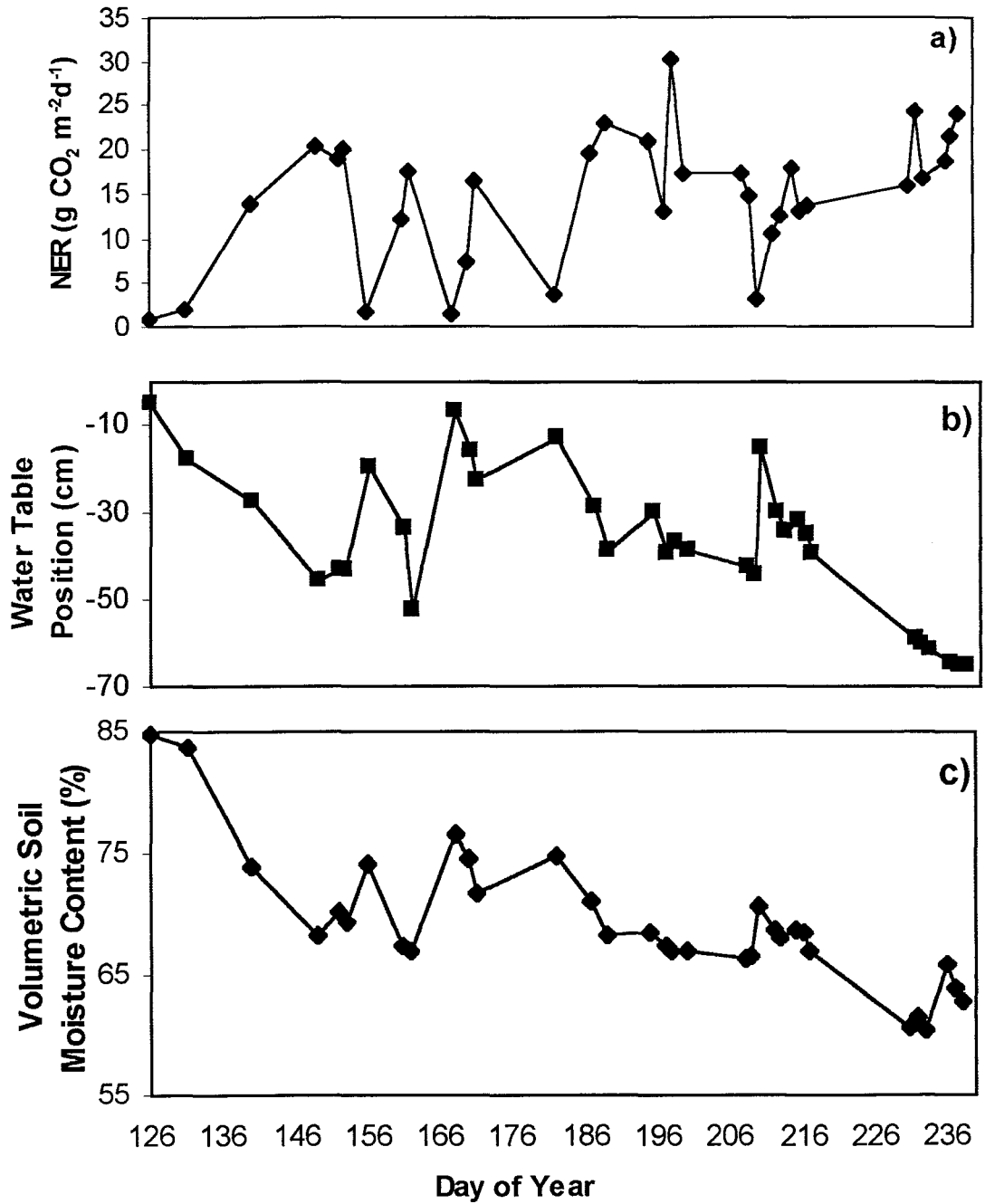
Figure 2.7: (a) Mean daily NER (g CO_2 m^{-2} d^{-1}), (b) mean daily water table position (cm), and (c) volumetric soil moisture content at 5 cm depth (%) at the OLD site.

The second drought period near the end of the study season was represented by a low water table position (-61.5 cm on JD 233), low volumetric soil moisture content (60.5%) and high NER (30.2 gCO_2 $m^{-2}d^{-1}$). A wetting event that occurred between JD 233 and 236 resulted in NER decreasing to 18.6 g CO_2 m^{-2} d^{-1}, when volumetric soil moisture content increased to 66.0%, although the water table decreased to −64.5 cm. The rain event was absorbed and retained as it infiltrated through the peat profile before reaching the water table, resulting in a change in soil moisture and NER although not water table position.

2.23 Seasonal carbon fluxes

The empirical relationship between PAR and NEE varied throughout the summer at the NATURAL site (Table 2.2) and between different topographic units (hummocks versus lawns) of vegetation. The collars in the natural site were subdivided into micro-topographic units of lawns and hummocks due to the presence of dwarf shrubs on the hummocks to determine the correlation between NEP and PAR. The correlation between NEP and PAR (r^2= 0.63 to 0.66) falls into the range found in other studies (Bubier *et al*, 1998; Frolking *et al*, 1998; Waddington and Roulet, 1996).

Table 2.2: Ecophysiological parameters and corresponding r^2 values from curve fitting relationships for PAR and NEE curves. See text for details on parameter descriptions. Standard deviation are in brackets.

Site	α	GP_{max}	r^2	n	Mean Daily NER
Lawn	0.008 (0.001)	8.2 (2.57)	0.66	132	-7.2
Hummock	0.064 (0.032)	18.1 (2.59)	0.63	82	-13.1
Natural	0.036 (0.017)	13.5 (2.58)	0.65	214	-9.5

Adding the modeled NER presented earlier, and the modeled NEP results for the NATURAL site, each site was a source of CO_2 to the atmosphere during the study period (Table 2.3). The OLD site represented the largest source of CO_2 to the atmosphere, followed by the YOUNG and NATURAL sites (397, 363, and 138 g C-CO_2 m^{-2}, respectively).

Table 2.3: Seasonal CO_2-C flux (g C m^{-2}) for NATURAL, YOUNG, and OLD sites. Negative numbers represent a gain of carbon to the peatland.

	NEP	NER	NEE
NATURAL	-164	301	138
YOUNG	0	363	363
OLD	0	399	399

Although the NATURAL site was a source of CO_2 to the atmosphere during the study period, CO_2 emissions still increased approximately 300% in the YOUNG and OLD sites

due to drainage and the removal of vegetation. This increase in CO_2 emissions from drained sites is similar to that reported in Silvola (1986) and Martikainen *et al* (1992).

2.3 DISCUSSION

2.31 *Environmental variables*

The mean water table position was 18.8 to 23.3 cm lower in cutover sites than the natural bog, as has been found in other studies (LaRose *et al*, 1997; Price, 1997). The OLD site had the lowest mean water table, indicating the effect of an increase in bulk density and decrease in specific yield with post-harvest time. Price (1996) suggests increased bulk density decreases the specific yield of the peat, thereby resulting in a greater magnitude of water table fluctuations. Because the YOUNG site is only two-years post-cutover, the bulk density is lower than the OLD site (bulk density = 0.084 g cm^{-3} and 0.119 g cm^{-3}, YOUNG and OLD, respectively). This results in the magnitude in water table fluctuations being less in the YOUNG site than the OLD. The lower water table position in these cutover sites resulted in a deeper aerobic zone than at the NATURAL site.

Diffusion of gases through the peat profile is inversely proportional to volumetric soil moisture content because diffusion of gases in water is approximately 10 000 times slower than through air. This is especially prevalent in cutover peatlands, where there is a wide range of volumetric soil moisture content throughout the summer, depending on the frequency and magnitude of precipitation events. Therefore, the wide range in volumetric soil moisture content at 5 cm depth was the main cause of the great variability

in NER measured in the cutover sites. Saturation still may be reached at a lower volumetric soil moisture values due to an increase in bulk density from peat subsidence (J.S. Price, pers. comm., 1999) later in the study season. Observed small diurnal variations in soil moisture at 5 cm depth were due to daily evaporation and nightly condensation onto the peat. Furthermore, due to a decrease in saturation, the total thermal capacity of the soil diminishes, resulting in more variable surface soil temperatures with a greater magnitude of diurnal fluctuations. An increase in soil temperature increases the potential for oxidation by increasing the rate of microbial activity (Moore and Knowles, 1987; Stewart and Wheatly, 1990; Hobbie, 1994).

