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Peatland Restoration: An Ecohydrological Assessment

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PEATLAND RESTORATION:

AN ECOHYDROLOGICAL ASSESSMENT

By

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ABSTRACT

Pristine peatlands store approximately one-third of the world's soil carbon through the long-term accumulation of carbon as peat (Gorham, 1991). In Canada and Europe, peatlands are exploited for peat fuel and horticultural peat, which has an impact on the hydrological conditions and carbon balance of these ecosystems. Recent advances in peatland restoration techniques (e.g., Rochefort, 2000) have succeeded in the re-vegetation of *Sphagnum* moss on previously cutover surfaces. However, a peatland can only be considered functionally 'restored' after the newly formed moss layer has achieved a thickness such that the water table position in a drought year does not extend into the underlying formerly cutover peat surface (i.e. an acrotelm is developed). This study determines ecohydrological and hydrophysical properties of a newly formed peat layer, compares them to those of a nearby natural site and a naturally revegetated site and examines the spatio-temporal development of a new peat layer at a restored peatland, and from this, estimates of when the newly developing moss layer in a restored peatland will become a functional acrotelm are made.

The properties of the new peat layer differed significantly between the sites, especially for the lower (8-12 cm) layer. Lower samples for the natural and naturally revegetated sites had a bulk density of 43 ± 5 kg m^{-3} and 41 ± 11 kg m^{-3} respectively, almost twice as high as the value for lower samples from the restored site (24 ± 4 kg m^{-3}). *Sphagnum rubellum* capitula density (ρ_C) was significantly higher ($p < 0.05$) for the restored peatland (28726 # m^{-2}) compared to the natural site (26050 # m^{-2}). Residual moisture content at 200 mb (-

200 cm in soil tension) (θ_r) was significantly lower ($p < 0.05$) for the restored site in comparison to the natural and naturally revegetated sites for the lower samples (8-12 cm). This suggests that *Sphagnum rubellum* in a natural peatland is able to hold onto more moisture under increasing soil-tension than the same species growing in a restored site likely due to its higher bulk density and relatively more decomposed state.

The new moss layer thickness increased from 2.3 ± 1.7 cm in 2003 to 13.6 ± 6.5 cm in 2007 at the restored site. For the cutover (unrestored) portion of the peatland, the mean thickness values were significantly lower than the mean values for the restored portion of the site for each year ($p < 0.001$ for all years). Accumulated new peat layer biomass at the restored site increased over the six years post-restoration, ranging from 47 ± 43 g m^{-2} in 2000 to 1692 ± 932 g m^{-2} in 2005. The cutover (unrestored) portion of the site showed higher biomass accumulation for ericaceous vegetation, but lower *Sphagnum,* other mosses and other vascular biomass accumulation. A simple hydrological model was developed and determined that for the Bois-des-Bel peatland, given the mean summer water deficit at the site (-64 mm) and the storativity properties of the new moss layer (S_y = 0.34), a 19 cm thick moss layer would be required to offset summer deficit induced water table drop. Clymo's (1984) model for acrotelm growth was parameterized to estimate how long it would take to develop a 19 cm moss layer at the restored site. Model results coupled to a GIS database for the site suggest that within 17 years post-restoration, more than 50% of the site would be above the 19 cm thickness threshold, an indication that peatland ecosystem restoration from a carbon accumulation and hydrologic perspective may be achieved in the medium-term. This ecohydrological approach will aid in designing a sampling strategy that can be useful in assessing the long-term impact of restoration on peatland ecohydrology and modelling carbon sequestration.

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I am grateful to my parents who have always motivated me to pursue my goals. I also thank my brother and my friends for always showing interest in my work and for their support throughout the past few months.

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CHAPTER 1: INTRODUCTION

1.1 Introduction

Pristine peatlands store approximately one-third of the world's soil carbon through the long-term accumulation of atmospheric carbon dioxide (CO_2) as peat (Gorham, 1991). Despite their large net carbon sink function, they only cover ~5% of the global land area (Matthews and Fung, 1987), and can, therefore, be considered valuable ecosystems for the global carbon cycle (Gorham, 1991). However, this net long-term sink function of ~23 - 29 g C m^{-2} yr^{-1} can be impacted by climate and land-use change which in turn change the ecohydrology of these ecosystems. Studies suggest that pristine peatlands disturbed through drainage and peat extraction become a large and persistent source of CO_2 to the atmosphere due to increased soil respiration and CH_4 oxidation (Waddington et al., 2002; Gorham, 1991).

In both Canada and Europe peatlands are exploited for peat fuel and horticultural peat which has an impact on the hydrological conditions and carbon balance of these ecosystems. The uppermost layer of peat (acrotelm) that is characterized by a high porosity, specific yield and saturated hydraulic conductivity is removed during extraction resulting in a generally lower and 'flashier' water table position, limiting water availability for peatland vegetation (Price et al., 2003). This is particularly damaging to *Sphagnum*, one of the dominant peat-forming moss species in peatlands, since *Sphagnum* is a non-vascular plant that requires a constant supply of water (Price, 1997).

Recent advances in peatland restoration techniques (e.g., Rochefort, 2000) have succeeded in the re-vegetation of *Sphagnum* moss on previously cutover surfaces. However, it remains unknown whether the restored moss layer is able to return to a self-regulating layer that aids in returning the natural peat accumulating function of the peatland. While research has demonstrated that a peatland can return to a net sink for CO_2 within three to five years post-restoration (Greenwood, 2005), it can be argued that a peatland can only be considered functionally 'restored' once the newly formed layer becomes an acrotelm (Smolders et al., 2003; Wheeler and Shaw, 1995). That is, that this new layer has achieved a thickness such that the water table position in a drought year does not extend into the underlying formerly cutover peat surface. From a peatland development perspective, Clymo (1984) defines this as when the diplotelmic nature of the bog is restored and accumulation of organic matter is transferred to the catotelm for the long term storage of carbon in the system. For this, a functional acrotelm must be present, so that water table fluctuations are maintained within this 'self-regulating' layer. The aim of this thesis is to examine the development of a new peat layer in a restored peatland, determine whether this regenerating layer has similar characteristics to those of a natural peatland and develop a model to predict when, from an ecohydrological perspective, this layer will be considered a functional acrotelm.

1.2 Peatland Ecosystems

Peatlands are ecosystems characterized by waterlogged conditions in which accumulation of organic matter below the surface takes place (Gorham, 1991) to depths greater than 40 cm (National Wetlands Working Group, 1988). They include bogs, fens, marshes and some swamps (Tarnocai, 1998), and are classified according to nutrient conditions, water sources and vegetation composition (Rydin and Jeglum, 2006). The amount of carbon stored in peatlands is estimated to be 455 Pg, representing almost one-third of the total global soil carbon (Gorham, 1991).

Bogs are ombrotrophic peatlands, isolated from laterally moving groundwater and receive all of its water inputs from precipitation. *Sphagnum* moss is a keystone species in peatlands, as it helps maintain a high water table and decrease pH levels, thereby stimulating the development of peat bogs (Chirino et al., 2006). This species is the dominant peat forming species and peat accumulation occurs when old remains of the moss die and become buried (Clymo, 1984). Clymo and Hayward (1982) suggest that *Sphagnum* species likely store more carbon, in living tissue and as peat, than in any other plant genus in the world. The process of peat formation has been described by Clymo (1984). Briefly, plant matter is added to and accumulates on the surface due to primary productivity and as a result of rapid decomposition in this layer, the structure of buried remains of the moss is lost, increasing bulk density and decreasing pore space in the lower layers of the peat profile.

1.3 The Diplotelmic Peatland

Many peatlands have a diplotelmic structure consisting of an upper acrotelm overlying a lower catotelm (Ingram, 1978). The acrotelm consists of living and poorly decomposed plant material in the surface layer, is typically ~50 cm thick and encompasses the full range of water table fluctuations in the summer season (Ivanov, 1981; Price et al., 2003). The hydrological properties of the acrotelm often change greatly with depth in the profile. For example, specific yield has been shown to decrease from 0.5 to 0.1 (Price, 1992) while hydraulic conductivity decreases by up to 4 orders of magnitude (Romanov, 1968). These changes in hydrological properties are a result of the collapse of large pore spaces between plant materials in this layer with depth below the surface (Ingram, 1983; Clymo, 1984). Consequently, the position of the water table in the acrotelm strongly controls the amount of lateral water flow, where under wet conditions, the large pore spaces are already filled and additional water input is allowed to flow through the acrotelm and during dry periods, water input is immediately used to fill these large pore spaces and loss through lateral seepage is minimal (Ingram, 1983; Romanov 1968). The catotelm is composed of older and well decomposed peat, permanently saturated (Clymo, 1984) and is characterized by its low and less variable hydraulic conductivity (Ingram, 1978; Boelter, 1965) and specific yield (~0.05) (Price, 1996). The smaller pores present in this layer, a result of its high state of decomposition, are more difficult to drain under gravity so water retention in the catotelm is higher (Price et al., 2003). Conversely, the water storage properties of the acrotelm, mainly the specific yield, stabilize the water table and maintain it close to the surface, an essential factor for the development of *Sphagnum* species in these ecosystems (Clymo, 1983). Consequently, the structural differences of the acrotelm and catotelm regulate water transport and storage in natural bog systems (Romanov, 1968), such that damage or disturbance to this structure can alter the hydrological conditions and natural functioning of these systems (Price et al., 2003).

As mentioned earlier, decomposition in the catotelm is very slow leading to long-term peat accumulation. During peatland development peat is transferred to and stored in the catotelm as the water table rises and captures dead plant material deposited in the acrotelm (Clymo, 1984). At steady state, the rate at which matter is transferred from the base of the acrotelm to the top of the catotelm is given by:

$$\frac{dM_a}{dt} = p_a - \alpha_a * M_a \qquad [1.1]$$

where M represents the accumulated mass (kg m^{-2}), p is the annual input of dry biomass (kg m^{-2} yr^{-1}), α is a ratio representing mass lost by decay, t represents time in years, and subscript a refers to the acrotelm. Moreover, Clymo (1984) suggests that a constant, very small proportion of all peat in the catotelm is lost by anaerobic decay and hence peat accumulation in the catotelm can be described by:

$$\frac{dM_c}{dt} = p_c - \alpha_c * M_c \qquad [1.2]$$

A steady state, where no net gain or loss of matter takes place and peat accumulation is suppressed, is reached when mass lost by decay at all depths equal plant production by mass. Therefore, the thickness at which decay integrated over all depths balances productivity is considered the stable thickness or the limit to peat bog growth (Clymo, 1984). Since its original publication, several modifications have been suggested to the original peat growth model (Belyea and Baird, 2006; Clymo et al., 1998; Clymo, 1992; Winston, 1994; Yu et. al., 2003). Decay rate was allowed to decrease with proportion of original mass remaining in Clymo's (1992) model, which contradicts the original concept of a limit to bog growth but still allows accumulation to decrease with time.

Peatland formation is therefore highly dependent on the balance between ecological and hydrological processes. This coupling between ecology and hydrology is of fundamental importance for the development and natural functioning of peatlands and disruption of these fundamental peatland processes can lead to profound impacts to peatland ecosystems.

1.4 Peat Extraction

Canada contains roughly one-third of the global peatlands, which cover 113 million hectares or 12% of the country's total area (Daigle et al., 2001). Due to its high commercial use and value, peat and peat moss are valued natural resources and exploitation of peatlands is an important industry in some regions of Canada (Robert et al. 1999; Daigle et al., 2001). Horticultural uses include peat for fertilizer mixes, composting and potting media among others (Bélanger et al., 1988). Approximately 12,000 ha of Canadian peatlands have been drained and harvested for horticultural purposes (Cleary et al., 2005), and although this represents less than 2% of Canada's total peatland resources (Daigle et al., 2001), areas of intense exploitation such as the St. Lawrence Lowlands have lost over 70% (Lavoie and Rochefort, 1996) of their original peatlands.

Historically, peat moss extraction was carried out by manual block-cutting of the peat. Drainage ditches were dug to facilitate peatland drainage and ease of cutting, surface vegetation removed, and trenches were created by cutting parallel rows of peat blocks that were stacked and placed on the undisturbed surfaces or baulks, located between trenches. Peatlands exploited by this method were characterized by an alternating pattern of trenches and baulks that can usually still be detected on the present day surface of these sites. The block-cut method was later replaced in the late 1960's by the vacuum mechanized method. Intense drainage to increase the bearing capacity for mechanized cutting is necessary for this method due to the use of heavy machinery. Surface vegetation is completely removed, and the surface is then harrowed to allow for drying of peat and vacuum harvested with machines (Robert et al., 1999; Frilander et al., 1996). Extraction of peat, by both the block-cut and vacuum extraction approaches inevitably impacts the natural conditions of peatlands such that the hydrology of an extracted peatland is markedly different from a natural peatland (McNeil and Waddington, 2003).

The removal of the acrotelm and exposure of the much denser and less easily drainable catotelm at the new surface of the peatland combine to cause the water table position to decrease and to become much more variable. During the summer months, when evaporative demands are high which in turn causes soil moisture and soil-water pressure to decrease, water demands are supplied mainly through changes in water storage of the unsaturated zone (Price 1996; 1997). Moreover, peat subsidence can result from this lowering of the water table due to shrinkage and oxidation of peat above the water table and compression below (Schothorst, 1982; Price and Schlotzhauer, 1999), which in turn decreases pore size. Hydraulic conductivity and specific yield also decrease as a result of extraction (Price et al., 2003) in the short term due to removal of the acrotelm and exposure of the catotelm, and in the longer term due to peat subsidence and oxidation. Price (1996) showed that removal of the acrotelm caused S_y to decrease from 0.55 to 0.2 due to exposure of the catotelm. S_y then declined to 0.05 within 5 years of extraction as a result of peat subsidence and oxidation.

Sphagnum moss is tremendously impacted by the hydrological changes following peatland extraction. When soil-water pressure drops below -100 cm (mb), which is rather common in drained peatlands (Price and Whitehead, 2001), this non-vascular plant becomes unable to generate capillary forces to draw water upwards, and therefore cannot survive long periods of drought. If left abandoned, cutover peatlands have little or no chance to recover their natural conditions and functions (Heathwaite, 1994). Nevertheless, Lavoie and Rochefort (1996) noted that spontaneous regeneration of

Sphagnum in block-cut peatlands was possible; however this was only seen in certain areas where hydrological conditions appeared to be more beneficial. Spontaneous *Sphagnum* regeneration is much rarer in vacuum extracted peatlands, where low water table levels throughout the site prevent *Sphagnum* recolonization (Lavoie et al., 2003).

1.5 Peatland Restoration

Active restoration of exploited peatlands has as one of its main goals to re-establish the keystone *Sphagnum* vegetation at cutover sites (Wheeler and Shaw, 1995), since this is fundamental for the restoration of the hydrology, carbon sequestration properties and to the successful repair of these ecosystems (Ferland and Rochefort, 1997; Rochefort, 2000). Another main goal of restoration is to return the diplotelmic hydrological layers that characterize natural peatlands (Rochefort, 2000), through the development of an upper living moss layer and the return of its natural peat formation function. In this manner, promoting the reestablishment of *Sphagnum* moss aids in the development of a new acrotelm, responsible for maintaining the water table close to the surface and regulating water storage and flow in peatlands, further facilitating hydrological conditions for *Sphagnum* growth. This coupling between the ecology and hydrology is of fundamental importance for the development and natural functioning of peatlands, and is the basis for developing restoration measures for extracted and abandoned bogs.

The North American approach to peatland restoration is described in detail in Rochefort et al. (2003) and includes the following processes: field preparation, *Sphagnum* diaspore collection, diaspore introduction, straw mulch application, and in some cases fertilization. In order to return suitable hydrological conditions for *Sphagnum* moss re-establishment, drainage ditches are blocked to rewet the peat surface (Quinty and Rochefort, 2003) and dykes are created to retain spring melt water and prepare the field. *Sphagnum* diaspores are introduced from a natural donor peatland. Straw mulch is applied to protect diaspores, and help improve microclimatic conditions by reducing tension stress in the surface layer and increasing volumetric moisture content (Price et al., 1998). Finally, phosphorus fertilizer is applied in some cases to enhance vascular plant colonization, which act as companion species to *Sphagnum* growth and development (Rochefort et al., 2003). As a result, peatland restoration has the potential to return cutover and abandoned sites to carbon accumulating systems (Waddington and Price, 2000).