2.32 *Net ecosystem respiration*

The increase in the depth of the aerobic zone and day-time soil temperature in the cutover sites led to an increase in NER in comparison to emissions from the NATURAL site, with NER in the OLD site greater than the YOUNG site. However, large wetting events would result in a sharp decrease in NER in the cutover sites. It was hypothesized by Schothorst (1977) and modeled by Gilmer *et al* (1998) that CO_2 emissions decrease with post harvest time as the concentration of labile carbon in the peat decreases. However, both observed and modeled results in this study show an opposite trend to this with the loss of CO_2 being marginally greater in the OLD site than the YOUNG site. This finding of increased decomposition occurring in a peat substrate with greater humification was also reported by Stewart and Wheatly (1990), Prevost *et al*, (1997), and

Silvola and Ahlholm, (1989). Considering that neither Schothorst (1977) nor Gilmer *et al* (1998) made direct measurements of oxidation, that could be the reason for this difference. Given that the carbon content of 0 to 2 cm layer was significantly greater in the young site (0.44% ±0.06%) than the old site (0.38% ±0.03%) and that the carbon content in the 5 cm layer were significantly the same between sites (0.39% ±0.03%), the increase in NER with humification in this study is not due to carbon content. Instead, differences may be caused by other factors. Foremost, the greater state of humification found in the OLD site likely resulted in nitrogen mineralization, with a higher availability of nitrogen found in the soil, acting as a fertilizer, thereby increasing the rate of peat oxidation. Ferland and Rochefort (1997) and Wind-Mulder *et al* (1996) found that there were increased concentrations in NH_4-N, NO_3-N and other nutrients in peat waters in post harvest peatlands, compared with natural bogs. Secondly, during the first half of the study season, the YOUNG site may have been wetter than the OLD due to the more poorly decomposed peat in the YOUNG site. Due to the larger pore size in the peat at the YOUNG site, evaporation would result in the upper 2 cm of peat drying at low suctions. The large pore size would result in less capillary rise, resulting in less upward flow of water. In the OLD site, the greater degree of humification would have resulted in an increase in the number of smaller pores, which improves capillary rise and therefore greater loss of water through evaporation. With shrinkage and oxidation occurring throughout the study season, the pore size would be decreased in the YOUNG site, improving capillary flow upwards over time, resulting in much drier conditions.

Spatial variability was higher at the YOUNG site, with collars YOUNG 4 and 5 found to be significantly more productive than YOUNG 1, 2, and 3. Although NER at the OLD site was significantly higher than at the YOUNG site, there was no significant difference between collars YOUNG 4 and 5 and the OLD site in the first half of the study season when peat conditions were wetter and significantly higher in the latter half of the season when the peat conditions were drier. Mean surficial soil moisture (0 to 2 cm depth) decreased by 2.5% in the OLD site (59.4 to 56.9%) between the first and second halves of the study season, and 11.5% in the YOUNG site (59.5 to 48%) Whereas NER in the OLD site increased significantly by 27% between the first and second halves of the study season, NER in YOUNG 1, 2 and 3, increased by 84% and YOUNG 4 and 5 by 95%, attributing to a greater increase in NER in drier conditions in the YOUNG site.

The higher NER measured at the OLD site suggests that the influence of both potential nitrogen fertilization and lower moisture values in the OLD site have a greater influence on NER than the poorer labile carbon of the peat.

2.33 *Seasonal carbon fluxes*

Due to the drier than normal conditions during the study period (observed rainfall was only 78% of the normal), the NATURAL site had a positive NEE throughout the summer. The NEE values were much higher than values reported in other CO_2 source peatlands (Silvola, 1986; Silvola *et al*, 1996; Waddington and Roulet, 1996; Bellisario *et al*, 1998). Regardless of the NATURAL site being a source of CO_2 to the atmosphere for

the study period, the total emissions of CO_2 from the cutover sites were approximately three times higher. This was due to a combination of increased NER and the reduction of the NEP to zero. Any loss of CO_2 through plant and root respiration which would have been removed through cutover operations would have been offset by an increase of soil respiration.

2.34 *Global implications*

Currently, peatland drainage in Canada does not contribute to a significant loss of peatlands. Only 0.01% of the total peatlands being drained in Canada for horticultural peat (Rebec and Thibault, 1998). Rebec and Thibault (1998) estimate Canadian peatlands cover $13.9x10^7$ ha, with only 16 000 ha used for the horticultural peat industry. Using Gorham's (1991) current estimate of carbon storage in peatlands of 23 g C m^{-2} yr^{-1}, this equates to a net sink of $3200x10^7$ kg CO_2-C yr^{-1} for Canada. Assuming a conservative annual oxidation rate of 4000 kg C ha^{-1} (emissions from the OLD site over 4 months) from cutover peatlands, there would be a net release of $6.4*10^7$ kg CO_2-C yr^{-1} to the atmosphere. This would result in a loss of carbon to storage ratio of approximately 0.2 %. Therefore, under current natural to drained/harvested peatland area ratios, drainage and harvesting of peat does not result in a significant loss in the net carbon sink in Canada. Using these same figures, only 5.5% of peatlands in Canada (or a specific region) need to be drained/harvested to exceed the natural carbon sink. Assuming 5.5% of peatlands in Canada were drained and harvested, the total natural peatland area would be $13.2x10^7$ ha, representing a carbon storage rate of $3000x10^7$ kg C. Carbon loss from

the drained peatland area (765×10^7 ha) using the oxidation rate of peat (4000 kg CO_2-C ha yr^{-1}) would equal 3100×10^7 kg C. Consequently the net sink function in Canada would be lost and changed to a net source of CO_2-C to the atmosphere if drained/cutover peatlands exceeded 5.5% of the total peatland area. Throughout some countries in Europe drainage of peatlands already exceeds 5.5% of the total peatlands area (Gorham, 1991). According to estimates by Gorham (1991), the Fennoscandia region exceeds this 5.5% drained/natural peatland ratio, with 31.4% of peatlands drained and other regions are approaching this; e.g. Russia 2.6%, USA 1.1%, and the global average is 3.3%. Considering that drainage of peatlands for fuel combustion, forestry and the horticultural peat industry has increased the annual carbon release seven fold since 1940 (Armentano and Menges, 1986), it is expected that the global loss ratio will only increase in the future.

2.4 CONCLUSIONS

NER in drained and cutover sites was found to be dependent more on the volumetric soil moisture content in the upper 5 cm of the peat profile, than changes in the thermal regime or the labile carbon of the peat. CO_2 emissions increased with post-harvest time regardless of a decrease in labile carbon. CO_2 emissions increased 270 to 300% over that of an adjacent natural peatland due to drainage and cutover operations. Directly measuring the net ecosystem CO_2 exchange in cutover peatlands, as opposed to determining loss of carbon from bulk density determinations, results in a better picture of how drainage and harvesting operations affect the carbon balance in peatlands.

The net carbon sink in Canadian peatlands is not presently in danger of becoming a net carbon source due to peatland drainage and harvesting activities. However, assuming the CO_2 evolution rates are representative globally, the global carbon sink is nearing the threshold of being changed from a net carbon sink to a net carbon source. Some regions of Canada (e.g., eastern Quebec, New Brunswick) where drainage of peatlands for horticulture is prevalent may already exceed this threshold. Results from this study suggest that if peat resources continue to be extracted then active restoration efforts are needed to restore the net carbon sink to these ecosystems. This is especially important in light of the implications of the recent Kyoto Protocol (UNFCCC, 1997).

CHAPTER 3

COMPARISON OF NET ECOSYSTEM CO_2 EXCHANGE BETWEEN PARTLY RESTORED AND NATURALLY RE-VEGETATED CUTOVER PEATLANDS

3.0 INTRODUCTION

Peatlands contain approximately one third of the world's soil carbon (Gorham 1991). Through drainage and harvesting operations, the net carbon sink has been reduced or eliminated, resulting in an increased net CO_2 output of between 0.009 Pg/yr (long term release rate) and 0.042 Pg/yr (short term release rate) to the atmosphere (Gorham, 1991). Consequently, drainage and harvesting of peatlands have the potential to significantly alter the global carbon balance.

Vegetation uses light energy to convert CO_2 into weaker reduced biochemical bonds in the form of carbohydrates (Schlesinger, 1997). Although the production of organic matter is lower in peatlands than in most terrestrial ecosystems (Frolking *et al*, 1998), slow decomposition rates result in net carbon sequestration in peatlands, leading to the formation and accumulation of peat. Drainage operations increase the depth of the aerobic zone, thereby increasing the rate of decomposition (Chapter 2). Consequently, the net ecosystem respiration (NER), which consists of microbial and plant respiration, is increased as the rate of decomposition is increased. Moreover, net ecosystem production (NEP), which is the net exchange of CO_2 less NER, can be reduced to zero as the carbon fixing vegetation is removed. This substantially changes the carbon balance of a peatland, increasing net ecosystem exchange (NEE) emissions by approximately 300%

(Chapter 2; Gorham, 1991). However, recent research (Rochefort *et al*, 1995; Campeau and Rochefort, 1996; Lavoie and Rochefort, 1996; Bugnon *et al*, 1997; Ferland and Rochefort, 1997; Quinty and Rochefort, 1997; Price *et al*, 1998; Price, 1996; Price, 1997; LaRose *et al*, 1997; Price and Schlotzhauer, 1999) indicates it is possible to partially restore abandoned peatlands. The goal of this study is to compare the NEE between a restored and natural peatland.