Encouraging results have been observed from studies conducted at restored peatlands. For example, Shantz and Price (2006) showed that runoff decreased, mean water table position increased and volumetric moisture content in the upper 5 cm layer increased and was maintained above 50% following the restoration of a peatland. However, the range of water table fluctuation increased in comparison to the non-restored portion of the peatland. The authors suggest that restoration contributed to the recovery of the hydrological conditions necessary for *Sphagnum* re-establishment. Waddington and Warner (2001) showed that CO_2 emissions to the atmosphere decreased post-restoration, with ~70% of this decrease accountable by an increase in gross ecosystem production. Moreover, Greenwood (2005) suggests that it is possible to return a restored peatland to a net sink for atmospheric CO_2 in less than five years post- restoration due to a reduction in peat respiration and an increase in ecosystem productivity from evolving vegetation.

1.6 Thesis Objectives

While it has been demonstrated that standard peatland restoration practices can facilitate the return of hydrological and ecological conditions that more closely resemble those of pristine peatlands (e.g. Shantz and Price 2006; Greenwood, 2005), it may take many years following restoration for an acrotelm to develop and consequently, as it has been hypothesized, for the carbon exchange functions and hydrology of the site to be considered fully restored (McNeil and Waddington, 2003). Gaining a better understanding about the development of a new peat layer post-restoration as well as its ecohydrological properties is essential for evaluating peatland restoration. Comparing the ecohydrological properties of natural and naturally revegetated peatlands to restored peatlands can aid in assessing peatland restoration success and further our understanding about the characteristics of colonizing *Sphagnum*. This in turn aids peatland restoration through an enhanced knowledge on this keystone species. Furthermore, this can also provide insight in predicting when a new peat layer can be considered restored from an ecohydrological perspective, in other words when the new moss layer will become a self-regulating acrotelm as this is the ultimate goal of peatland restoration. Studying restored peatlands can provide insight and valuable information in order for researchers and managers to develop more efficient long-term monitoring programs and strategies that can be useful in assessing the long-term impact of restoration on peatland ecohydrology and modelling carbon sequestration, ultimately enhancing restoration through better planning and management.

In this manner, this thesis examines the development of the new peat layer in a restored peatland, determines the properties of this newly forming layer and compares it to those of an adjacent natural site. More specifically, the objectives of this work are to:

1. Examine the spatio-temporal development of a new peat layer at a restored peatland,
2. Determine ecohydrological and hydrophysical properties of this newly formed peat and compare them to those of a nearby natural site and a naturally revegetated site, and
3. Develop a simple ecohydrological model to estimate when the developing layer in a restored peatland will become a functional acrotelm.

This work will enhance knowledge on the impacts of restoration on peatland ecosystems. Peatland restoration, through good management practices and scientific research, can aid in the re-habilitation and eventual restoration of otherwise damaged bogs and also has the potential to lessen anthropogenic impacts to natural peatlands. This research study will further the understanding of the link between hydrology and ecology in peatlands and also provide basis to the enhancement of models and knowledge for the improvement of restoration techniques.

CHAPTER 2: METHODOLOGY

2.1 Overview

This study adopts field, laboratory and modeling approaches to meet the objectives of the thesis. Ecological properties and measurements including biomass accumulation, net primary productivity (NPP), peat decomposition rates, moss thickness and capitula density were determined at a formerly vacuum extracted peatland undergoing restoration in the Bas-Saint Laurent region (see below). Moreover, hydrophysical properties including bulk density, porosity, von Post degree of decomposition, specific yield and moisture retention were determined for the restored peatland as well as for two other sites: a natural peatland and a naturally revegetated block-cut peatland. A simple ecohydrological model, coupled to a geographic information system (GIS), was used to assess the long-term effects of restoration on peatland hydrology and ecology and to estimate when the peatland could be classified as 'ecohydrologically restored'.

2.2 Study Areas

This study was conducted at two peatlands (Bois-des-Bel and Cacouna bog) in the Bas-Saint-Laurent region of Québec, ~ 14 km east of Rivière-du-Loup (42°58' N, 69°25' W). The 30-year normal (1971-2000) annual temperature for the region (St. Arsène, 5 km south) is 3.2 °C, with mean January and July temperatures of -12.6 and 17.8 °C, respectively. The mean annual precipitation is 962.9 mm, with 29% falling as snow (Environment Canada, 2008).

The Bois-des-Bel (BDB) peatland is a 200 ha treed bog of which an 11.5 ha section of the peatland was drained in 1972 and the upper 80 cm of peat was extracted from this site using the vacuum approach from 1973 to 1980. The site was then abandoned until 1999 (Figure 2.1). The abandoned peatland was divided into 11 fields (30 m x 300 m each), and separated by drainage ditches running parallel (north-south) to the fields. In the autumn of 1999, ecosystem scale restoration commenced using the standard Canadian restoration technique as outlined in Rochefort et al. (2003). Initially the peat surface was cleared of all vegetation and woody material that was present after abandonment. Drainage ditches were blocked and dykes (low-lying peat walls) were created to retain snowmelt. *Sphagnum* fragments were also introduced from a natural donor peatland, and a straw mulch cover applied at a rate of 3000 kg ha^{-1} in order to protect the fragments. Lastly, phosphorus fertilizer was applied at a rate of 15 g m^{-2} to enhance vascular plant colonization (Rochefort et al., 2003). The cutover peatland was separated into two sections: a 7.5 ha restored section (westernmost peat fields 1-8) and a 1.8 ha cutover section (easternmost fields 10-11) with a buffer strip left in-between the two sections (field 9). The restored section of the Bois-des-Bel peatland was further divided into four zones (1-4) and separated by dykes. Zones 2, 3 and 4 were restored in the fall of 1999, while zone 1 was restored in the fall of 2000. The average peat depths of the restored and cutover sites are 1.5 and 1.6 m respectively, and the peatland is underlain by a layer of marine clay that hinders vertical flow through the base of the peat deposit. The dominant species found at Bois-des-Bel post-restoration include: *Polytrichum* spp., ericaceous shrubs, *Eriophorum vaginatum* and *Typha latifolia*. Furthermore, *Sphagnum rubellum* was present at the restored site while *Picea mariana* and *Betula* spp. were present at the cutover site. Lavoie et al. (2001) provide further details on the site conditions and vegetation distribution within the undisturbed portion of the peatland. The natural portion of the peatland is located adjacent to the restored site.

The Cacouna bog was originally 210 ha but conversion to agricultural lands and construction of roads has reduced its total area to 172 ha (Girard, 2000) (Figure 2.2). A railway constructed in the mid-1800s divides the peatland into north and south sections along a natural groundwater divide. The two sections of the bog now show distinct hydrological properties (Van Seters and Price, 2001). Peat extraction for horticultural use began in 1942 using the block-cut method, and a series of trenches and raised baulk areas resulted from using this peat extraction method. Drainage ditches, created to facilitate peat extraction, were blocked naturally and manually (in 2006) by peat slumping after the peatland was abandoned in 1976 (Girard, 2000). The Cacouna peatland has been naturally colonized by ericaceous shrubs and *Sphagnum* species (primarily S. *capillifolium*). For the purpose of this study, *Sphagnum capillifolium* samples were collected from this peatland to determine the hydrophysical and ecohydrological properties of a naturally revegetated moss layer in comparison to the main study site, the actively restored Bois-des-Bel peatland.

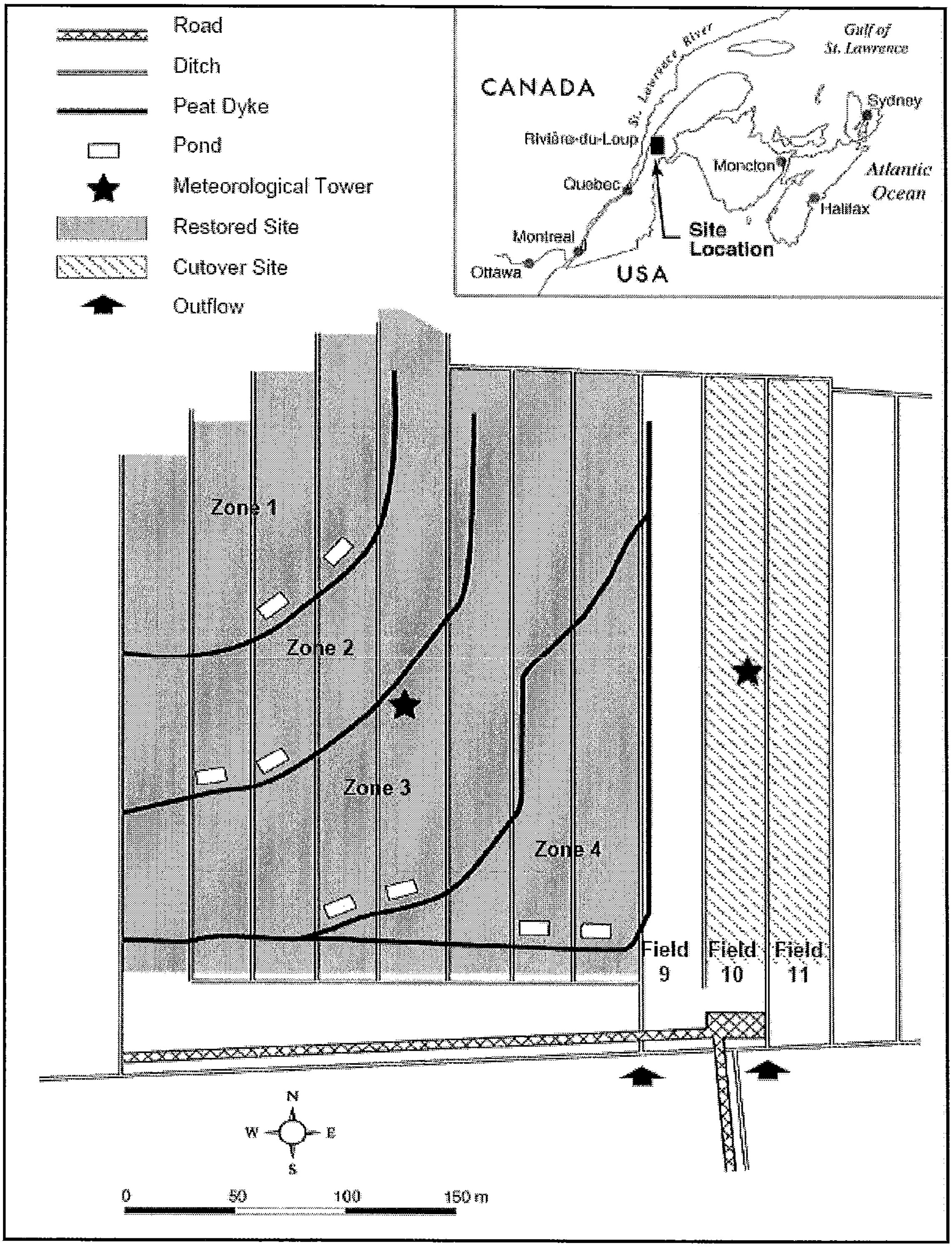

Figure 2.1: Site map of the Bois-des-Bel peatland.

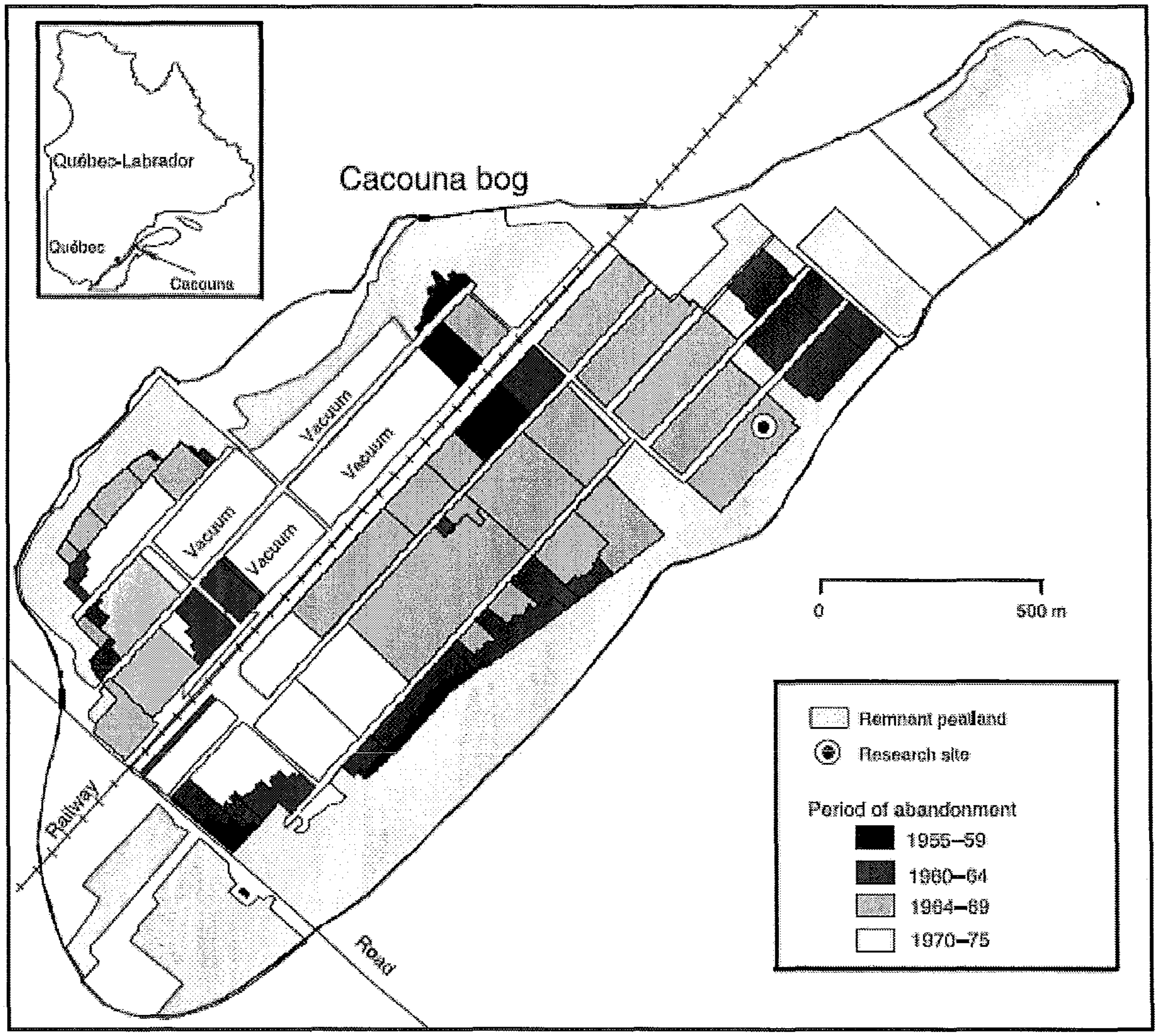

Figure 2.2: Site map of the Cacouna bog.

2.3 Ecological Properties of the New Peat Layer

2.3.1 Biomass accumulation

At the end of the each growing season starting in 2000 until 2005, biomass samples were removed from 74 quadrats each measuring 25 by 25 cm by clipping all above ground vegetation present over the cutover peat substrate surface. Samples were obtained along 10 transects, which corresponded to the fields at the site (1-8 restored site, 10-11 cutover portion). If there was no *Sphagnum* present, biomass was cut to the discoloration line of the *Polytrichum* mosses. Samples were frozen and returned to Laval University for analysis.

Sample material was sorted by species and into leaves and stems following the procedure of Moore et al. (2002). Specifically for *Polytrichum* mosses; brown, red or yellow mosses are considered green biomass. For herbaceous species, any leaves that were totally dead (denoted by dark colour) were placed in the litter category. However, different conditions apply if only portions of the leaf were dead, specifically, if more than 50%, then the dead portion was removed and put it in the litter category, if less than 50% dead it would be included as green biomass. For vascular species, the whole leaf was kept in the green biomass category unless it was 100% dead. If there were any adventitious roots present (often in samples with a lot of *Sphagnum*) on ericaceous plants, they were cut and placed in the belowground biomass division. Plant tissues were dried at 70°C for 48 hours and weighed and biomass was converted to g m^{-2}. NPP was calculated as the change in biomass accumulation over each one year period.