Prior to restoration, during peatland abandonment, the peat undergoes significant structural changes post-drainage. Water is removed from the peat, resulting in subsidence, which includes shrinkage and oxidation in the aerobic zone, and compression of the underlying peat below the water table as the over burdening peat layer loses its buoyancy and the weight of the drained peat compresses the underlying peat (Schothorst, 1977; Mulqueen, 1986). Furthermore, increased bulk density of the peat results in decreased specific yield, increasing the magnitude of water table fluctuations, thereby increasing the depth of the aerobic zone and the potential for increased oxidation. Moreover, Price (1997) has shown that increased soil tension in the upper surface of the peat layer is not conducive to re-colonization of *Sphagnum* species, which, if left unaided, will desiccate and die.

Natural re-vegetation of *Sphagnum* in abandoned cutover peatlands appears to be a function of the method of harvesting. Lavoie and Rochefort (1996) reports *Sphagnum* re-vegetation of an abandoned peatland in eastern Quebec which utilized the traditional block-cut method, although most of the vegetation cover were trees and dwarf shrubs with some cover of *Sphagnum* mosses in the trenches. At this site, the microtopography

created by the trench-baulk system provided shelter as well as a viable seed base for re-colonization (Campeau and Rochefort, 1997). In modern cutover sites this is not possible as the upper 40 to 60 cm of peat is cut and removed from the site, leaving a flat, barren surface with no seed source. Although blocked drainage ditches aid in rewetting the peatland, surface moisture tensions readily exceed −100 mb (Price, 1997), the threshold where *Sphagna* can no longer generate enough suction to hold water in their cells, causing desiccation of the plant (Hayward and Clymo, 1982).

Several studies have investigated various methods for restoring *Sphagnum* moss onto these harsh conditions both through the study of the growth of the *Sphagnum* moss using methods such as of shelter techniques (Rochefort *et al*, 1995; Campeau and Rochefort, 1996; Lavoie and Rochefort, 1996; Bugnon *et al*, 1997; Ferland and Rochefort, 1997; Quinty and Rochefort, 1997) and by determining what ecophysiological thresholds and hydrological and microclimatic restraints exist for *Sphagnum* (Price *et al*, 1998; Price, 1996; Price, 1997; LaRose, *et al*, 1997; Price and Schlotzhauer, 1999).

Based on limited productivity studies (Campeau and Rochefort, 1997) regrowth of *Sphagnum* due to rewetting techniques should increase the rate of CO_2 fixation in partly restore peatlands. CO_2 emissions, however, may also increase due to an increase in labile organic material. Nevertheless, there is a potential to restore these ecosystems back to net atmospheric CO_2 sinks.

Consequently, the specific objectives of this study are to assess the potential of restoring peatlands to a net carbon sink by determining the difference in net ecosystem

CO_2 exchange between a natural peatland, a partly restored and a naturally re-vegetated *Sphagnum* peatland.

3.1 MATERIALS AND METHODS

3.11 *Study area*

This study was conducted at two cutover peatlands, in the Sainte-Marguerite-Marie peatland in the Lac Saint Jean (LSJ) region (48°47'N, 72°10'W) (Chapter 2) and at the Cacauna-Station bog near Rivière-du-Loup (RDL) (47°53'N, 69°27'W), located in central and eastern Québec, respectively. The average annual temperature is 2.2 °C at LSJ and 3.3 °C at RDL (Environment Canada, 1993). Mean annual precipitation is 908.5 and 924 mm at LSJ and RDL, respectively, with 66% falling as rain at LSJ and 73% in RDL (Environment Canada, 1993).

The Sainte-Marguerite-Marie peatland is situated on a terrace of deltaic sands in the Lac Sainte Jean lowlands (Price, 1997), forming a 4315-ha bog-poor fen complex which is classified as a plateau bog (National Wetlands Working Group, 1988). The Cacauna-Station Bog is in the maritime floristic region, originally covering 175 ha with a thick peat mass (up to 4 m deep) underlain by a thick deposit of Champlain Sea clay (Van Seters, 1999). This study examined a net ecosystem exchange (NEE) in hummocks and lawns in a natural bog which was adjacent to sites in cutover and restored sections at LSJ, as well as a naturally re-vegetated abandoned trench site at RDL.

A portion of the Sainte-Marguerite-Marie peatland was drained and the acrotelm harvested utilizing modern cutover techniques with heavy machinery, with operations

commencing in 1990. Traditional block-cut methods were used in RDL starting in 1942 until 1975, leaving a series of peat baulks and trenches (Van Seters, 1999). Drainage ditches were blocked manually or through peat slumping in both locations to rewet the sites. While there is no natural re-colonization of vegetation in the cutover sites at LSJ, various tree and shrub species and cotton grass have invaded the abandoned sites at RDL, with *Sphagnum* moss cover being sparse, occurring mostly in the wet trenches (Lavoie and Rochefort, 1996, Van Seters, 1999).

Net ecosystem CO_2 exchange was measured at 16 locations in three sites at the LSJ study area between 5 May and 27 August of 1998. The sites were located at a natural site (NATURAL), a restored trench (REST), and an adjacent non-restored patch adjacent to the REST site (REST-Bare). The natural bog consisted of lawns and small hummocks, approximately 0.3 m above lawn. Both the hummocks and lawns were interspersed with *Sphagnum magellanicum,* and *S. angustifolium,* with *Polytrichum strictum, Ledum groenlandicum, Kalmaia angustifolia, and Larix laricina* present in the hummocks and *Vaccinium oxycoccos,* and *Andromeda glaucophylla* occurring in the lawns (S. Campeau, pers. comm). The natural bog was fairly open with some sparse covering of *Picea mariana* and *Larix laricina*. The depth of peat was approximately 2.7 m. The REST site had been drained in the fall of 1990 and cutover in 1991 with a peat depth of approximately 1.7 m. A section of the cutover peatland was actively restored with *Sphagnum fuscum,* and some *S. capillifolim* in trenches in the REST site in 1997 (Campeau and Rochefort, 1997).

In the Cacauna-Station bog, three sites were monitored for NEE and environmental variables in both re-vegetated hummocks (RVEG) and the adjacent bare patches (RVEG-Bare). The hummocks were composed of mostly *Sphagnum fallax, S. capillfolium, S. magellanicum,* and some *S. fuscum* (Lavoie and Rochefort, 1996). *Ericaceous* shrub covering was removed to simulate similar conditions with those at the partly restored REST site at LSJ.

3.12 *Methodology*

Measurements of net ecosystem CO_2 exchange (NEE) were made with a climate-controlled enclosure and a PP systems EGM-1 infrared gas analyzer (IRGA) assembly placed and sealed over PVC collars set into the organic substrate. Net ecosystem CO_2 respiration (NER) was measured using an opaque chamber or a clear chamber covered with an opaque shroud. Instantaneous measurements of soil temperature and water table position were taken to establish a relationship between these and NER. Net ecosystem CO_2 production (NEP) was measured using a clear chamber and neutral density shrouds were used to determine a relationship between PAR and NEP. The chambers cover a surface area of 0.05 m^2 with a volume of 20 L. CO_2 concentrations were monitored for a five minute duration at one minute increments. Fans inside each chamber ensured a circulation and mixing of air during the sampling period with the clear chamber utilizing a cooling device and manual pump circulating cold water through a copper piping radiator inside the chamber to ensure air temperature inside the chamber remained similar to ambient.

The following environmental variables were recorded every minute, then averaged for hourly measurements using Campbell Scientific CR10 data loggers. Peat temperature was measured continuously at various depths (0, 2, 5, 10, 25, 50, and 100 cm) at both the NATURAL and cutover sites. PAR was monitored continuously at the NATURAL site only using a Li-Cor quantum light sensor. Water table position was measured in a well using a water level recorder at both the NATURAL and cutover sites. Precipitation was measured using a tipping bucket rain gauge 0.5 m above the surface at the cutover site.

A time series model using continuous measurements of PAR, soil temperature (5 cm), water table position or volumetric soil moisture (5 cm) was developed using empirical relationships of NEP with PAR and NER with soil temperature, water table position or soil moisture (at 5 cm depth). NEE was modeled using:

$$\text{NEE} = \frac{mT_{5cm}}{msT_{5cm}} \times \frac{\alpha * PAR * GP_{\max}}{\alpha * PAR + GP_{\max}} \times NER \tag{5}$$

where, the first term is a weighting term using a 7 day running mean of the daily peat temperature (mT_{5cm}) divided by the seasonal average (msT_{5cm}) (from Trumbore *et al*, 1999) to express a factor of seasonality on the measurements. In the second term, ecophysiological parameters (alpha (α) and maximum gross production ($GP_{\max}$)) were established from the coefficients of a best fit line of a quadratic curve (NEP vs PAR) using software created by Dundee Scientific, UK. NEP was modeled with a non-rectangular hyperbola using continuous PAR measurements. In the third term, multi-

linear regression was used to determine a relationship between NER and soil temperature (5 cm), and water table position (WT); (NER = x(T 5cm) + y(WT) + constant).

3.2 RESULTS

3.21 *Environmental variables*

The total precipitation of 286.5 mm during the study period was approximately 78% of the 30 year mean (Chapter 2). Although there was less total rainfall throughout the study period, most of the rain events were small and uniform.

The mean water table position was −11.8 cm in the NATURAL site, -35.8 cm in the REST and REST-Bare sites, and -26 cm in the RVEG (based on point measurements only) (Figure 3.1). The water table remained near the surface in the NATURAL area except for during the second drought period in mid August, when the water table reached its lowest point of the study season (-40.9 cm).

Mean 5 cm peat temperature was 14.1°C in the natural and REST sites, 15.0°C in the REST-Bare site, and 18.4°C in the RVEG (based on point measurements in RVEG) (Figure 3.2). Peat temperature was the highest in the REST-Bare site throughout the study season with the REST exceeding the NATURAL site in the first half of the season. Peat temperature ranged between 7.1 to 19.7°C, 6.7 to 19.6°C, and 7.6 to 20.6°C in the NATURAL, REST and REST-Bare sites, respectively, with the coldest temperatures occurring in mid June.
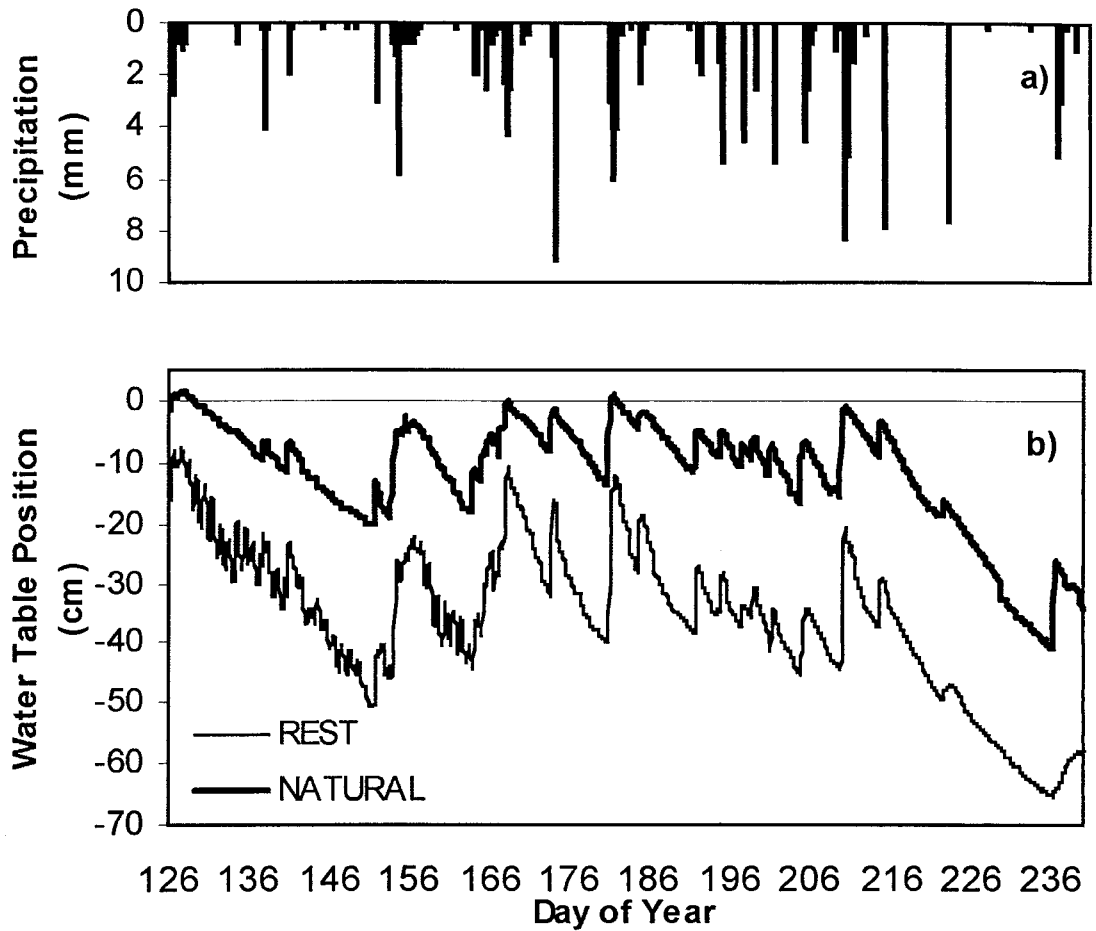
Figure 3.1: (a) Precipitation (mm), and (b) water table position (cm) (b) at LSJ for the NATURAL and REST sites. Signal noise due to equipment error resulted in an exaggeration in water table fluctuation at the REST site between julian days 126 and 166.
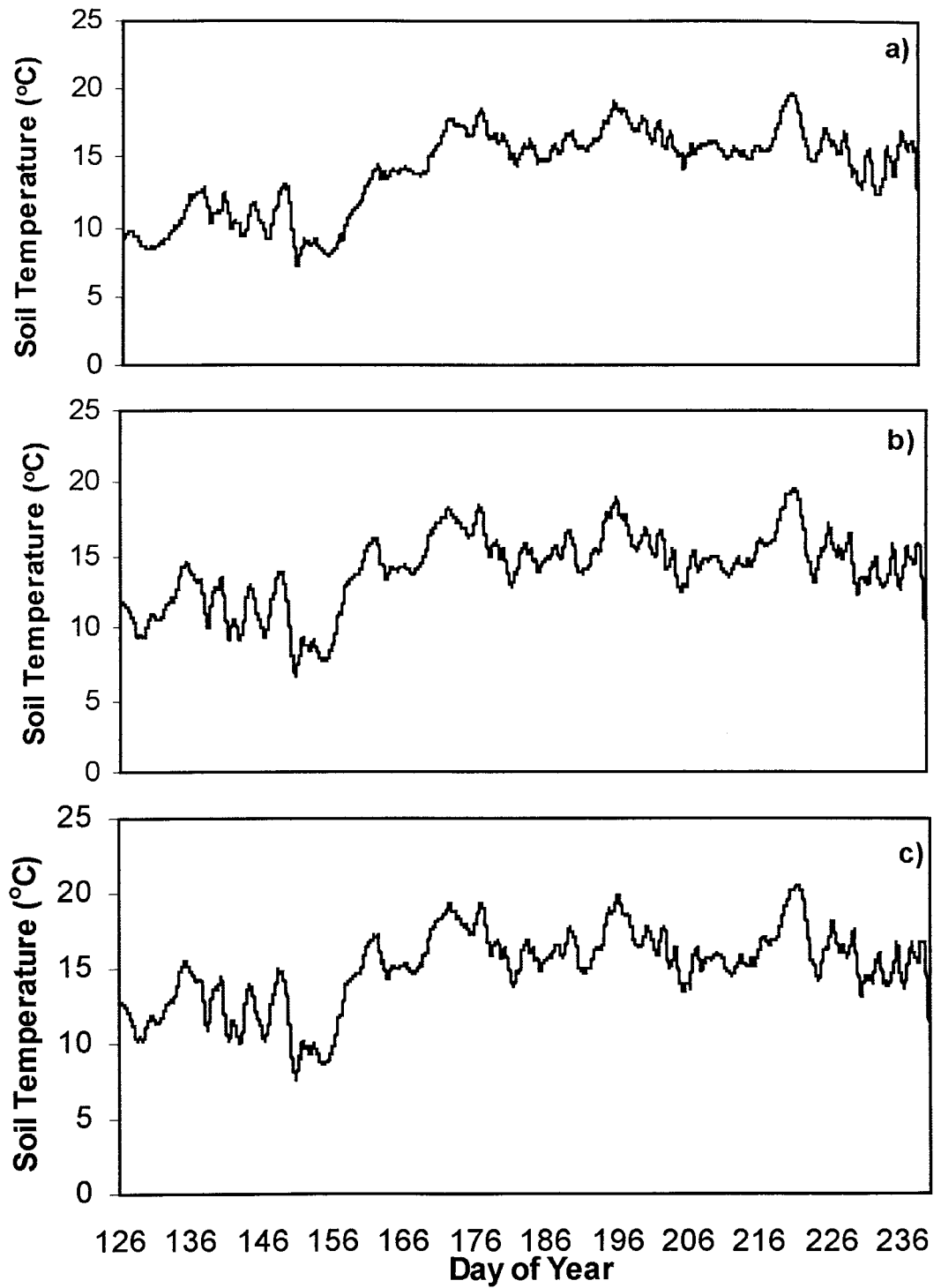
Figure 3.2: 5 cm soil temperature (°C) at the (a) NATURAL, (b) REST and (c) REST-Bare sites.

3.22 *Net ecosystem respiration*

Although the relationship between net ecosystem respiration (NER) and water table was weak, there was a discernible pattern of decreasing NER with wetting events and a gradual increase in NER during drying periods (Figure 3.3a and 3.3b). Between julian days (JD) 148 and 157, the water table rose from −39.5 to −23 cm resulting in NER decreasing from 22.0 to 7.2 g CO_2 m^{-2} d^{-1} in the REST site. In the REST-Bare site, the water table rose from −40 cm to −12 cm between JD 162 to 168 causing NER to decrease from 12.2 to 2.7 g CO_2 m^{-2} d^{-1}. Following wetting events, NER was lower in the REST-Bare site than the REST site. Variables for modeling NER using multi-linear regression are given in Table 3.1

Table 3.1: Variables and corresponding r^2 values from multi-linear regression. NER = x(T 5cm) + y(WL) + constant. Mean daily NER was used for regression except in the LAWN site, which had a stronger relationship with observed data.