2.3.2 New peat layer thickness survey

During the 2003, 2005 and 2007 field seasons, the thickness of the newly formed peat layer (above the abandoned cutover peat surface) was measured at 755 point locations, at the center of each grid cell (see Figure 2.3), including the restored and cutover portions of the Bois-des-Bel peatland by inserting a metal rod into the peat until the older, highly decomposed and compacted cutover peat layer was reached. This transition between the older cutover surface and new peat layer was easy to identify due to very distinct physical properties of the older catotelmic peat and the newly formed moss layer (see Waddington et al., 2003). The survey was conducted along transects 7.5 meters apart, that ran the length of the 11 peat fields present at the site, starting from the south end, every 20 meters. Thickness values were recorded and entered into a GIS database for the site.

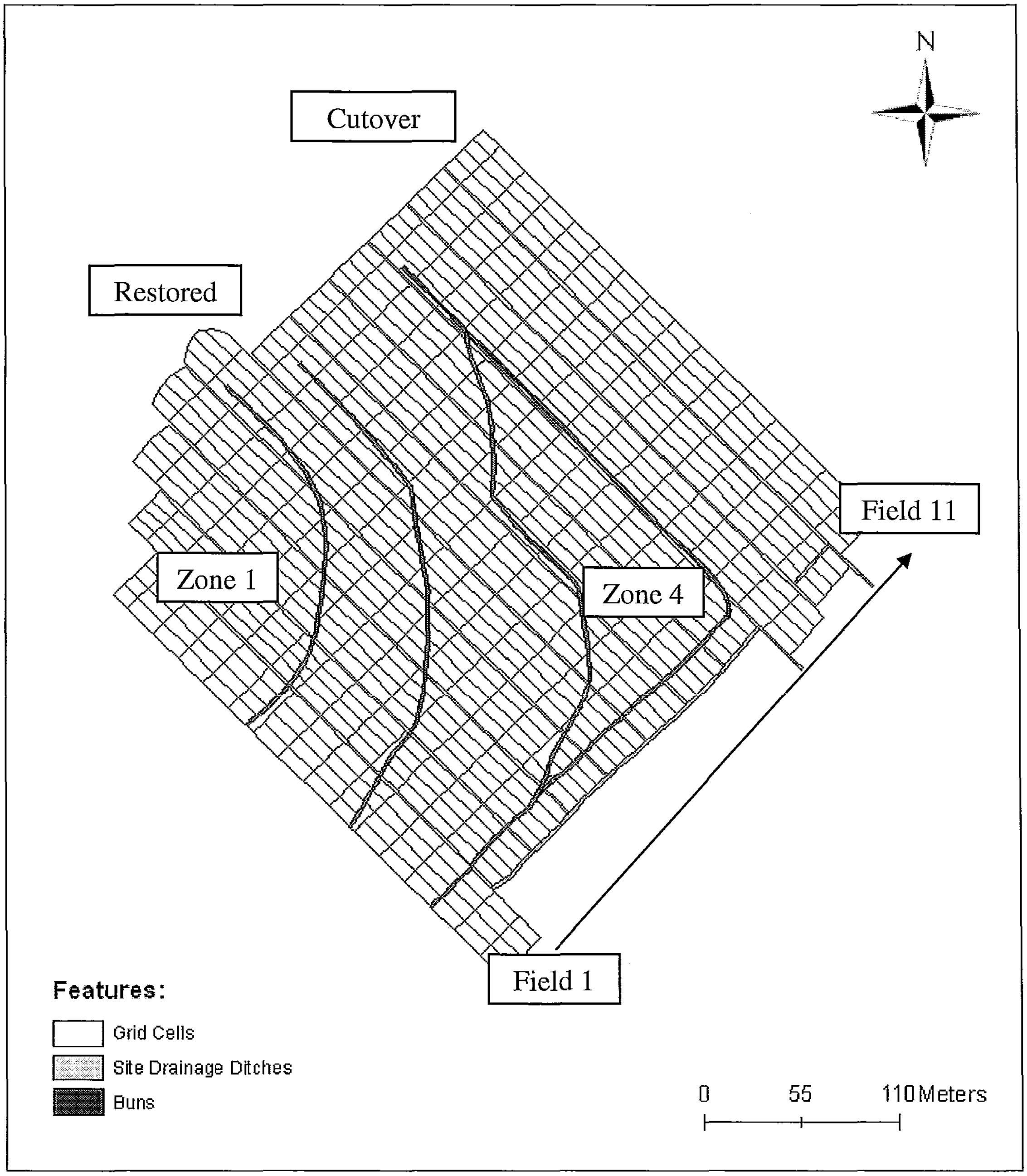

Figure 2.3: Grid cell scheme used for sampling new moss layer thickness.

2.3.3 Decomposition rates

Sphagnum decomposition rates were determined over one growing season using the litter bag technique (Moore, 1984). Different treatments were applied, namely: peat type present in bag (old peat present before restoration or new peat comprised of over 90% *Sphagnum rubellum* 2-3 cm below the capitula), depth (placed in the old peat or in the newly formed layer), five dominant conditions at the surface (predominantly *Sphagnum* cover with few vascular plants, *Sphagnum* cover with ericaceous shrubs present, *Polytrichum* moss cover with ericaceous shrubs, non-restored surface with no plant cover and natural peatland), and three different replicates (locations). Five litter bags were used for each set of treatment for a total of 210 litter bags.

Litter bags were made of fine nylon netting with small opening sizes (~ 0.5 mm). Bags were weighed, filled with plant material, dried at 60 °C for 24 hours, and then weighed again in order to record the amount of dry material present. Litter bag replicates were installed in restored zones 1, 3 and 4 on October 16, 2006 and removed on October 11, 2007. Following removal, litter bags were returned to the laboratory at Laval University where roots and fine particles were removed from outside of the bags. The remaining plant material within the litter bag was dried at 60 °C for 24 hours. Mass loss was calculated as the difference between the initial dry mass and final dry mass, in proportion to the initial dry mass. Values that showed negative loss (mass gain) over the one year period were discarded from the dataset, which resulted in loss of ~10% of the total dataset.

2.3.4 Capitula density

Sphagnum rubellum capitula density (expressed as the number of capitula per m^2) was determined at 18 and 23 sites in the natural and restored portions of the BDB peatland as well as from 7 sites at the Cacouna bog. High resolution (8 megapixels) digital photographs were taken of 100 cm^2 *Sphagnum* quadrats, with a ruler as a scale in each photo. Photographs were analyzed in a photo editing program and used to provide a count of the number of capitula per 10 x 10 cm^2 quadrats. Photos that did not provide a clear view of capitula or could not yield 10 x 10 cm^2 quadrats were disregarded.

2.4 Hydrological Properties of the New Peat Layer

2.4.1 Moisture survey

Spatial moisture surveys were conducted to examine volumetric water content (VWC) throughout the restored portion of the Bois-des-Bel peatland in July and August of 2007. The same grid cell scheme used for the thickness surveys were used for the moisture surveys, yielding a total of 500 point locations sampled within the restored site. Surveys were conducted over the course of a day period. A WET Sensor (Type WET-2, 7 cm long probe, ± 4% accuracy) (Delta T-Devices, Cambridge, UK) provided instantaneous readings for the dielectric properties of the soil which were used to calculate VWC. VWC values were determined later in the laboratory by calibrating the sensor for the peatland's specific peat soil by obtaining simultaneous measurements of WET Sensor VWC readings and actual VWC as calculated by weight and volume of samples. Specifically, a linear relationship between the refractive index (the square root of WET sensor VWC reading) and actual VWC determined by weight was used to provide a calibration curve.

2.4.2 Water table position

Water table position at Bois-des-Bel was monitored during the growing seasons from 1999-2002 and in 2006. Manual point water table measurements were taken regularly at permanent slotted PVC wells spatially distributed throughout the restored and cutover portions of the Bois-des-Bel peatland in order to assess the impact of restoration on the hydrology of the site. The well network included wells installed up and down slope of the four main dykes to evaluate water table gradients.

2.4.3 Specific yield

Specific yield was determined by allowing free drainage of saturated soil samples and determining the ratio of water loss (Price, 1996). Peat-blocks were removed from the three study sites (two from the natural Bois-des-Bel site, two from the Cacouna bog and seven from the Bois-des-Bel restored site), frozen and returned to the Ecohydrology Laboratory at McMaster University. Samples from the upper and mid sections of the peat blocks were cut while frozen to be contained in cylindrical PVC 'pucks' (8 cm diameter, 4 cm deep) and held in place by a fine nylon screen attached to the bottom of the pucks. For the Bois-des-Bel peatland, a total of 28 samples for the new moss layer (16 samples for the upper 0-4 cm depth from the new moss surface and 12 samples for the 4-8 cm depth) and 29 samples for the old peat layer (15 samples for upper 0-4 cm depth from the old peat layer surface and 14 samples for 4-8 cm depth) were considered, with samples obtained throughout the site from seven different locations. A specific yield (S_y) profile was developed for the restored site new peat layer and underlying catotelm peat. From the natural site, a total of 11 samples (4 for upper 0-4 cm and 7 for 8-12 cm) were considered to determine S_y of the moss layer. For the Cacouna bog, 10 samples (5 for upper 0-4 cm and 5 for 8-12 cm) were used.

Samples were saturated from below, by slowly filling a small tub containing the samples until water reached a level just below the top of the samples and allowing them to saturate for 24 hours. Samples were then allowed to drain for 24 hours and specific yield (S_y) was calculated using the following equation:

$$S_y = \frac{(M_{SAT} - M_{DRAIN}) / \rho_W}{M_{SAT} / \rho_W} \qquad [2.1]$$

where S_y ($m^3\ m^{-3}$) represents the specific yield ($\leq$ porosity), M_{SAT} is the saturated weight of the sample, M_{DRAIN} is the weight of the sample after it was allowed to drain and ρ_W is the density of water, assumed to be 1 g cm^{-3}.

2.4.4 Soil-water retention curves

Soil samples were cut into several cylindrical capitula and mid-acrotelm cores (8 cm diameter, 4 cm deep) with a hole-saw while frozen to prevent compression and disturbance to the soil structure at the McMaster Ecohydrology Laboratory. Capitula cores were taken from the upper 0-4 cm depth from surface and mid-acrotelm samples taken from the 8-12 cm depth. Seven replicates of sub-samples at each depth for all three sites, Bois-des Bel restored, Bois-des Bel natural and Cacouna bog were analyzed to determine differences between sites and depths in soil-water retention curve characteristics.

Soil-water retention curves were determined on all cores using a methodology fully described by Dane and Hopmans (2002) and Klute (1986). Briefly, the frozen soil samples (contained in PVC rings) were thawed and saturated in de-aired water for 48 hrs prior to testing to achieve full saturation of the peat. Samples were then placed on top of a high flow 0.5 bar porous ceramic plate cell (effective pore size 6.0 micron, hydraulic conductivity .0000311 cm/sec and approximate porosity of 50% by volume) within a 5 bar pressure plate extractor (Model 1600, Soil Moisture Equipment Corp., Santa Barbara) and sealed. Various pressures were applied (5, 10, 15, 20, 25, 30, 35, 40, 60, 80, 100, 120 and 200 mb) using a pressure manifold (±1% accuracy) (Model 700-3, Soil Moisture Equipment Corp. Santa Barbara) until equilibrium of the samples was reached within the pressure cell. A water outflow tube was connected from the pressure extractor into a burette, which allowed outflow to be measured. Equilibrium of the samples at each pressure was reached when the outflow of water had ceased (generally 1 to 2 days). Samples were weighed prior to moisture extraction and after each pressure change once equilibrium was reached. The porous plate was sprayed with a fine mist of water after each equilibrium point to re-establish the hydraulic contact between soil and porous plate. Cheese cloth was also attached to the bottom of the samples, a practice which ensures soil loss is minimized, prevents plates from clogging due to soil loss and aids in maintaining hydraulic contact between the soil and porous plate (see Klute, 1986). Volumetric water content (θ) at any given pressure (ψ) was determined at equilibrium using the following equation:

$$\theta = \frac{(M_{WET} - M_{DRY})}{\rho_w * V_s} \qquad [2.2]$$

where M_{WET} is the mass (g) of the moist soil at equilibrium, M_{DRY} is the mass (g) of the oven-dry soil (obtained by oven drying samples at 70 °C for 72 hours), ρ_w is the density of water (g cm^{-3}), and V_s is the volume of the soil sample (cm^3). The soil-water retention relationship is then described using tension and VWC values.

The soil water retention model RETC (van Genuchten et al., 1991) with van Genuchten's (1980) model to describe soil water retention curves was used to fit the observed moisture retention data. Curve fitting parameters m, n and α were estimated with RETC for θ (ψ) relationships, fitted through least square error analysis. The model was run with the empirical constant m=1-1/n. The soil-water content as a function of pressure head (h) is given by:

$$\theta(h) = \theta_r + \frac{(\theta_s - \theta_r)}{\left[1 + (\alpha * h)^n\right]^m} \qquad [2.3]$$

where θ_s is the saturated moisture content, θ_r is the residual moisture content, and α, m and n are curve fitting parameters.

2.4.5 Bulk density, porosity and von Post degree of decomposition

Bulk density of soil samples used in the moisture retention experiment was determined by oven drying samples of known volume for 72 hours at 70 °C. Dry mass was recorded and used to calculate bulk density, given that bulk density (ρ):

$$\rho = \frac{M_{DRY}}{V_s} \qquad [2.4]$$

where M_{DRY} is the mass (g) of the oven-dry soil and V_s is the volume of soil sample (cm^3).

Porosity (ϕ) was calculated from the obtained bulk density values:

$$\phi = 1 - \frac{\rho}{\varphi} \qquad [2.5]$$

where ρ is bulk density (g cm^{-3}) and φ is particle density (1.55 g cm^{-3}) based on literature values for peat (Clymo, 1970).

Von Post degree of decomposition was determined for lower samples using a standard method (von Post, 1922). The peat samples were squeezed within a closed hand, and according to observations regarding the colour of the solution that was expressed between the fingers, the nature of the fibers, and the proportion of the original sample that remained in the hand, the soil sample was placed in one of ten classes (H1-H10)

2.5 Ecohydrological Model for Acrotelm Development

2.5.1 GIS analysis

The thickness of the new peat layer was measured throughout the site at the center of each grid cell (as mentioned earlier) and entered into a GIS database for analysis. ArcGIS 9.2 GIS and mapping software (ESRI) was used for analysis of data and production of maps. Point values for thickness survey were spatially referenced and converted into a shapefile based on a standard grid cell shapefile developed for the site. Data was compiled into a main shapefile that contained thickness values measured in 2003, 2005 and 2007 for each grid cell.

2.5.2 Acrotelm growth model

The concepts of Clymo's (1984) model for acrotelm growth were used and the model parameterized for the Bois-des-Bel peatland to predict the growth of the newly forming peat layer and to compare model predictions to actual field measurements. The model provides a representation of biomass accumulation per unit area over time and takes into consideration productivity as an input and rate of loss to decay as an output. A modification to the decay component of the original model (Clymo, 1984) was made by Clymo (1992); allowing decay to decrease with proportion of original mass remaining and this was used in the acrotelm growth model below:

$$\frac{dM}{dt} = p_a - \alpha_a (m_t / m_o) * M \qquad [2.6]$$

where M represents cumulative mass (kg m^{-2}), t represents time in years, m_o original mass, m_t mass after time t, p_a is the annual input of dry biomass (kg m^{-2} yr^{-1}) and α_a is a ratio representing the mass lost by decay. In order to obtain the annual thickness of the accumulated layer, biomass accumulation values were converted to linear growth by incorporating the bulk density of the acrotelm layer into equation 2.6.

In addition, a simple hydrologic model was developed to predict when the newly forming moss layer will be thick enough to contain all water table fluctuations and become a functional acrotelm. The model takes into account overall site water storage, represented by the difference of precipitation and evapotranspiration, storage properties of the layer, characterized by specific yield, and changes in water table position. Assuming the system re-saturates itself every year during snowmelt, we can infer that net water table drop over the growing season will be a result of the water deficit in the summer, or the difference between water input through precipitation and output through evapotranspiration (since runoff is considered negligible in the summer months) and the relative ability of the layer to offset this deficit and store water.

This can be represented by the following equation:

$$\Delta h = \frac{P - ET}{S_y} \qquad [2.7]$$

where h is the depth to the water table below the surface, P is precipitation (mm), ET is evapotranspiration (mm) and S_y is the mean specific yield for the new peat layer. The P-ET term represents the cumulative summer water deficit at the site, and S_y represents the ability of the new peat layer to offset water deficit and maintain the water table high. The depth to the water table will change depending on the balance between these two processes, and therefore reflects the ability or failure of the new peat layer to execute its ultimate function of maintaining a high water table. Consequently, as the new peat moss layer grows over the cutover peat and increases in thickness its ability to store water decrease the range in water table response and it will eventually reach a thickness where all water table fluctuations are contained within the new peat layer itself throughout the summer. At this point, the thickness of the new peat layer would equal or exceed the overall summer induced water table drop. As mentioned in section 1.5, this is the ultimate goal of restoration, since only when the new peat layer is able to act as a fully functional acrotelm will the peatland return to a functioning ecosystem.

CHAPTER 3: ECOHYDROLOGICAL PROPERTIES OF A NEW PEAT LAYER IN A RESTORED PEATLAND

3.1 Introduction

The characteristic hydrological properties of peatland ecosystems aid in creating conditions beneficial for *Sphagnum* moss biomass accumulation and slow peat decay. The balance between these two ecological processes coupled to the water logged conditions of peatland systems leads to organic matter being transferred into the catotelm for the long term accumulation of carbon in the system. However, peat extraction alters this delicate coupling between the hydrology and ecology of peatlands by the removal of the acrotelm layer. Restoration has proven to significantly improve the hydrological conditions of cutover sites (Shantz and Price, 2006) which in turn facilitates the re-establishment of *Sphagnum* vegetation (Price and Whitehead, 2001). Indeed, the ecological and hydrological properties of the newly forming moss and peat layer are expected to return to conditions similar to those of natural sites if *Sphagnum* moss is to recolonize cutover bogs and an acrotelm to redevelop at these sites. Specific conditions are necessary for recolonization. For instance, Price and Whitehead (2001) observed that in a naturally regenerated peatland, sites where *Sphagnum* moss had colonized were characterized by soil moisture > 50%, soil-water pressure > -100 mb and high water table (mean -29 cm), because *Sphagnum* cannot generate capillary forces necessary to extract moisture when those threshold conditions are exceeded (Price, 1997). The authors suggested that this provides a clue to threshold conditions necessary for *Sphagnum* re-establishment. Once those conditions are in place, a new layer can start to develop and eventually a new acrotelm can be regenerated. This chapter will focus on characterizing the ecohydrological properties of a newly formed peat layer at the restored Bois-des-Bel site with the ultimate goal of assessing peatland restoration.

3.2 Peat Layer Ecohydrological Properties

3.2.1 Ecohydrological properties

Bulk density was significantly higher for lower samples (8-12 cm) in comparison to upper (0-4 cm) samples for all sites ($p < 0.05$ for restored and $p < 0.01$ for natural and revegetated sites) (Table 3.1). However, the difference was much smaller between upper and lower samples for the restored site (19 ± 3 kg m^{-3} for upper samples, 24 ± 4 kg m^{-3} for lower samples). Lower samples for the natural and naturally revegetated sites had a bulk density of 43 ± 5 kg m^{-3} and 41 ± 11 kg m^{-3} respectively, almost twice as high as the value for lower samples from the restored site. Values were closer among the sites for the upper samples, ranging from 19 ± 3 kg m^{-3} for the restored site to 24 ± 3 kg m^{-3} for the natural site. Bulk density was also statistically different among the three sites for upper as well as lower samples ($p < 0.05$ for upper and lower samples).

Porosity was significantly higher for upper samples for all sites ($p < 0.001$ for revegetated, $p < 0.01$ for natural and $p < 0.05$ for restored site). Porosity (%) ranged from 97.2 ± 0.3 % for the natural site to 98.5 ± 0.3 % for the restored site for lower samples. Less variance was seen for the upper samples, which ranged from 98.4 ± 0.2 % for the natural site to 98.8 ± 0.2 % for the restored site. Porosity was significantly different among the three sites for upper and well as lower samples ($p < 0.05$ for upper and lower samples).

Table 3.1: Hydrophysical properties of the new peat layer at the three study sites (mean ± standard deviation). S_y is the specific yield; ρ_b bulk density; Ø porosity; and ρ_C the capitula density.

a) Upper (0-4 cm)

	S_y ($m^3\ m^{-3}$)	ρ_b ($kg\ m^{-3}$)	Ø (%)	ρ_C ($\#\ m^{-2}$)
Natural	0.47 ± 0.06	24 ± 3	98.4 ± 0.2	26050 ± 5573
Restored	0.52 ± 0.08	19 ± 3	98.8 ± 0.2	28726 ± 3384
Revegetated	0.54 ± 0.07	20 ± 1	98.7 ± 0.1	27743 ± 3670

b) Lower (8-12 cm)

	S_y ($m^3\ m^{-3}$)	ρ_b ($kg\ m^{-3}$)	Ø (%)	von Post (H)
Natural	0.22 ± 0.01	43 ± 5	97.2 ± 0.3	4
Restored	0.36 ± 0.06	24 ± 4	98.5 ± 0.3	2
Revegetated	0.33 ± 0.08	41 ± 11	97.4 ± 0.7	4-5

Specific yield was higher for upper samples in comparison to lower samples for all three sites ($p < 0.01$ for natural, restored and revegetated sites). Specific yield for the lower samples was higher for the restored and naturally revegetated sites (0.36 ± 0.06 and 0.33 ± 0.08 respectively) in comparison to the natural site (0.22 ± 0.01) ($p < 0.05$). Higher values were observed for the upper samples, with values ranging from 0.47 ± 0.06 for the natural site and 0.54 ± 0.07 for naturally revegetated.

Lower (8-12 cm) samples from the naturally revegetated and natural sites had a higher degree of decomposition (von Post = 4-5) than samples from the restored site (von Post = 2), with restored site samples more loosely packed and with better preserved *Sphagnum* remains than samples from the other two sites. Upper (0-4 cm) *Sphagnum* samples were noted to be mostly between undecomposed to almost undecomposed for the three sites. Samples from the natural and restored sites showed higher presence of small leaves and fragmented branches than samples from the naturally revegetated site. The restored and natural sites also demonstrated the presence of *Polytrichum* mosses, which appeared to be higher for the restored site samples.

Sphagnum rubellum capitula density (ρ_C) for upper *Sphagnum* samples was significantly higher ($p < 0.05$) for the restored peatland (28726 ± 3384 # m^{-2}) when compared to the natural site (26050 ± 5573 # m^{-2}).

3.2.2 Soil-moisture retention

Soil moisture retention characteristics for lower samples (8-12 cm below the surface) differed greatly for the restored site in comparison to the two other sites (Figure 3.1). Residual moisture content at 200 mb (-200 cm in soil tension) (θ_r) was significantly lower ($p < 0.05$) for the restored site (13.7 ± 1.3 %) in comparison to the other two sites (Figure 3.1 and Table 3.2). VWC at 100 mb (θ_{100}) was significantly higher ($p < 0.05$) for the natural site (31.5 ± 4.2 %) than for the restored site (15.5 ± 1.7 %). While the overall shape of the curves is similar, the restored site had a lower VWC at any given pressure when compared to the other two sites.

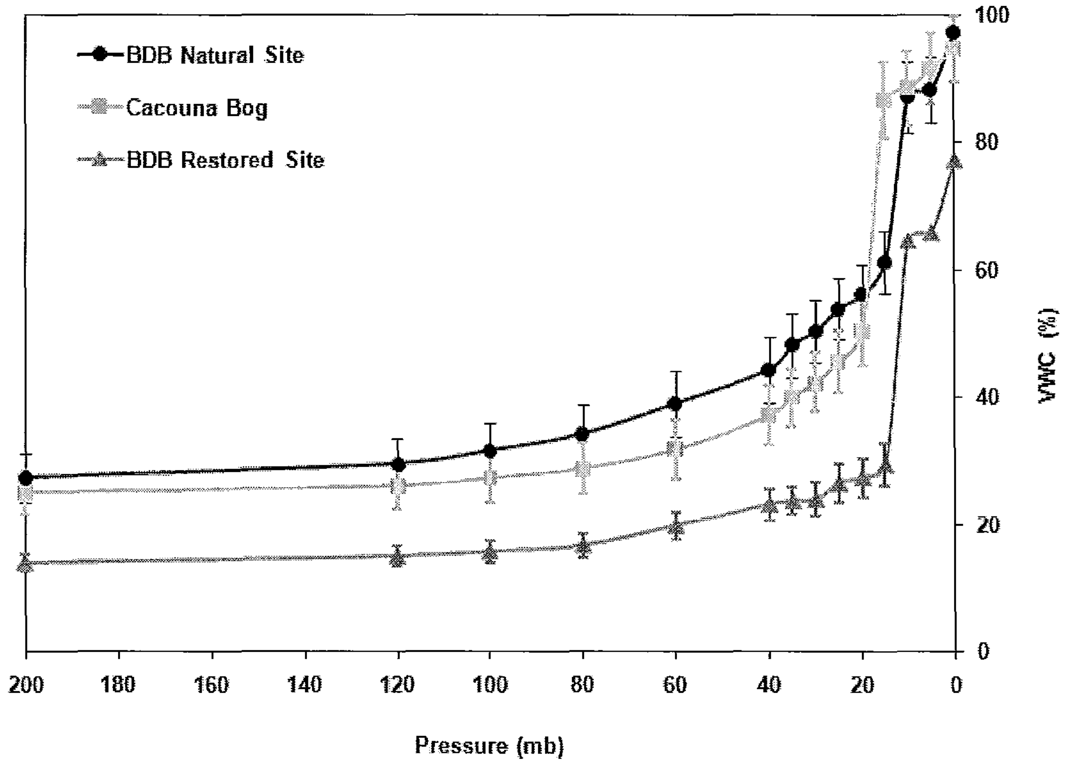

Figure 3.1: Soil-moisture retention curves for lower samples (8-12 cm) at the natural, naturally revegetated and restored sites.

Table 3.2: Summary of moisture retention properties for the lower (8-12 cm) samples (mean ± standard deviation).

	$\theta_{100\ mb}$	θ_r
Natural	31.5 ± 4.2	27.2 ± 3.9
Restored	15.5 ± 1.7	13.7 ± 1.3
Revegetated	27.2 ± 3.9	24.9 ± 3.5

Moisture retention curves for the upper (0-4 cm) samples showed a similar pattern to the lower samples (Figure 3.2). In general, at a given pressure, VWC followed the trend: natural > naturally revegetated > restored. Differences in VWC among the sites were much smaller than what was observed for the lower samples. The naturally revegetated site had slightly higher values for lower pressures than the restored site, but at higher tension VWC values were very similar for these two sites. Residual moisture content at 200 mb for the naturally revegetated and restored sites was 11.1 ± 3.0 % and 10.0 ± 4.5 % respectively, and slightly higher retention was seen for the natural site at residual moisture content (15.5 ± 2.3 %); however this difference was not significant. The same pattern was observed for VWC at 100 mb, with differences among sites also not significant.

Differences between the upper and lower moisture retention properties are small for the restored site. Both residual moisture content (13.7 ± 1.3% for the lower samples and 10.0 ± 4.5 % for the upper samples) and moisture content at 100 mb (15.5 ± 1.7% for lower and 12.2 ± 4.5 % for upper samples) for this site were not significantly different. This is not the case for the other two sites, which had lower residual moisture content and lower moisture content at 100 mb for the upper samples in comparison to the lower samples ($p < 0.001$ for revegetated site and $p < 0.05$ for natural site for each of residual and 100 mb moisture content).

Pore size distribution was calculated based on the capillary rise equation and relates to tension. The proportion of water lost was higher (52% of total) for the 60-74 μm pore radius interval for the revegetated site and higher for the 75-99 μm pore radius interval for the natural (37% of total) and restored (56% of total) sites for the lower samples. For the upper samples, all three sites had the highest proportion of water lost over the entire run for the 100-148 μm pore radius interval.

Cumulative water loss with increasing pressure was consequently much more similar among the three sites for the upper samples in comparison to the lower samples (Figures 3.3 and 3.4) due to less distinguishable differences in retention curves between upper samples. Slightly more gradual loss was observed for the lower samples in comparison to the upper samples, which appeared to drain quickly and reach values closer to the residual moisture content faster than lower samples. For the lower samples (Figure 3.4), at any given pressure lower than 100 mb the restored site had lost more water than the natural site.

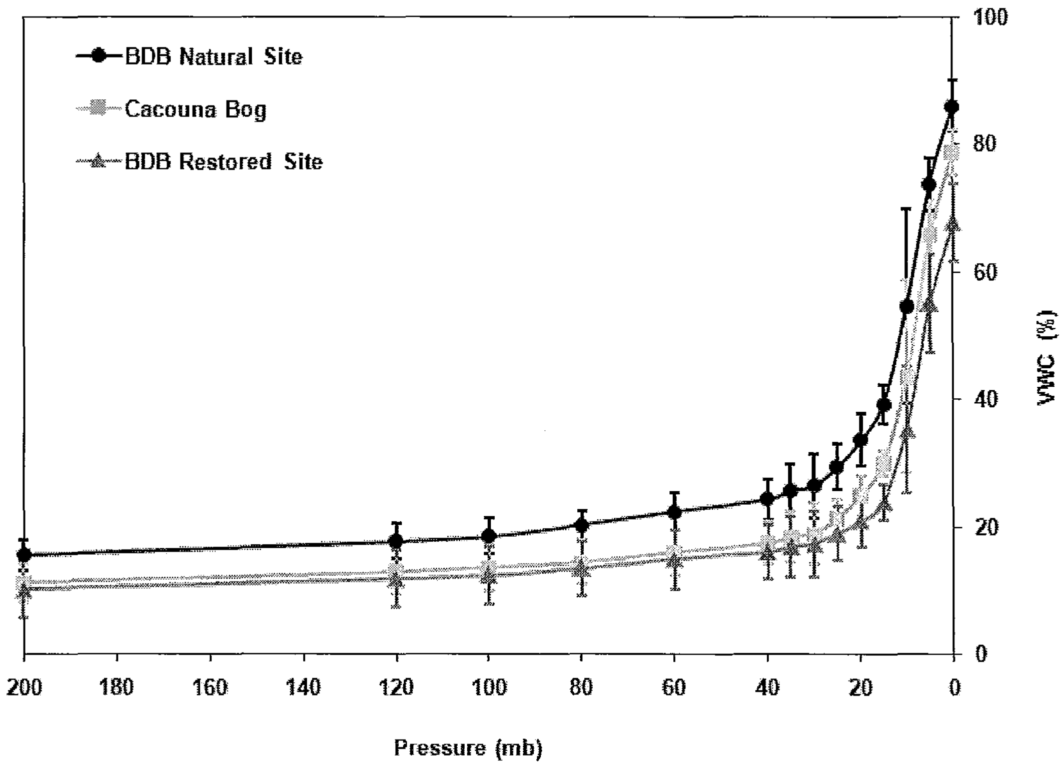

Figure 3.2: Soil-moisture retention curves for the upper samples (0-4 cm) at the natural, naturally revegetated and restored sites.

Table 3.3: Summary of moisture retention properties for the upper (0-4 cm) samples (mean ± standard deviation).