Site	n	r^2	x	y	Constant
LAWN	131	0.67	0.33	-0.23	-1.47
HUMMOCK	10	0.54	1.08	0.38	-8.52
REST	17	0.32	0.38	-0.15	-0.13
REST-Bare	19	0.52	0.94	-0.14	-9.65

Modeled NER was variable both spatially between sites and temporally (Figure 3.4). Larger changes in volumetric soil moisture content and soil temperatures resulted in variations in NER at the end of the season when the water table position did not change.
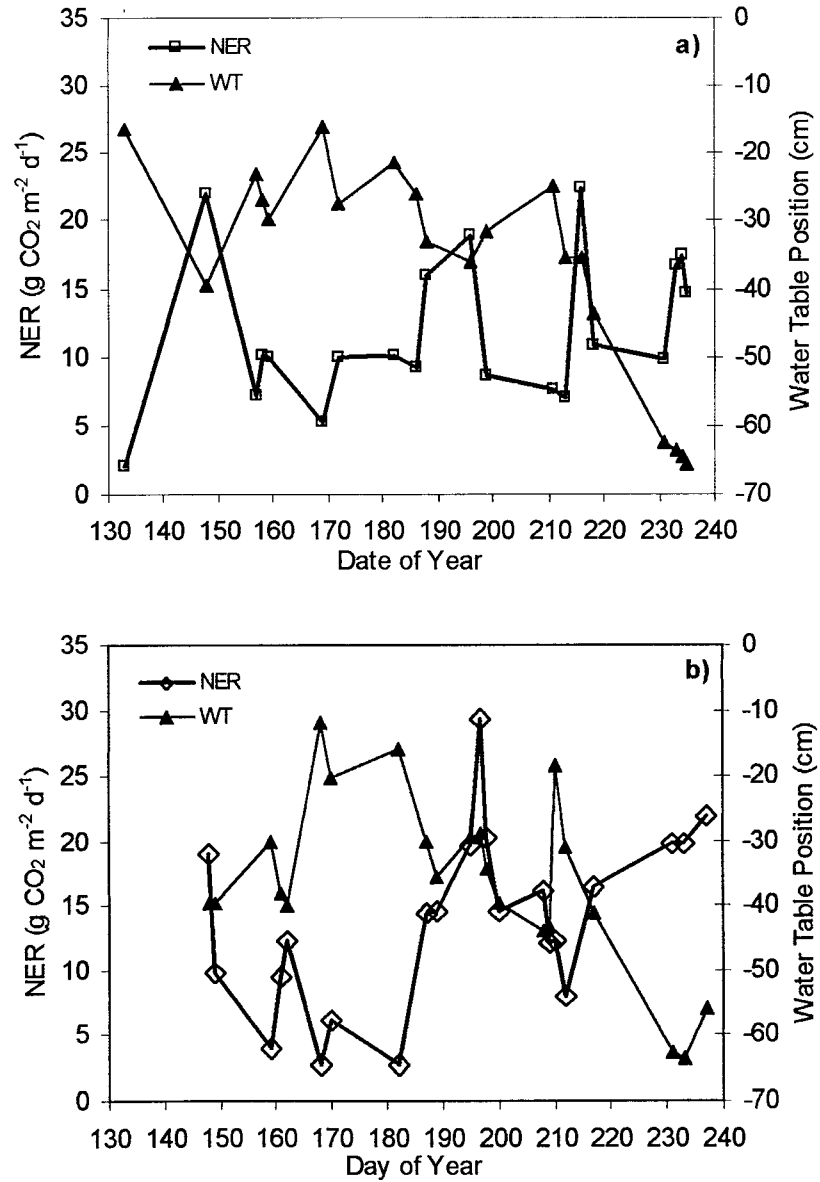
Figure 3.3: Mean daily net ecosystem respiration (g CO_2 m^{-2} d^{-1}) and water table position (cm) in (a) REST site, and (b) REST-Bare site.

Mean NER was the greatest in the HUMMOCKS (12.4 g CO_2 m^{-2} d^{-1}), followed by REST (10.5 g CO_2 m^{-2} d^{-1}), REST-Bare (10.4 g CO_2 m^{-2} d^{-1}), and LAWN (6.9 g CO_2 m^{-2} d^{-1}).

Mean daily NER was consistently higher in the HUMMOCKS than the LAWNS, with the greatest NER occurring during mid-summer in the HUMMOCKS, with the exception of during the low water table periods in late may and mid-august. NER ranged from 1.5 and 2.3 g CO_2 m^{-2} d^{-1} to 19.1 and 13.5 g CO_2 m^{-2} d^{-1} in the HUMMOCKS and LAWNS, respectively. NER was much more variable in the cutover sites than the NATURAL sites, ranging from 5.4 and 1.3 to 15.6 and 23.1 g CO_2 m^{-2} d^{-1} in the REST and REST-Bare sites, respectively. The magnitude of the fluctuations in the REST-Bare site was much greater than in the REST site, with NER increasing as the summer progressed.

Seasonal NER was the greatest in the NATURAL hummocks with an average of 1.4 kg CO_2 m^{-2}. This was followed by the REST and REST-Bare sites with 1.2 and 1.1 kg CO_2 m^{-2}, respectively. NER from the lawns was the lowest from all of the sites in LSJ with 0.8 kg CO_2 m^{-2}.

The similarity in NER values at the REST and REST-Bare sites was not found in the RVEG and RVEG-Bare sites. Despite equal water table positions and higher temperatures at the RVEG site, NER at the RVEG-Bare site was approximately double that of the RVEG site for each measurement period (Table 3.2).
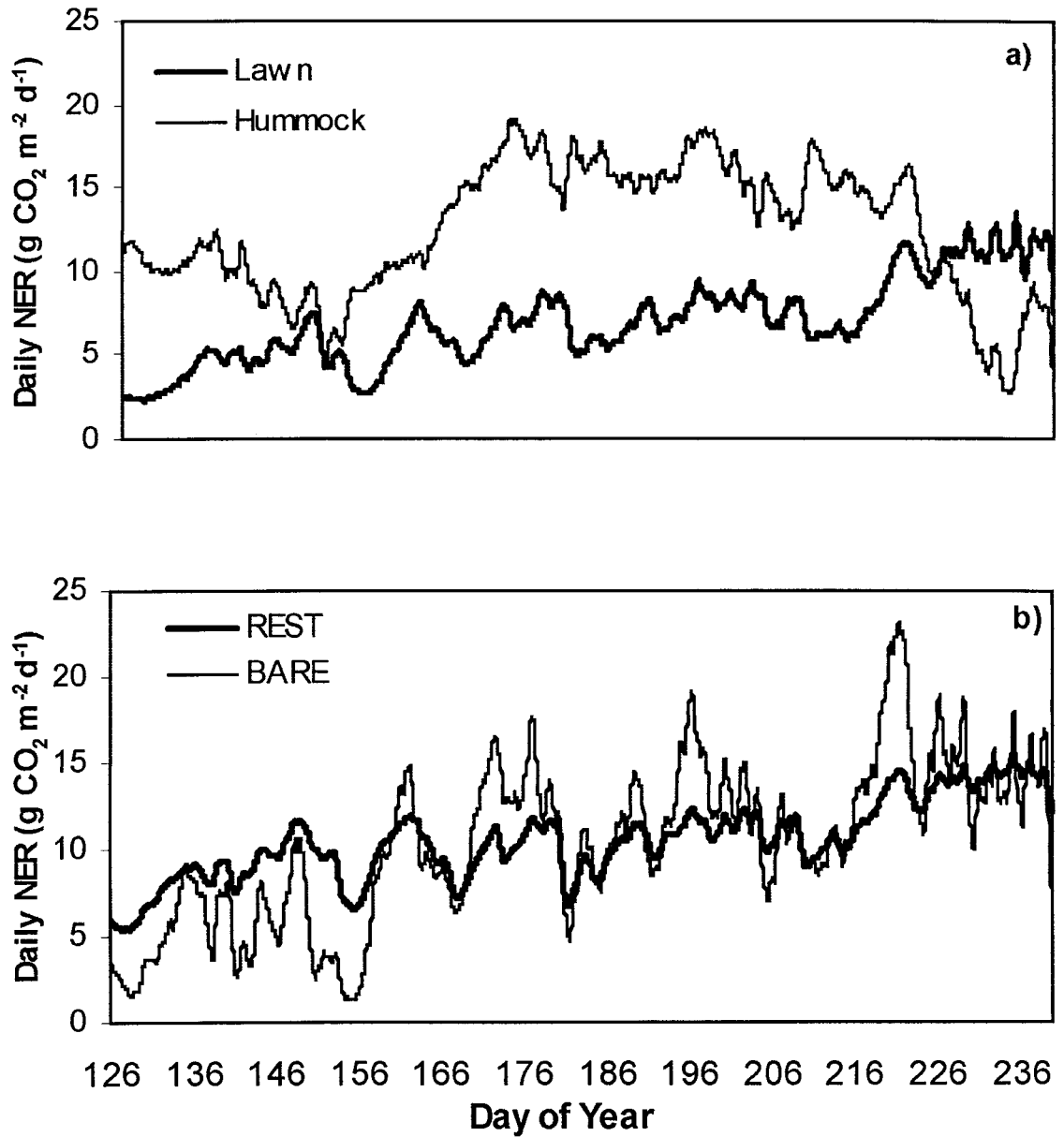
Figure 3.4: Modeled net ecosystem respiration in the (a) NATURAL (HUMMOCK and LAWN) and (b) Restored/cutover sites (REST and REST-Bare). Units are in g CO_2 m^{-2} d^{-1}. See text for a description of the modeling procedure.