	$\theta_{100\ mb}$	θ_r
Natural	18.3 ± 2.8	15.5 ± 2.3
Restored	12.2 ± 4.5	10.0 ± 4.5
Revegetated	13.4 ± 3.6	11.1 ± 3.0

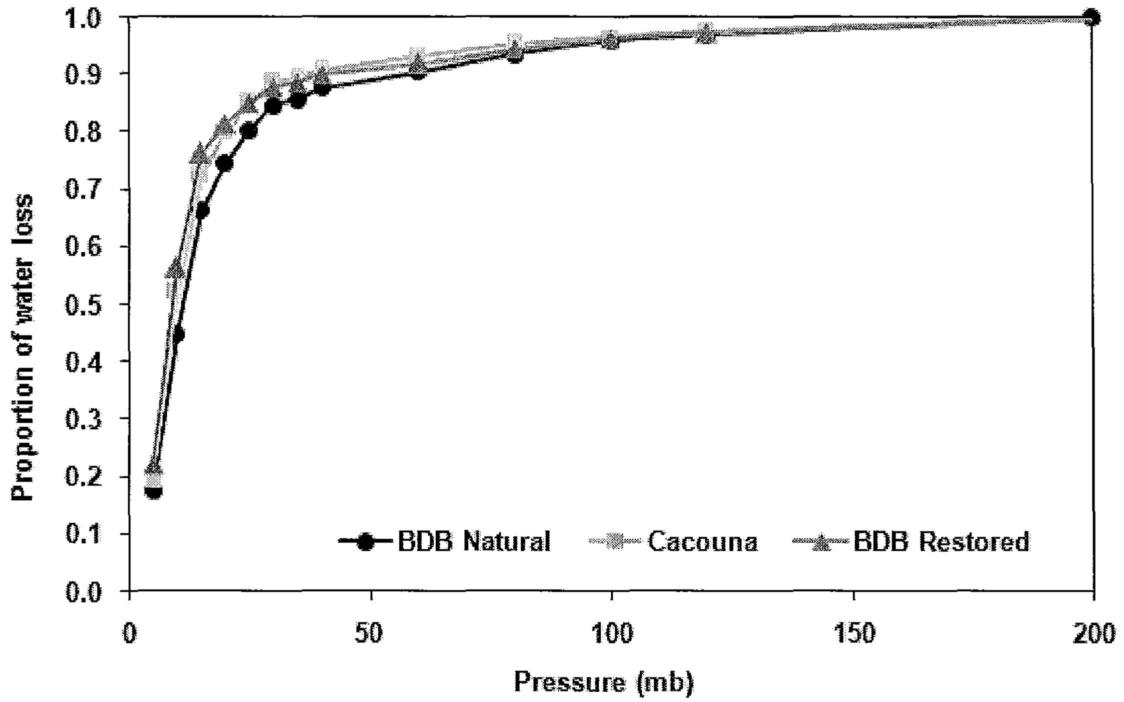

Figure 3.3: Cumulative proportion of water loss with increasing pressure for upper (0-4 cm depth) samples.

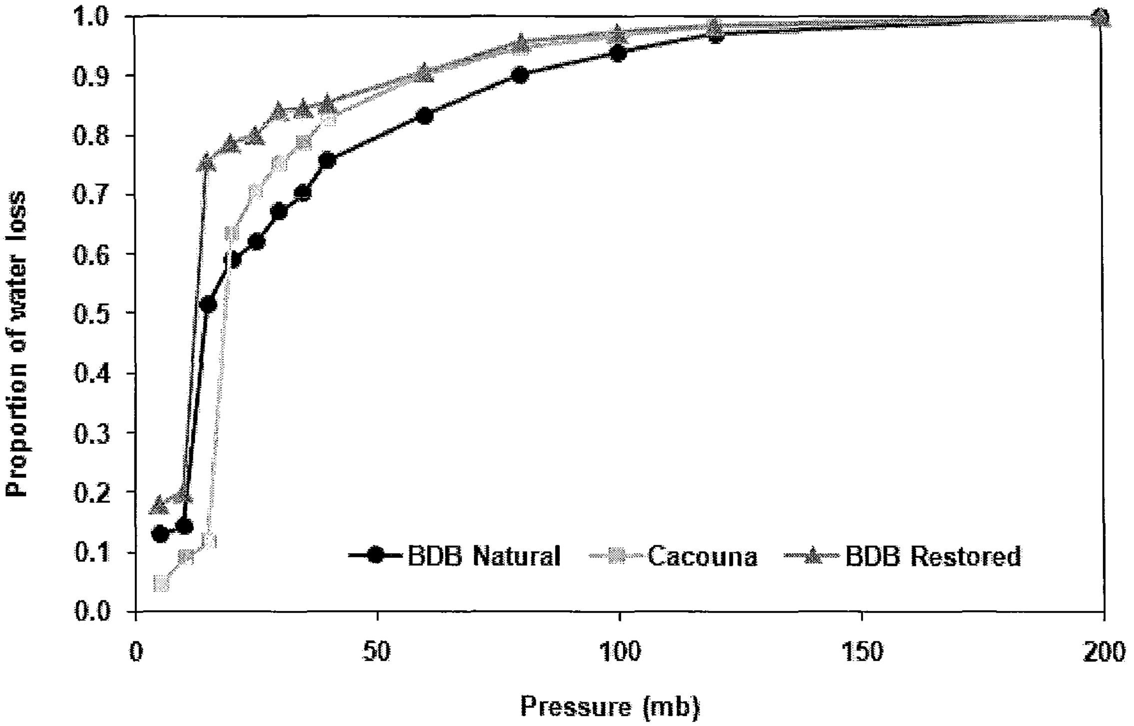

Figure 3.4: Cumulative proportion of water loss with increasing pressure for lower (8-12 cm depth) samples.

Modeled moisture retention curves using the RETC code (van Genuchten, 1991) showed a good fit to observed data (Figures 3.5 and 3.6). Modeled curves fit the observed data with an R^2 value of 0.96 or better. Parameter values are reported in Table 3.4.

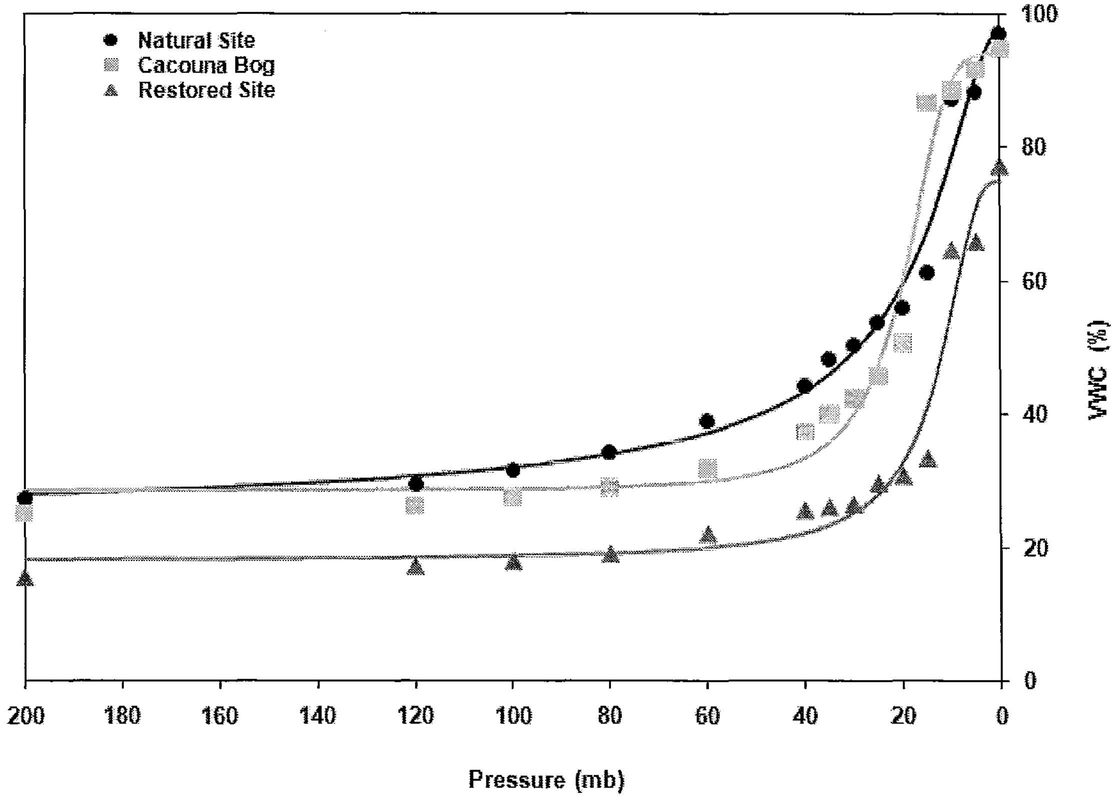

Figure 3.5: RETC model fit to observed moisture retention data for the lower samples. Actual data is represented by points; modeled data is represented by lines.

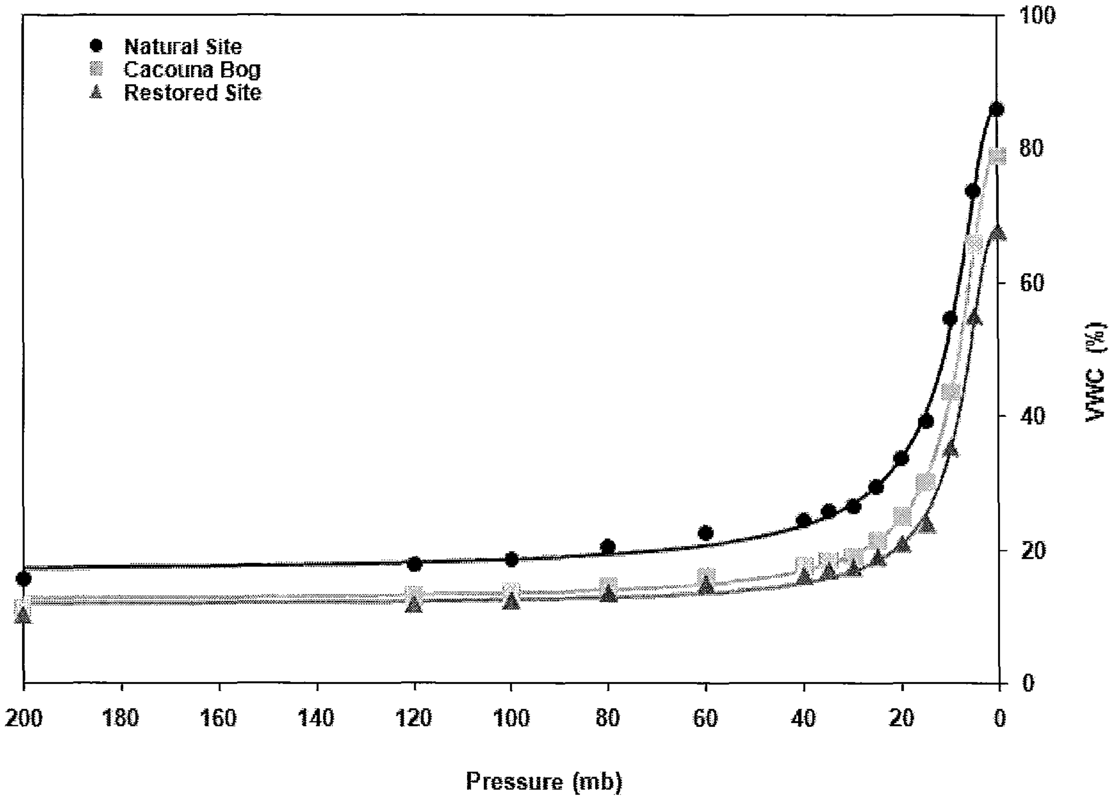

Figure 3.6: RETC model fit to observed moisture retention data for the upper samples. Actual data is represented by points; modeled data is represented by lines.

Table 3.4: RETC model curve fitting parameters (α, n and m) for upper and lower samples from all three study sites.

	α	n	m	R^2
Lower (8-12 cm)				
Natural	0.09	2.01	0.50	97.8
Restored	0.09	2.96	0.66	95.9
Revegetated	0.06	4.18	0.76	97.3
Upper (0-4 cm)				
Natural	0.14	2.32	0.57	99.8
Restored	0.16	2.51	0.60	99.7
Revegetated	0.15	2.53	0.60	99.9

3.2.3 Volumetric water content survey

VWC was measured at 500 locations throughout the restored site two times in 2007: on a wet day in July (Figure 3.7) and on a dry day in August (Figure 3.8). Mean VWC was 64.4 ± 24.6 % and 50.7 ± 26.4%, with coefficients of variance of 0.38 and 0.52 for the wet and dry days respectively. Mean VWC value for the July survey was significantly higher ($p < 0.001$) than mean VWC for August. 71 % of locations sampled in July had a VWC value > 50%, while 44 % of the locations sampled in August had a VWC value > 50%. Based on the site specific moisture retention curves developed for the restored site, the VWC value corresponding to a tension value of - 100 mb was determined. This was incorporated to the moisture survey maps to determine which locations were stressed from a hydrological perspective at the site. In July (wet day), only three locations had a tension value ≥ - 100 mb, which corresponded to ~1% of the site. In August (dry day), 12 locations had a tension value ≥ - 100 mb, approximately 3% of the site.

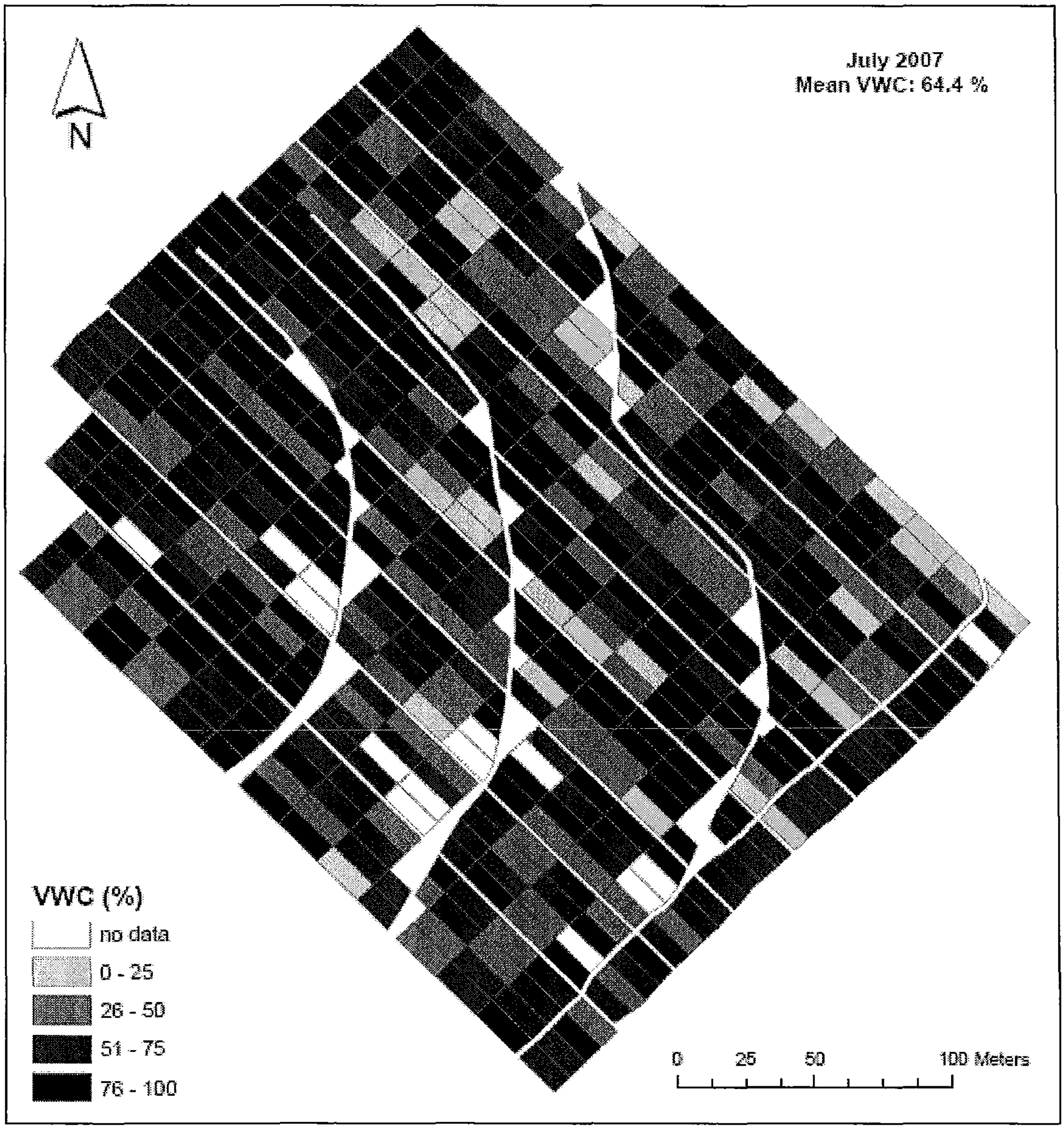

Figure 3.7: Point measurements of VWC at the restored site, July 2007.