Table 3.2: Environmental variables and mean NER (g CO_2 m^{-2} d^{-1}) for RVEG and RVEG-Bare sites at the RDL peatland.

Period	Soil Temp (at 5 cm in °C)		Water Table (cm)		NER (g CO_2 m^{-2} d^{-1})	
	RVEG	Bare	RVEG	Bare	RVEG	Bare
Early 174-177	18.1	17.1	-19	-19	5.29	10.43
Mid 204-207	18.6	16.3	-20	-20	7.13	12.86
Late 224-227	N/A	N/A	-39	-39	8.07	15.23

Volumetric soil moisture content (at 5 cm) was much greater under the *Sphagnum* hummocks (87%) than the adjacent bare areas (0.75) (G. Whitehead, pers. comm). A higher moisture content results in both lower soil respiration due to slower rate of gas diffusion through the peat (Chapter 2).

3.23 Net ecosystem production

To compare NEP between the sites, analysis of the ecophysiological parameters created from NEP-PAR response curves was undertaken (Table 3.3).

Table 3.3: Nonlinear NEP-PAR Rectangular Hyperbola Curve Fit Parameters with standard deviations given in brackets, α and GP_{max} are in g CO_2 m^{-2} d^{-1}.

Site	α	GP_{max}	r^2	n	NER
Lawn	0.01 (0.001)	8.2 (2.6)	0.66	132	-7.2
Hummock	0.06 (0.032)	18.1 (2.6)	0.63	82	-13.1
REST	0.02 (0.006)	17.5 (2.9)	0.60	180	-10.5
RVEG	0.01 (0.005)	6.6 (1.7)	0.54	118	-6.8

The alpha coefficient (α) is the initial slope of the light response curve which indicates how a plant responds to increasing light levels. The α values were low in all of the sites, (lawns, hummocks, REST and RVEG sites; 0.008, 0.064, 0.023, and 0.011 respectively) when compared with those found by Waddington and Roulet (1996) but are similar to alpha coefficients found by Frolking *et at* (1998) and *Bubier et* al (1998). The α coefficient was the greatest in the hummocks indicating that they had the greatest uptake of CO_2 with increasing PAR. In comparison for the sites with only *Sphagnum* cover, the α was highest at the REST site, which was more than two times greater than RVEG and LAWN sites.

The point of light saturation or maximum gross photosynthesis (GP_{max}) was greatest in the HUMMOCKS, followed by the REST, LAWN, and RVEG sites (19.9, 18.0, 8.3, and 6.5 g CO_2 m^{-2} d^{-1}, respectively). Again, the values of the HUMMOCKS fall within the range of those found in other studies (Frolking *et at,* 1998; Bubier *et* al, 1998), but the LAWNS are lower than those found by Waddington and Roulet (1996). GP_{max} at the REST site was more than double that from the natural LAWN. Cumulatively, the REST site fixed more carbon throughout the study period than the LAWN, but less than the HUMMOCKS (505, 199, and 1003 g CO_2 m^{-2} d^{-1}, respectively) (Table 3.4).

Although the slopes of the relationships do vary between sites, there is a positive relationship between NEP and NER at the four sites indicating that an increase in labile carbon from recently fixed CO_2 leads to increased decomposition (Figure 3.5).
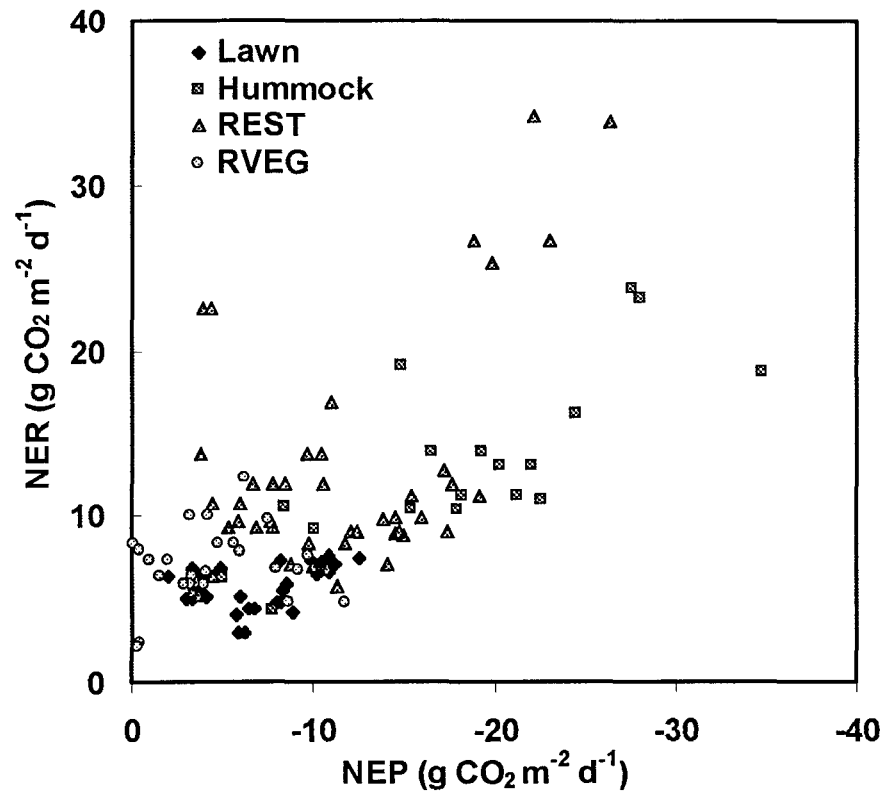
Figure 3.6: Net ecosystem respiration plotted against net ecosystem production for *Sphagnum* covered sites. Units are in g CO_2 m^{-2} d^{-1}.

The REST site has the greatest slope and the slope of the line for the HUMMOCKS site is similar to the REST site. The REST site is in a more aerobic environment than the HUMMOCKS with both sites having great potential for decomposition of labile carbon inputs. Both of these sites were very productive at sequestering atmospheric carbon, with high NEP values providing an excellent source of labile carbon for decomposition. The slopes of the lines for the RVEG and LAWNS are very different from the HUMMOCKS and REST, perhaps due to the low NEP values in these sites. The potential for decomposition is not as great in these sites as they are wetter environments than the REST and HUMMOCKS. There is little difference in NER with increased NEP at RVEG and LAWNS as any input of labile carbon was small due to the low NEP values.

3.24 *Net ecosystem exchange*

By combining the NER and NEP results, average NEE was found to be 138 g C m^{-2} in the NATURAL site, approximately two times greater at the cutover portion with 295 g C m^{-2} (REST-Bare), and reduced by approximately half to 170 g C m^{-2} in the REST site (Table 3.4).

Table 3.4: Seasonal CO_2-C flux (g C m^{-2}) for NATURAL, REST, and REST-Bare sites. Negative numbers represent a gain of carbon to the peatland.

	NEP	NER	NEE
NATURAL	-164	301	138
REST	-138	307	170
REST-Bare	0	295	295

Drainage resulted in an increase in CO_2 emissions of 220% in the cutover site over that of

a natural peatland and a reduction in CO_2 emissions from a cutover peatland of 58% with

active restoration.

3.3 DISCUSSION

3.31 *CO_2 flux in natural peatland*

As mentioned in chapter 2, the NATURAL site had the lowest release of CO_2 of

all the study sites. NER was high in the NATURAL site when compared with other

studies (Whiting, 1994; Waddington and Roulet, 1996; Alm *et al*, 1997; Bellisario *et al*,

1998; Bubier *et al*, 1998). The LAWNS and HUMMOCKS had opposite relationships

with the water table, with NER increasing in the lawns as water table position decreased

and NER at the hummocks decreasing with water table position. Due to the species found

in each site, there is likely an optimum moisture content where *Sphagnum* moss increases

production and therefore respiration, as the moss dries, if it is above the optimum

threshold (eg. LAWNS) or production will decrease if it is below (eg. HUMMOCKS).

These opposite relationships resulted in little variation in the modeled NER data with

changes in water table position.