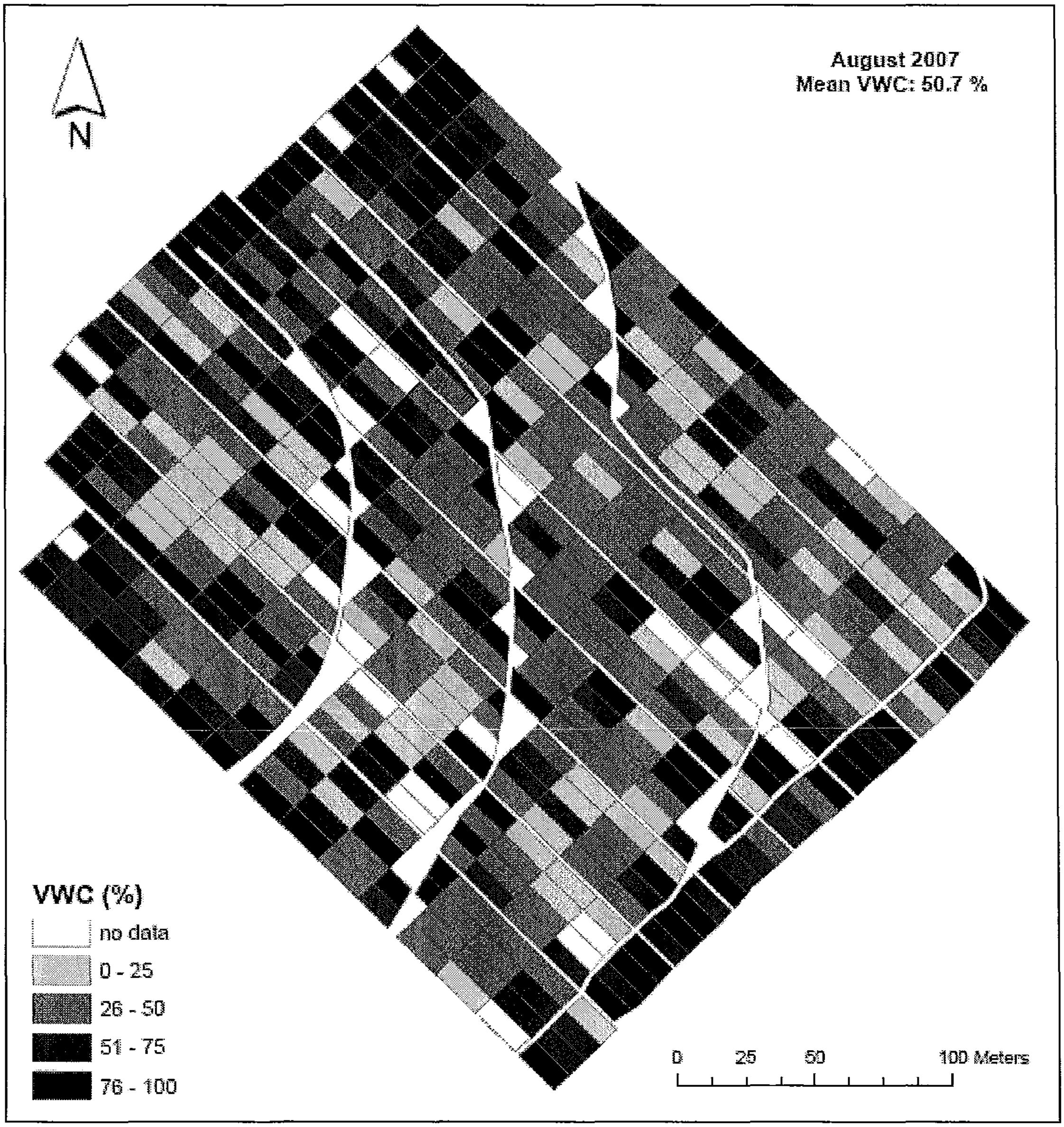

Figure 3.8: Point measurements of VWC at the restored site, August 2007.

3.3 Discussion

3.3.1 Ecohydrological properties and moisture relations

The ecohydrological properties of the new peat layer in a restored site were found to differ from those of an adjacent natural peatland and of a naturally revegetated peatland. This difference was especially noticeable in the lower 8-12 cm layer. The lower bulk density of the lower layer at the restored site in comparison to the other two sites (24 ± 4 versus 43 ± 5 and 41 ± 11 for the natural and the revegetated sites respectively) is an indication that the degree of compaction and decay in this layer is less than at the same depth in the other two sites. This is reinforced by the less decomposed nature according to the von Post scale and higher porosity of the lower samples from the restored site in comparison to the natural and revegetated sites. Despite successful recolonization and growth of *Sphagnum* moss at the restored site demonstrated by the return of properties that resemble those of a natural site in the 0-4 cm layer, properties at the 8-12 cm layer differ for the restored site. It is likely that the time post-restoration has not yet been sufficient to allow for some important processes in peatland development such as collapse of plant structure and compaction to be restored to their normal functions. Specific yield values obtained are within reported values for the acrotelm layer in natural peatlands (e.g., 0.5 to 0.1; Price, 1992). S_y was lowest for the 8-12 cm samples from the natural site (0.22 ± 0.01) when compared to the restored site (0.36 ± 0.06) and revegetated site (0.33 ± 0.08), and also lower for the natural site than for the other two sites for the 0-4 cm samples. Van Seters and Price (2001) found similar S_y values for a natural peatland (within 5 km of Bois-des-Bel) and the Cacouna bog (revegetated site) (0.11 and 0.08

respectively) for the 5-10 cm section. To our knowledge there are no other values reported in the literature for S_y for a restored peatland. All sites had higher S_y for the upper layer in comparison to the lower layer. A high S_y in the upper layers of the acrotelm is essential to limit water table drawdown during dry periods and flooding during wet periods. This higher S_y value is expected and was observed for upper samples in relation to lower samples, as S_y depends on porosity and pore size, which are known to decrease with depth in the peat profile (Rizzuti et al., 2004). Capitula density was lower for the natural site, and highest for the restored site, a finding that agrees with what Waddington et al. (2003) observed, as those authors found higher capitula density at restored plots compared to natural plots. The authors suggested that this could be a mechanism developed to cope with harsher hydrological conditions initially present at restored peatlands in comparison to natural sites, allowing *Sphagnum* mats to keep high moisture contents due to tighter arrangement, avoiding desiccation which could have resulted in high production. However, this suggestion is the opposite that observed in the moisture retention laboratory experiment.

Despite higher capitula density for the restored site, moisture retention was not higher for this site in comparison to a natural and a naturally revegetated site. Samples from the restored site appear to hold less water at any given pressure, especially for the depth 8-12 cm. Furthermore, this site showed no significant differences in moisture retention properties between upper and lower samples, whereas the other two sites did. Visual observation of lower restored samples in comparison to naturally revegetated and natural sites revealed that peat fibers in restored samples were less compacted and more preserved. The restored site had a lower VWC at any given pressure when compared to the other two sites. This suggests that *Sphagnum rubellum* in a natural peatland is able to hold onto more moisture under increasing soil-tension than the same species growing in a restored site, likely due to higher bulk density and relatively more decomposed state of *Sphagnum* samples (see von Post in Table 3.1 b). Moisture retention increases with higher bulk density (Boelter, 1968), which can help explain the differences observed between the moisture retention curves for the natural and restored sites, especially at the 8-12 cm depth. In this manner, as observed for the lower samples, the proportion of water lost at any given pressure lower than 100 mb was higher for the restored site than for the natural site. Soil water retention model RETC provided a good fit to the moisture retention data obtained in the laboratory and was used to describe the observed moisture retention data. It had been suggested that RETC could be a useful way of easily predicting hydraulic conductivity properties from moisture retention data (van Genuchten et al., 1991), which could provide much needed insight into understanding water flux dynamics in the living moss layer (Price et al., 2008). Price et al. (2008) tested this hypothesis and determined that unsaturated hydraulic conductivity (K) could not be accurately modeled based on moisture retention curves alone and suggested that the theoretical pore-size distribution model (Mualem, 1976) does not represent well the structure of mosses. Nonetheless, observed moisture retention data in this study and modeled curves for the sites provide additional information that may not necessarily be as easily measured in the field with respect to the water storage properties of the *Sphagnum* moss layer.

The return of the ecohydrological properties of the *Sphagnum* moss layer is essential for restoration of a peatland ecosystem; however specific hydrological conditions are necessary for the re-establishment of this peatland species. Price (1997) determined that when soil-water tension drops below -100 mb, the moss cannot regenerate capillary forces necessary to extract moisture from the cutover peat. Soil-water pressure was suggested to be a better indicator of soil suitability and moisture availability than water table or moisture content as it is more sensitive and directly controls water supply to *Sphagnum* (Price and Whitehead, 2001). Mean VWC was 64.4 ± 24.6 % and 50.7 ± 26.4% for July and August respectively at the restored site. This is comparable to values obtained in a moisture survey conducted by Price and Whitehead (2001) at the Cacouna bog. Those authors found at the naturally revegetated bog that average VWC for the 0-4 cm layer in July and August in 1998 was 55.4 ± 11.5 % and 49.3 ± 14.8 % respectively. Furthermore, approximately 83% of *Sphagnum* that recolonized at this bog did so in locations where VWC > 50% and where pressure was maintained above -100 mb (Price and Whitehead, 2001). Local morphology was found to be a strong indicator of hydrological conditions and consequently related to moss colonization in the Cacouna bog as the trench-baulk landscape associated with block harvested peatlands appears to provide for some localized suitable hydrological conditions for moss re-establishment in low lying areas within the landscape. Lavoie and Rochefort (1996) noted that spontaneous regeneration of *Sphagnum* in block-cut peatlands was possible; however this was only seen in areas where hydrological conditions appeared to be more suitable. Spontaneous *Sphagnum* regeneration is much rarer in vacuum extracted peatlands, where low water table level throughout the site prevents *Sphagnum* recolonization (Lavoie et al., 2003). Vacuum harvested peatlands such as Bois-des-Bel are characterized by a much flatter and regular surface, creating hydrological conditions that are typically less variable but worse (drier) than block-cut sites throughout the abandonment cutover peat surface (Price et al., 2003). They usually require active restoration to allow the return of the more suitable hydrological conditions needed for *Sphagnum* survival, by blocking old drainage ditches and creating bunds to retain water on site (Rochefort et al., 2003; Price et al., 2003). A field survey of abandoned peatlands revealed that *Sphagnum* was consistently found in association with *Polytrichum strictum* (Groeneveld et al., 2007). Groeneveld et al. (2007) showed that *Polytrichum* can be used as a nurse plant in restored peatlands, facilitating *Sphagnum* growth, as their carpets were able to keep *Sphagnum* more humid, provided less variable temperatures and were able to trap more seeds than bare peat.

Boelter (1968) showed that when capillary pressure was equivalent to a water table depth of - 100 cm (-100 mb), water content in undecomposed *Sphagnum* carpets was 10%, compared with 70% for decomposed peat. Hayward and Clymo (1982) showed that the moisture content in living and poorly decomposed *Sphagnum* when the water table was lowered to -100 cm was 10-20%, and that drainage of porous hyaline cells (responsible for storage and transport of water) occurred below this pressure. In this study, the moisture retention curves obtained for the restored site were used to determine which locations exceeded the -100 mb threshold based on the moisture content survey at the Bois-des-Bel restored site, which corresponded to only 1% and 3% of the site for July and

August respectively (below average of 14% in VWC). This demonstrates that tension and moisture conditions in the upper living layer appear hydrologically favorable, even in a dry period. Furthermore, 71 % of locations sampled in July had a VWC value > 50%, while 44 % of the locations sampled in August had a VWC value > 50%, which demonstrates that not only is the site meeting the 100 mb threshold but it is also maintaining a high moisture content on average at the site, even during a dry period. Maintaining high moisture content in the living moss layer is essential as water stored in the unsaturated zone controls water availability to the surface (Price, 1997). It is no surprise that *Sphagnum* moss was found predominantly in wetter areas at Bois-des-Bel (Petrone et al., 2004). Petrone et al. (2004) also found that moisture conditions at Bois-des-Bel were better characterized by larger areal coverage rather than high resolution sampling, further suggesting that the interaction of vegetation and soil hydrology must be quantified at scales pertinent to that of the restoration of an entire ecosystem. In this manner, a complete assessment of peatland restoration should involve assessing the entire ecosystem. Active restoration of the Bois-des-Bel peatland appears to have contributed to the recovery of hydrological conditions necessary for *Sphagnum* re-establishment to the site. Shantz and Price (2006) showed that in the initial 3 years post-restoration, before a significant moss layer had developed, runoff decreased, mean water table position increased and volumetric moisture content in the upper 5 cm layer increased and spatially averaged θ and ψ were maintained above 50% and -100 mb respectively. Spatial and temporal variations in these two parameters were reduced in comparison to the unrestored site. The authors also noted that these improvements occurred despite the increase in water table fluctuation in comparison to the non-restored portion of the peatland, which may be associated with the return of vascular vegetation at the site. This study shows that despite successful re-introduction of *Sphagnum* to the restored site and improved hydrological conditions, the properties of the newly formed moss layer still differ from those of a natural site, especially for the lower 8-12 cm depth of the layer. It is possible that an adaptive strategy of *S. rubellum* is to allocate more of its sequestered CO_2 to vertical growth in harsh hydrological conditions such as a restored peatland. Turetsky et al. (2008) recently determined that different *Sphagnum* species allocate carbohydrates differently as an adaptive strategy for carbon accumulation in differing micro-environments. Given the broad range of moisture retention characteristics of *S. rubellum,* carbohydrate allocation of recently restored *Sphagnum* warrants further investigation. Also, as previously observed in the Cacouna bog, despite recolonization of *Sphagnum* in some areas of the bog as well as similar S_y and mean water table position at the naturally regenerated portion of the block cut bog and a nearby natural site, acrotelm thickness, water table fluctuations and maximum water table depth differed remarkably between the two sites (Van Seters and Price, 2001; Price and Whitehead, 2001). It is evident that the naturally vegetated bog, even 30 years post-abandonment, had not yet developed an acrotelm able to stabilize water table fluctuations, which suggests that the presence of localized *Sphagnum* cushions (only about 10% of the site) with high S_y has limited effect on water storage (Price and Whitehead, 2001). Rather, the development of a continuous moss layer able to stabilize and contain water table fluctuations is required to restore the diplotelmic nature of peatlands. The next chapter will examine whether an acrotelm layer is already present at the restored Bois-des-Bel peatland.

3.3.2 Conclusion

Results demonstrate that the new moss layer at a restored site eight years post-restoration differs from the moss layer at a natural site with respect to its moisture retention characteristics and ecohydrological properties, especially at the lower depth of the moss layer. Moisture retention capacity of the new moss layer at the restored site will likely increase with time, as decomposition and compaction in this layer increase and carbohydrate allocation strategies potentially change with time. The return of a continuous moss layer at Bois-des-Bel is encouraging; however, the ecohydrological balance of the site must be restored in order for peatland restoration to be successful. True peatland restoration will not be achieved until the diplotelmic nature of the peatland is restored and the newly forming layer becomes an acrotelm, able to contain water table fluctuations and return the peat accumulating function of the peatland.