The cutover sites had a higher NER than the natural sites due to drainage

increasing the depth of the aerobic zone (Chapter 2). NER was slightly higher in the

REST site than the adjacent REST-Bare site. Although the REST-Bare site may have

been slightly drier and warmer due to the lack of vegetation cover, the influence of the

plant respiration in the REST site would likely increase NER to greater than the REST-Bare site. Although the pattern of NER in both the REST and REST-Bare sites were similar, the magnitude of fluctuations was much greater in the REST-Bare site perhaps due to the absence of *Sphagnum* cover acting as a regulator to volumetric soil moisture content and soil temperature (G. Whitehead, pers. comm.).

NEP was the greatest in the HUMMOCKS and lower than reported in other studies (Bubier et al, 1998; Frolking et al, 1998). The HUMMOCKS fixed more than three times the amount of CO_2 than the adjacent LAWNS. The presence of vascular shrubs probably was the reason that the α values were much greater in the HUMMOCKS than any other *Sphagnum* covered site. The height of the shrubs and hummocks above both the peatland surface and surrounding microtopography permitted the HUMMOCKS to capture a greater amount of PAR at lower solar angles.

The NATURAL site remained a source of CO_2 for the summer of 1998. The mean NEE for the summer was higher than found in other peatlands which were found to be a source (Whiting, 1994; Waddington and Roulet, 1996; Bellisario *et al*, 1998). This is likely due to a drier than normal summer with only 78% of the 30 year mean rainfall (chapter 2). The two periods of drought depicted by low water table positions were separated by June and July which reported more rainfall than normal for those months. The LAWNS lost more CO_2 than the HUMMOCKS, which was also reported in Waddington and Roulet (1996), Alm *et al* (1997), and Bellisario *et al* (1998). The modeled NEE was similar for both sites with the exception of the two drought periods when NEE decreased in the HUMMOCKS but not the LAWNS.

3.32 *Carbon flux in restored peatlands*

Mean NER was the highest in the cutover portions of the bog (Chapter 2). NER from REST-Bare site was similar to, although slightly lower than, NER measured at a nearby site (see OLD site in Chapter 2). Spatial variability in labile carbon or the moisture regime may have been the reason for this disparity. NER was not significantly greater in the REST site than the adjacent REST-Bare site

Both the α coefficient and GP_{max} ecophysiological values were much greater at the REST site than the natural LAWN, and followed the trend of HUMMOCKS > REST > LAWN > RVEG. Considering that the REST site was restored only the previous year, the *Sphagnum* moss is growing quickly and survived two drought periods without a straw mulch covering in 1998. The presence of the vascular shrubs on the HUMMOCKS would have resulted in a greater GP_{max} than the other sites. Despite this, the GP_{max} values of the REST site were almost equal to the HUMMOCKS. This indicates that the restored *Sphagnum* PEATLAND had a higher productivity during the study season than for sites devoid of vascular shrubs. The REST site was more than two times greater than the LAWN and three fold greater than RVEG. The REST site was dominated by *Sphagnum fuscum*, although not as productive in comparison to other *Sphagnum* species when at the same water content, it is a hummock species that is insensitive to water table position and therefore less prone to desiccation (Grosvernier *et al*, 1997). The higher rate in productivity at the REST site was likely due to the more minerotrophic conditions found

in the cutover sites due to decomposition, with increased concentrations of NH_4-N, NO_3-N, and PO_4 acting as fertilizers for the *Sphagnum* moss.

The high NEP in the REST site helped offset the higher NER values, resulting in a mean NEE slightly less than half of the adjacent REST-Bare area. This indicates the importance of restoration on the CO_2 balance of cutover peatlands. Although the drainage ditches were blocked to alleviate water loss, the water table was still relatively low (MWT –35.8 cm). Considering this, the REST site only lost slightly more CO_2-C than the LAWNS, which are similar in both species and topography. When weighting the LAWNS and HUMMOCKS values according to surface coverage to determine an average NATURAL site flux, the REST site had a NER slightly greater than the NATURAL area (10.5 and 9.7 g CO_2 m^{-2} d^{-1}, respectively) and a slightly lower NEP (4.4 and 5.3 g CO_2 m^{-2} d^{-1}, respectively). This resulted in a total NEE at the REST site slightly greater than the NATURAL bog and two-thirds of the barren cutover peatland (4.4, 6.1, and 9.5 g CO_2 m^{-2} d^{-1}, at NATURAL, REST and REST-Bare, respectively). As NER was similar at the REST and REST-Bare sites, the increased NEP at the REST site was the reason for this decrease in NEE.

3.33 *Carbon flux in naturally re-vegetated peatland*

The RVEG site best represents the dominant processes involved in cutover peatlands. This site showed that soil moisture took precedence over labile carbon in respects to NER. Although the water table was at the same position for the RVEG and adjacent RVEG-Bare site, and soil temperature was greater under the *Sphagnum*

hummocks, NER was more than two times greater from the RVEG-Bare collars than adjacent RVEG collars. With the addition of plant and root respiration as well as an input of more labile carbon from plant material and newly fixed carbon thereby increasing NER, the total NER was still less than the RVEG-Bare areas which would have a lower amount of labile carbon. The cause for the difference in NER between these two sites is that the volumetric soil moisture content was much lower at the exposed RVEG-Bare sites. The decrease in soil moisture would have resulted in a greater degree of aeration of the peat, allowing oxygen to diffuse through the peat profile quicker, increasing decomposition and CO_2 gas diffusion. Although the site had been abandoned for more than 25 years, there is still a substantial flux of CO_2 from the RVEG-Bare areas, leading to the speculation that abandoned peatlands might still be a large source of CO_2 after the first three years of drainage and harvesting, as was previously suggested Schothorst (1977) and Gorham (1991). A seven-year post cutover site yielded more CO_2 than a two-year post cutover site, regardless of any differences in labile carbon (Chapter 2).

NEP was low in the RVEG site when compared to other natural studies (Whiting, 1994; Waddington and Roulet, 1996; Alm *et al*, 1997; Bellisario *et al*, 1998; Bubier *et al*, 1998) and the sites at LSJ. These naturally re-vegetated *Sphagnum* hummocks may be under more stress due to drier conditions. *Sphagnum fallax,* which dominates the hummocks at RDL, is a hollow species and is more productive when the *Sphagnum* species occurring at the sites in LSJ than the water table is higher than 20 cm (Grosvernier *et al*, 1997). However, *Sphagnum fallax* is more prone to desiccation than

the other species when the water table is deeper than 20 cm, with water content values quickly decreasing to below the threshold needed for production to occur (Grosvernier *et al*, 1997). Unlike the sites at LSJ where the drainage ditches were blocked to restore the natural water relations, ditches were only partially blocked from natural slumps at RDL (Van Seters, 1999). These hummocks therefore were not as productive as those at the REST site. Currently *Sphagnum* species have only re-colonized trenches and moist depressions at RDL (Lavoie and Rochefort, 1996), with the original method of peat extraction perhaps aiding in re-vegetation by providing a viable seed source for re-colonization, as well as the trench-baulk topography providing natural windbreaks and trees providing a shading function (Whitehead, pers. comm.).

The mean NEE at RVEG was similar to that of the NATURAL site and slightly less than the REST. The presence of the *Sphagnum* hummocks prevented the underlying peat from drying which resulted in a NER little more than half of the adjacent bare patches. However, the poor CO_2 uptake of the moss could not offset even these low rates of respiration, resulting in this site being a source of atmospheric CO_2.

The RDL site however does represent a long term abandonment scenario for a reduction in NER at other restored sites. With time, the *Sphagnum* moss hummocks will prevent a loss in soil moisture through drying processes, thereby decreasing NER. With the higher rates of production found in actively restored site over those in the naturally re-vegetated, active restoration should return the natural carbon sink to cutover peatlands faster than natural re-vegetation.

3.4 CONCLUSIONS

Drainage and harvesting increased the emissions of CO_2 to the atmosphere through a combined increase in NER and a reduction of NEP to zero. The total NER is still similar between natural and cutover sites. With drainage and harvesting, plant and root respiration has been removed and the labile carbon of the peat lowered which offsets increases in soil respiration due to increasing the depth of the aerobic zone. Although all of the sites, including the NATURAL site, were a source of CO_2 for the summer of 1998, REST-Bare emitted the largest quantity of CO_2 at LSJ, with a large reduction in NEE at REST due to the presence of *Sphagnum* vegetation and its ability to fixate CO_2. This reduction in NEE demonstrates the effect of restoration on cutover peatlands. NEE was much greater at RVEG-Bare than RVEG, both due to differences in NEP at these sites and NER. Restoration, however, has reduced this loss of carbon to the atmosphere by increasing NEP. The RVEG site demonstrates the potential for a *Sphagnum* hummock to exert control over the local hydrology by increasing the volumetric soil moisture content in the underlying peat substrate, and thereby reducing NER. Soil respiration may have been reduced at the REST site with plant respiration resulting in NER emissions similar to the adjacent Bare peat. The long term implications of this is that an increase in soil moisture in the REST site should reduce the NER with time while the higher NEP in this site may result in a total NEE similar to that of the local natural peatland.