CHAPTER 4: AN ECOHYDROLOGICAL APPROACH TO ASSESSING PEATLAND ECOSYSTEM RESTORATION

4.1 Introduction

Cutover peatlands lose their net carbon sink function due to the removal of the vegetated moss layer and due to enhanced decomposition of older catotelmic peat exposed at the surface. The hydrology of these sites is highly altered (Van Seters and Price, 2001) which further imposes problems for *Sphagnum* moss re-establishment and survival in these systems (McNeil and Waddington, 2003). Restoration techniques have the potential to improve the hydrological conditions necessary for moss establishment at cutover sites mainly by increasing moisture content and water table level at the site (Shantz and Price, 2006). For instance, blocking old drainage ditches is an effective way of aiding in the creation of suitable hydrological conditions for *Sphagnum* species re-introduction (Girard, 2000). However, the hydrophysical characteristics of the old exposed peat on the cutover surface often present a problem in maintaining constant water levels (Price and Whitehead, 2001). This is necessary for *Sphagnum* growth and for the long term accumulation of carbon in peat, as aerobic conditions created by lower water table positions and larger fluctuations facilitate decomposition rather than aiding in the accumulation of organic matter as peat. Shantz and Price (2006) found that despite higher water table positions, water table fluctuations were higher in the years immediately following peatland restoration leading to enhanced decomposition (Waddington and Day, 2007). McNeil and Waddington (2003) suggested that water table fluctuations will likely remain problematic for restoration efforts until a suitably thick acrotelm regenerates. However, little research has focused on acrotelm development post-restoration and on determining the thickness of a new peat layer necessary to stabilize water table fluctuations at restored peatlands. Successful peatland restoration depends on the development of a new acrotelm layer which has the capacity for hydrological self-regulation (Waddington and Day, 2007) and the ability to maintain high and stable water table levels throughout the year (McNeil and Waddington, 2003). This chapter focuses on examining the growth of a new peat layer in a cutover peatland after restoration and predicting the thickness required for this layer to become a self-regulating acrotelm through the use of a simple hydrologic model and Clymo's (1984) acrotelm growth model.

4.2 New Peat Layer Development

4.2.1 New peat layer thickness

In 2003, four years post restoration, the thickness of the new peat layer was greater in the western portion of zone 3 and near the boardwalk in zones 3 and 4 (See Figures 2.1 and 2.3 for zones at Bois-des-Bel). The mean thickness of the new layer was 2.3 ± 1.7 cm, with a maximum thickness of 11.4 cm (Figure 4.1). In 2005, thickness was greater in the northern and southern portions of zones 2 and 3. The mean thickness at the site increased to 5.3 ± 4.6 cm in 2005, with a maximum thickness of 20 cm (Figure 4.2). In 2007, the thickness of the new peat layer was more evenly distributed throughout the site, with a patch of lower thickness observed in the northernmost part of zone 3. Mean thickness in 2007, eight years post-restoration, was 13.6 ± 6.5 cm, with a maximum recorded thickness of 34 cm (Figure 4.3). The minimum thickness was 0 cm for all years.

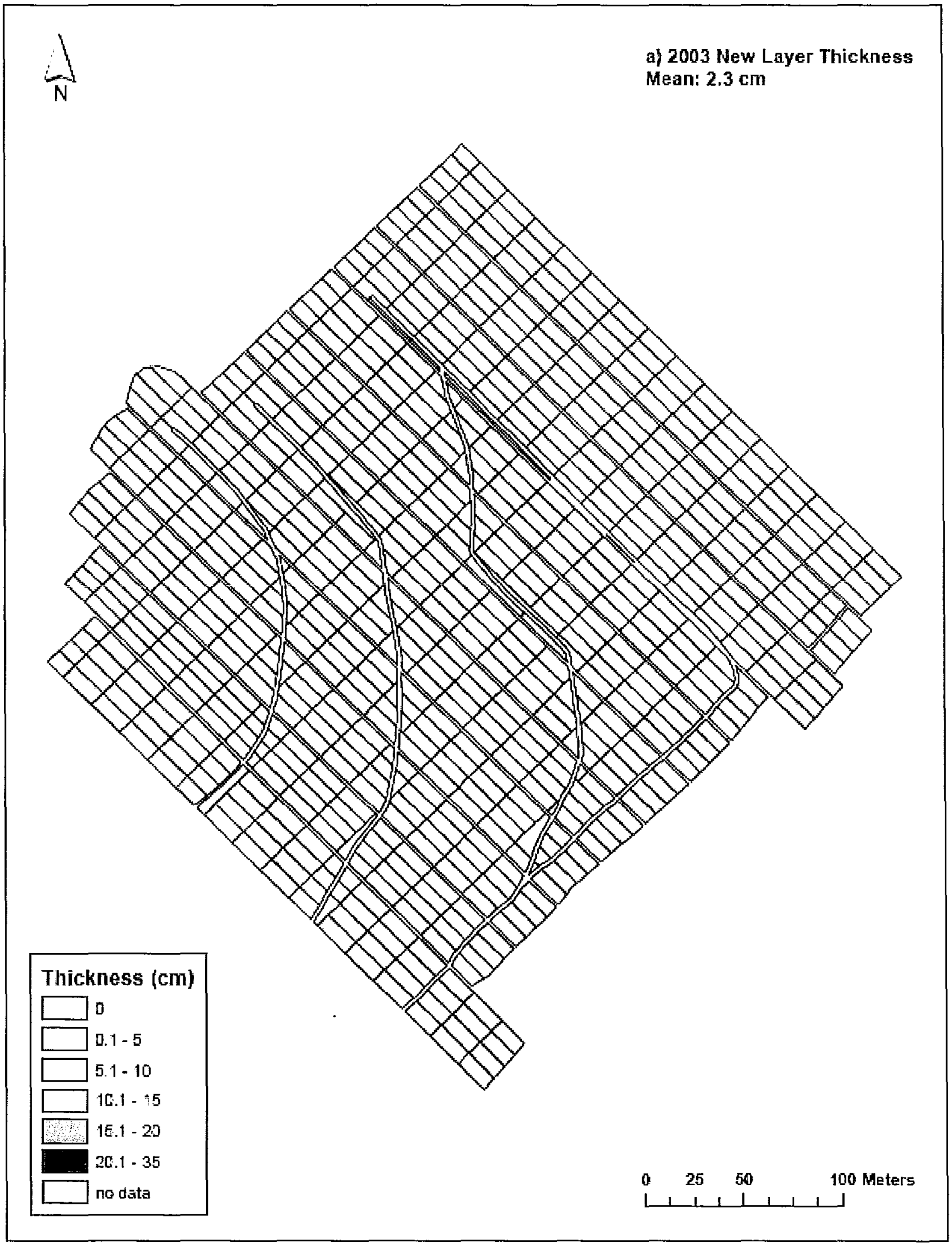

Figure 4.1: Thickness of the new peat layer at the restored and cutover portions of Bois-des-Bel in 2003 (4 years post-restoration).

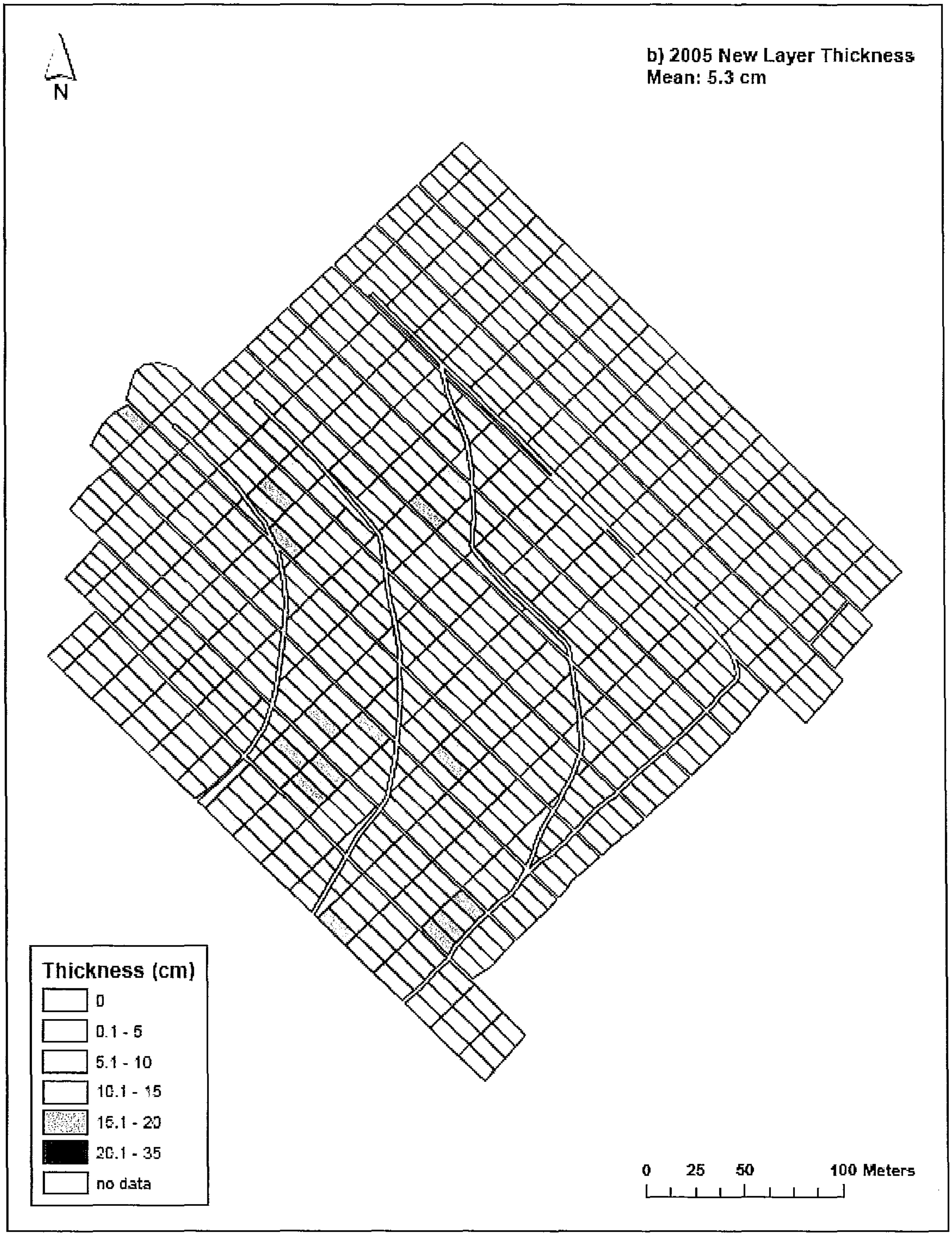

Figure 4.2: Thickness of the new peat layer at the restored and cutover portions of Bois-des-Bel in 2005 (6 years post-restoration).

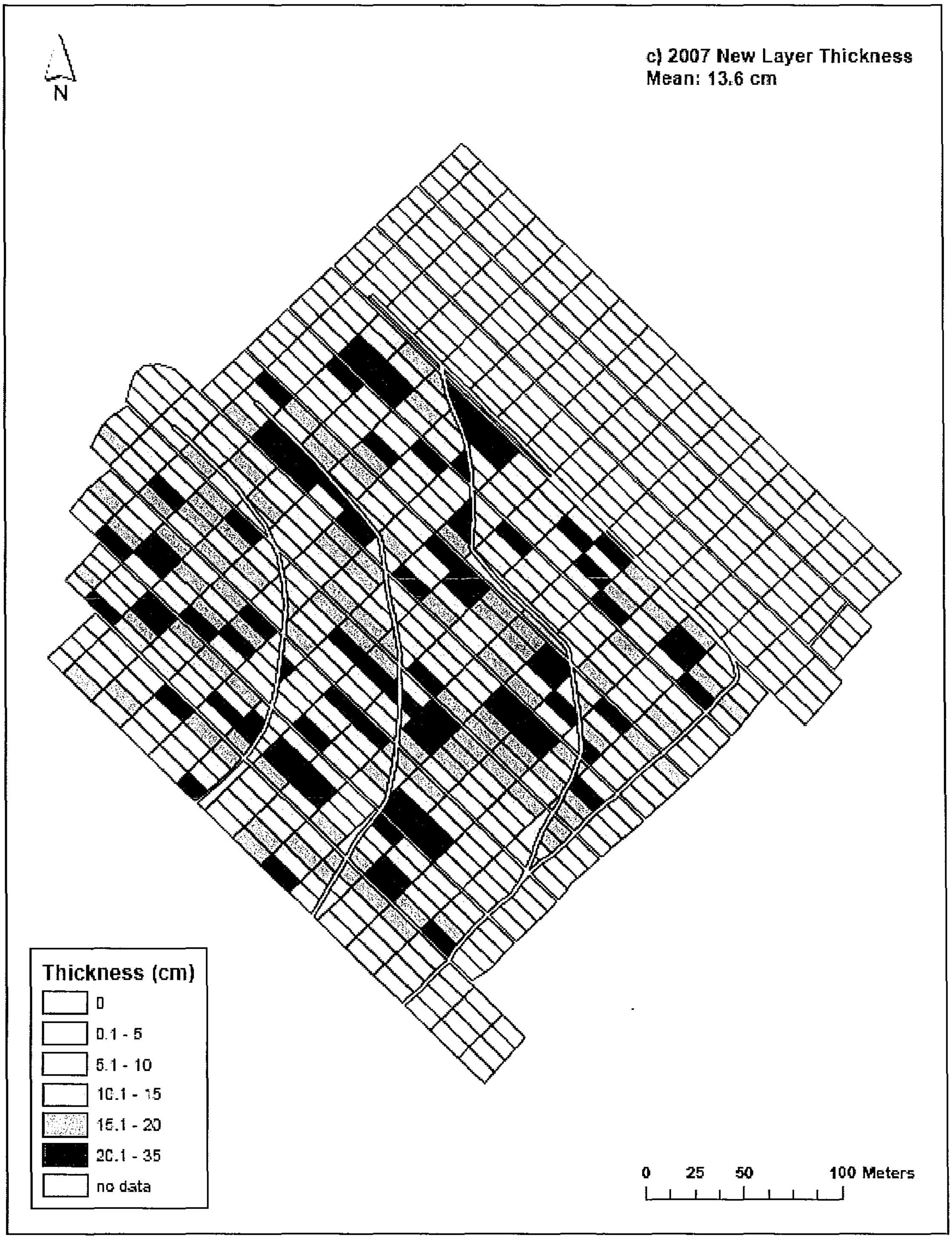

Figure 4.3: Thickness of the new peat layer at the restored and cutover portions of Bois-des-Bel in 2007 (8 years post-restoration).

There was higher spatial variation in the thickness of the new layer throughout the site in 2003 and 2005, with a significant difference in mean thickness between the 4 zones at the restored portion of the site for those years ($p < 0.001$ for each year). Mean thickness ranged from 1.8 cm (zone 1) to 2.6 cm (zone 4) in 2003; and from 3.3 cm (zone 4) to 7.5 cm (zone 2) in 2005. However, there were no significant differences in new layer thickness between the four zones in 2007 with mean thickness ranging from 13.4 cm (zone 4) to 13.7 cm (zone 3) (Figure 4.4).

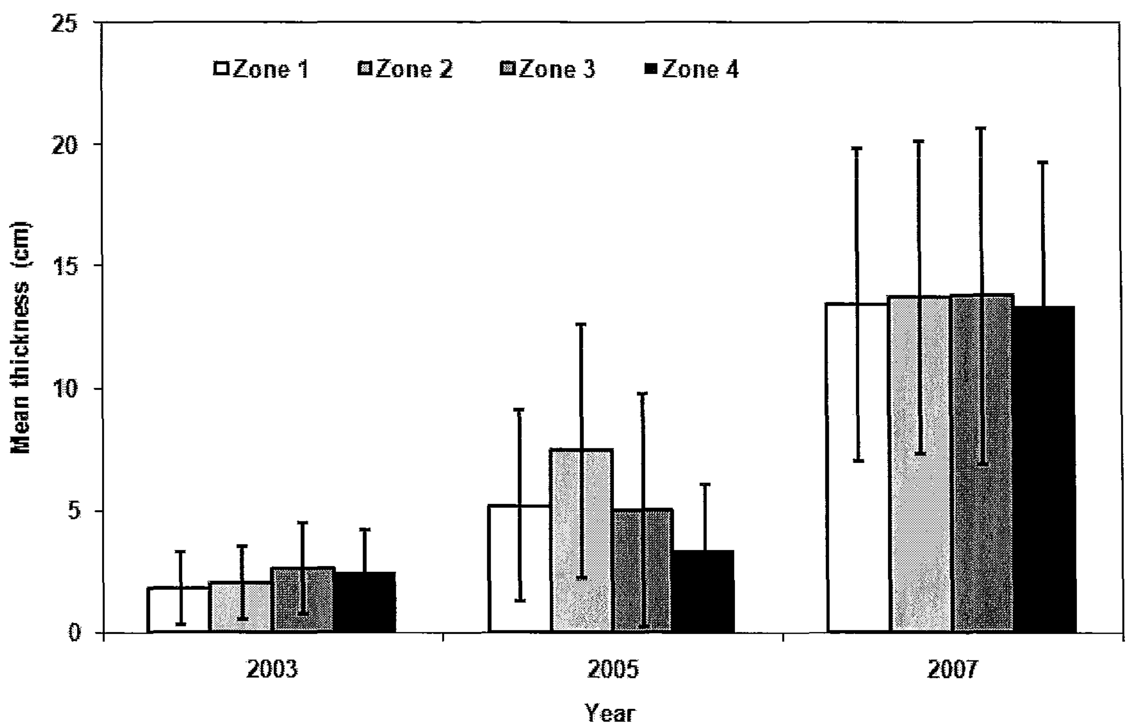

Figure 4.4: Mean thickness of the new peat layer at the restored site in 2003, 2005 and 2007 by zones. See figure 2.1 and 2.3 for the location of the different zones in Bois-des-Bel.