CHAPTER 4

SUMMARY

4.0 THE IMPORTANCE OF PEATLANDS

The realization of the importance of peatlands in global biogeochemical cycles has been increasing throughout this century. Whereas once the ecosystems were simply designated as a wasteland and reclaimed for agriculture, peatlands play an important role in influencing the hydrology and nutrient cycling between upland and downstream systems, as well as trace gas exchange between the atmosphere and biosphere. Peatlands play a major role in the global carbon cycle, acting as an atmospheric sink for CO_2, storing approximately one third of the global soil carbon, and are a source of CH_4. Peatlands also contribute a large quantity of allochthonous DOC to downstream ecosystems. Considering the growing concern of global warming and recent legislation by the United Nations such as the Kyoto Protocol, the value of peatlands as a sink for atmospheric CO_2 is important to countries in regions with large peatland resources such as North America, northern Europe and Asia. While there is much research in natural peatlands, there is a lack of research on carbon cycling and hydrology in drained/cutover peatlands. This chapter summarizes the results of this thesis by presenting a conceptual model of disturbed peatland carbon cycling.

4.1 CONCEPTUAL MODEL

4.11 *Peat as a resource*

The drainage and harvesting of peatland has greatly increased over the last 50 years as the versatility of peat as a natural resource has been realized. Today peatlands are drained for four major reasons: reclamation for agricultural land, improving tree stand growth in silvaculture, horticultural, and as a fuel source. However, the extraction operations severely alter peatland dynamics in respect to its hydrological, vegetation, nutrient dynamics, trace gas exchange, and carbon sink functions.

4.12 *Changes in environmental variables*

Drainage of peat results in a decrease in the thermal capacity of the peat layer as the volume of water, which has a high thermal capacity, is reduced, leaving a lower ratio of water to peat. Peat, alone, has a much lower thermal capacity than water. The magnitude in the fluctuations of soil temperature increases post drainage due to the decreased thermal capacity (Figure 4.1, line a). The range in soil temperature at the cutover sites in LSJ was 55% greater than the adjacent natural sites.

Drainage severely changes the local hydrology of a peatland, increasing the hydrological gradient to the ditches, resulting in a loss of water. The mean water table position drops from 0 to 10 cm to 40 to 60 cm below the surface (Figure 4.1, line c). The loss of water also decreases the volumetric soil moisture of the peat, especially in the upper 5 cm, resulting in great fluctuations in moisture content (Figure 4.1, line b). Compounding this is the loss of water through evaporation, often increasing soil moisture

tensions greater than (more negative) – 100 mb (Price, 1997), the threshold of water retention in *Sphagnum* mosses (Clymo and Hayward, 1984).

The loss of water from the upper peat layer results in increased subsidence through shrinkage and oxidation in the aerobic zone and compression of the peat below the capillary fringe (Schothorst, 1977). Increased bulk density of the peat due to subsidence results in a decrease in specific yield (Price, 1997) and specific storage (Price and Schlotzhauer, 1999). The decrease in specific yield results in a greater magnitude in the water table fluctuations which will increase as bulk density increases (Figure 4.1, line b).

Subsidence was greater in the YOUNG site than the OLD site which supports the findings by Schothorst (1977) that subsidence decreases with post drainage time. This was due to differences in bulk density only. The YOUNG site has a lower average bulk density throughout the study period than the OLD site, but emissions of CO_2 were greater at the OLD site.

4.13 *Changes in the carbon balance*

NER increases with drainage as the depth of the aerobic zone increases and saturation decreases (Figure 4.1, line d). Any loss of plant and root respiration due to harvesting is offset by increased soil respiration. As shown in the previous chapters, there is a small increase in NER in the cutover sites in comparison to the natural sites. NER is a function of soil moisture in cutover peatlands, with emissions reducing to near zero when the peat is saturated following wetting events, but quickly rising as the peat

dries. Temperature may be too variable in drained peat to determine a relationship with NER without continuous measurements of both peat temperature and CO_2 exchange. Labile carbon has been found to not affect NER in the cutover sites, with NER increasing with time from two to seven years post-drainage. In fact, NER from a twenty-five year post-drainage site in RDL yielded CO_2 emissions similar to those found in LSJ.

Plant production was reduced to zero following the harvesting of the upper peat layer, resulting in a NEP of zero (Figure 4.1, line e). The reduction of NEP is the major reason for increased NEE (Figure 4.1, line f), as NEP was found to have a direct relationship on NER. In the HUMMOCKS, NER was found to be greater than in REST-Bare (Chapter 3), but the high NEP resulted in the lowest NEE from any of the sites. The REST site was found to be more productive than the natural LAWNS, with both sites being similar in species composition.

Drainage of peatlands and cutover operations has been shown to be detrimental to peatland ecosystems, causing irreversible damage to the biology and hydrology if left unattended (Schouwenaars, 1993; Ferland and Rochefort, 1997). This results in these ecosystems becoming greater sources of CO_2 to the atmosphere. CO_2 emissions have been shown to increase by approximately 200 to 300% in cutover sites, but only 123% in restored sites.
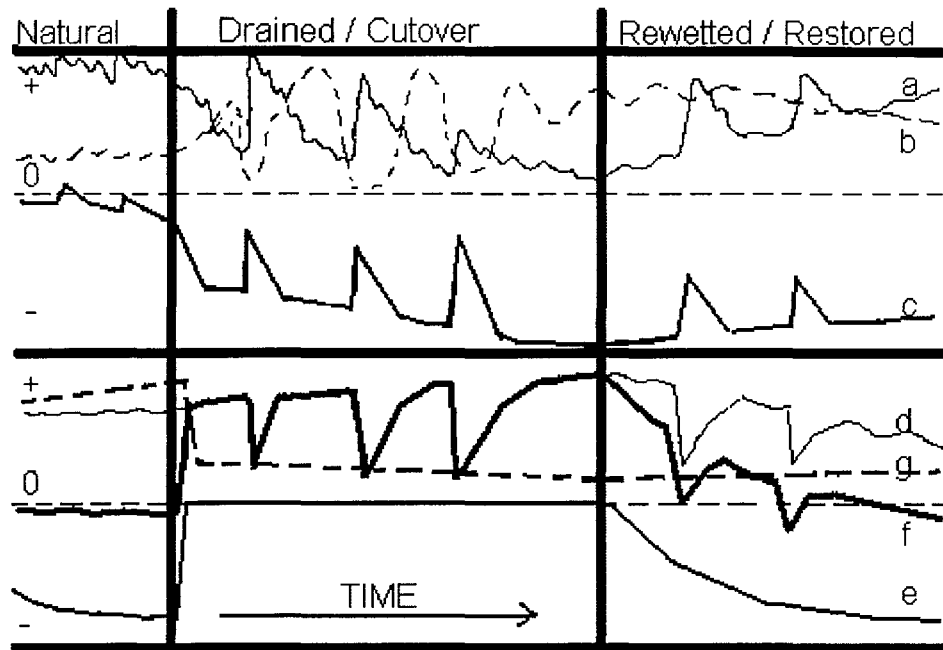
Figure 4.1. Conceptual model of the effect peatland drainage and restoration has on environmental variables and the peatland carbon balance. Components of the model are as follows: (a) volumetric soil moisture content, peaks represent wetting events, (b) soil temperature, (c) water table position, peaks represent wetting events, (d) net ecosystem exchange, (e) net ecosystem production, (f) net ecosystem exchange, and (g) total carbon accumulation.

4.2 IS NATURAL RE-VEGETATION OR ACTIVE RESTORATION EFFECTIVE?

The presence of *Sphagnum* cushions at both RDL and LSJ show that natural re-vegetation and active restoration both function in their respective sites. Whichever process should be adapted to restore the carbon sink function of peatlands is dependent on the method of peat extraction. The traditional block harvesting method is more conducive to natural re-vegetation of ombrotrophic species than more modern harvesting using heavy machinery.

Active restoration of cutover peatlands is needed with excessive harvesting operations such as those used in LSJ. Natural regeneration of *Sphagnum* and vascular plant species would occur at a very slow rate due to the lack of a viable seed base in the lower peat layer (Campeau and Rochefort, 1997). With older peat extraction techniques such as block cut method, the left over baulk material in the trenches allow natural re-vegetation to occur, such as in RDL. Even with a viable seed source to allow this, there is still only approximately 10% cover of moss, mostly in the trenches and moist depressions (Lavoie, 1997). Active restoration of cutover sites would result in a greater cover of *Sphagnum* moss and decrease the time in which natural peatland processes return to the site.

So far the natural carbon sink has not been restored in either site. This is in part due to the lack of the natural hydrologic functions which occurred before disturbance. Currently the depth of the aerobic layer may be too deep and the *Sphagnum* moss layer not extensive enough to significantly influence the soil moisture, soil properties or water table position. The decrease in NER in naturally re-vegetated *Sphagnum* hummocks

suggests that with time a complete *Sphagnum* cover may return the natural peatland

hydrology to the site and therefore return the carbon sink function of these peatlands.

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