For the cutover (unrestored) portion of the peatland, the mean values for 2003, 2005 and 2007 were significantly lower than the mean values for restored portion of the site for each year ($p < 0.001$ for all years), with mean thicknesses of 0.22 ± 0.69 cm, 0 cm and 0.81 ± 1.22 cm for the three years respectively.

4.2.2 Accumulated biomass and net primary productivity

Accumulated new peat layer biomass at the restored site increased over the six years post restoration (Figure 4.5), ranging from 47 ± 43 g m^{-2} in 2000 to 1692 ± 932 g m^{-2} in 2005. All four vegetation functional types (*Sphagnum* moss, other mosses, ericaceous shrubs and other vascular plants) showed the same pattern of increasing biomass with time post-restoration. In 2000, other mosses, ericaceous, other vascular and *Sphagnum* mosses constituted 53%, 26%, 7% and 14% of the total biomass respectively. In general, the proportion of *Sphagnum* biomass to total biomass increased over the years, as did other vascular plants. Other mosses showed a slight decrease in proportion with respect to total biomass over the six years. Ericaceous vegetation also showed a decrease in relative proportion over the years. In 2005, total biomass was more evenly distributed among the four functional types, with *Sphagnum,* other mosses, ericaceous shrubs and other vascular accounting for 27%, 34%, 10% and 29% of the total accumulated biomass respectively. Net primary production ranged from 79 g m^{-2} yr^{-1} (2002-2003) to 680 g m^{-2} yr^{-1} (2004-2005), with a mean value over the last four one year periods of 379 ± 225 g m^{-2} yr^{-1} (Table 4.1). Overall NPP generally increased over time, except in 2003 when the value decreased relative to the previous year. NPP six years post-restoration ranged from 66 ±

184 g m^{-2} yr^{-1} for ericaceous to 332 ± 694 g m^{-2} yr^{-1} for other vascular. *Sphagnum* NPP six years post-restoration was 179 ± 515 g m^{-2} yr^{-1}.

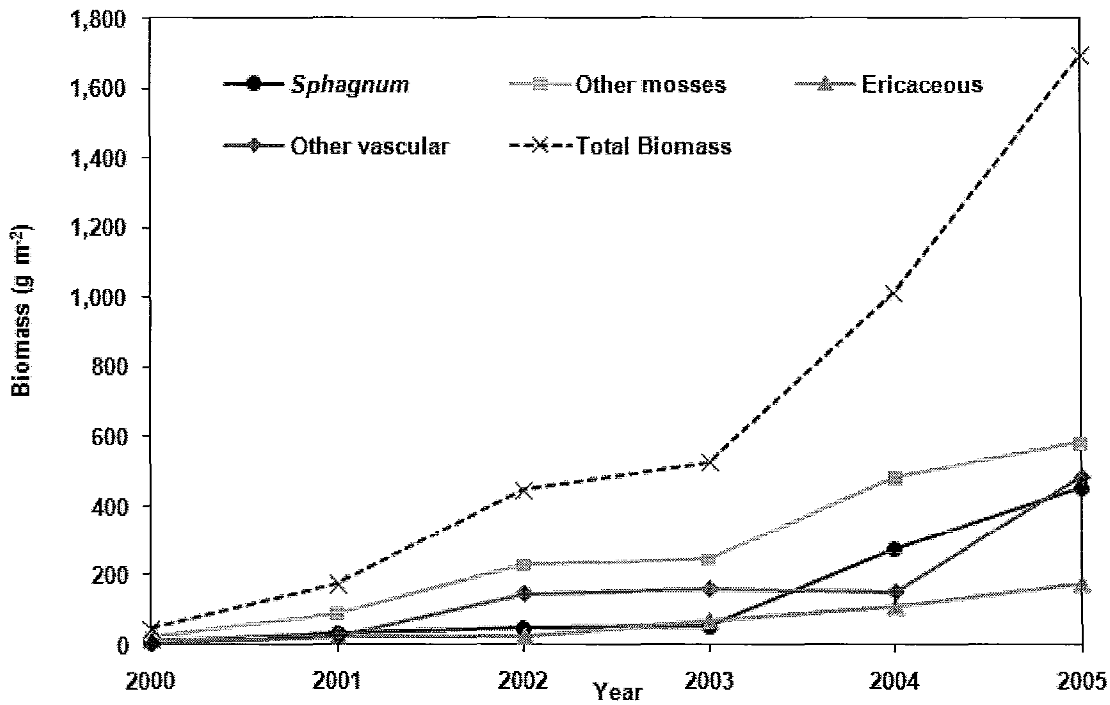

Figure 4.5: Biomass accumulation for each year post restoration for *Sphagnum* moss, other mosses, ericaceous shrubs and other vascular plants in the new peat layer. The dotted line represents total biomass accumulation.

Table 4.1: NPP (g m^{-2} yr^{-1}) by vegetation functional types for the restored site from 2000 to 2005.

	2000	**2001**	**2002**	**2003**	**2004**	**2005**
Sphagnum	6 ± 6	27 ± 38	13 ± 59	5 ± 79	220 ± 305	179 ± 515
Other mosses	25 ± 19	67 ± 92	138 ± 175	15 ± 241	234 ± 347	103 ± 531
Ericaceous	12 ± 36	12 ± 52	-1 ± 46	45 ± 70	41 ± 101	66 ± 184
Other vascular	4 ± 10	21 ± 55	122 ± 275	13 ± 442	-9 ± 370	332 ± 694
Total	47 ± 43	128 ± 125	272 ± 334	79 ± 514	487 ± 600	680 ± 1031

When compared to other species, *Sphagnum* biomass accumulation throughout the restored site over the six initial years post-restoration was comparable in value to other mosses (mainly *Polytrichum*, *Hepatica*, *Mylia*) and other vascular species (e.g. *Carex,* graminaceous, *Sarraceniaceae*) (Figure 4.6). Biomass accumulation at the restored site ranged from 583 ± 445 g m^{-2} for moss species (other than *Sphagnum*) to 175 ± 167 g m^{-2} for *Ericaceous* shrubs. *Sphagnum* moss biomass accumulation was 451 ± 420 g m^{-2} over the initial six years post-restoration. The total mean biomass production over six years was 282 g m^{-2} yr^{-1}.

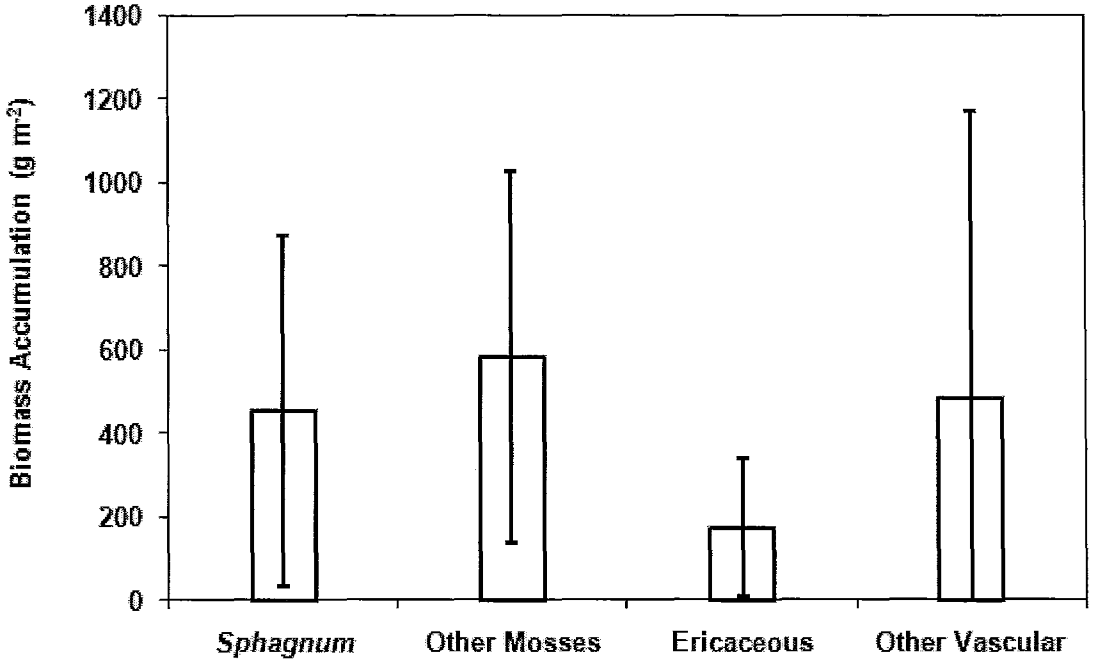

Figure 4.6: Biomass accumulation over six years post restoration for the four vegetation functional types.

In contrast, over the six years, the cutover portion of Bois-des-Bel showed higher biomass accumulation for ericaceous vegetation (mean 280 ± 310 g m^{-2} in comparison to 175 ± 167 g m^{-2} at the restored site), but lower accumulation of biomass for other mosses (72 ± 171 g m^{-2}), other vascular (8 ± 18 g m^{-2}), and almost no *Sphagnum* accumulation, except at one location (154 g m^{-2}) out of 16 locations sampled at the unrestored portion of the peatland, which provides a mean of 10 g m^{-2} for *Sphagnum* biomass. This represents an NPP value of 0.4 g m^{-2} yr^{-1} (assuming a 25 year accumulation period since peatland abandonment).

4.2.3 New peat layer decomposition

For the restored site, decomposition rates were significantly higher for both *Sphagnum* with ericaceous and predominantly *Sphagnum* treatments for litter bags filled with new peat and placed in the upper peat layer (8.1 ± 2.6 % and 9.1 ± 2.7 % respectively), than for bags placed in the old peat layer (3.7 ± 2.0 % and 4.5 ± 2.9 % respectively) ($p = 0.001$ and $p = 0.002$ for each treatment respectively) (Table 4.2). However, the opposite trend existed for litter bags filled with old peat, as decay was significantly higher in the catotelm for both vegetation functional types ($p < 0.001$ and $p = 0.01$ for ericaceous and *Sphagnum* surfaces respectively). *Sphagnum* decay rate was significantly higher ($p = 0.001$) than the *Sphagnum* with ericaceous cover treatment for bags filled with new peat and placed in the upper new peat layer but not significantly different between the two for the lower depth. The rate of decay over one year for the newly formed *Sphagnum* layer at the Bois-des-Bel restored site was 9.1 ± 2.7 %. Decay was similar for *Polytrichum* (dominant moss species at the site other than *Sphagnum*) dominated surfaces, whether the bags were filled with new or old peat (4.0 ± 3.3 % and 3.8 ± 1.6 % respectively). For the cutover portion of the site, bags were only placed in the old peat and the decay rate was significantly higher ($p = 0.001$) for bags filled with new peat (9.5 ± 3.3 %) than for bags filled with old peat (5.7 ± 2.1 %). For the natural peatland, decay rate was similar regardless of the peat type in the litter bag (9.8 ± 4.8 % vs. 8.3 ± 12.2 % for new and old peat filled bags respectively).

Table 4.2: Mean decomposition (%) over one year. New (N) or old (O) peat was placed in litter bags, and the bags placed on different surfaces (treatments) at two depths: in the newly formed peat layer (A) or in the old present before restoration (C).

Peat Type	Treatment	Depth	Mean (%)
N	*Sphagnum*	A	9.1 ± 2.7
N	*Sphagnum*	C	4.5 ± 2.9
N	*Sphagnum* with Ericaceous	A	8.1 ± 2.6
N	*Sphagnum* with Ericaceous	C	3.7 ± 2.0
N	*Polytrichum*	A	4.0 ± 3.3
N	Unrestored	C	9.5 ± 3.3
N	Natural	A	9.8 ± 4.8
O	*Sphagnum*	A	2.0 ± 1.1
O	*Sphagnum*	C	4.7 ± 4.0
O	*Sphagnum* with Ericaceous	A	1.9 ± 0.6
O	*Sphagnum* with Ericaceous	C	4.3 ± 1.6
O	*Polytrichum*	A	3.8 ± 1.6
O	Unrestored	C	5.7 ± 2.1
O	Natural	A	8.3 ± 12.2

4.3 Predicting Restoration Success

4.3.1 Water table and new layer thickness

The mean water table position at the restored portion of Bois-des-Bel significantly increased after restoration (in the fall of 1999), from a value of -65.4 ± 6.9 cm in 1999 to -30.0 ± 9.5 in 2000 (Figure 4.7) ($p < 0.01$). More subtle variation for mean water table values was observed among the initial years post-restoration, however a higher mean water table value was observed in 2006 (-12.4 ± 12.1 cm). The minimum water table position for the site also increased post-restoration, remaining variable between the initial years but decreasing in magnitude in 2006. Seasonal variability increased at the restored site as seen by the higher standard deviations for mean water table values.

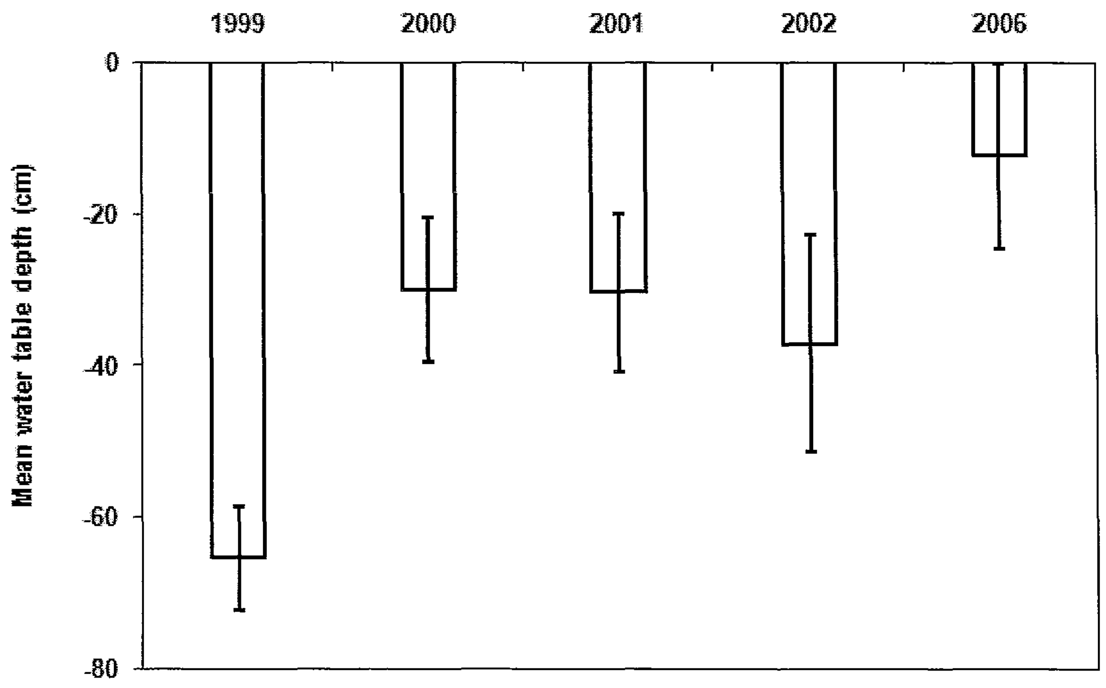

Figure 4.7: Seasonal mean water table position (WT) (in cm) over the summer months for the restored portion of Bois-des-Bel.

As previously discussed, in 2007, the mean thickness of the new peat layer at the restored site was 13.6 ± 6.5 cm. Despite this being greater than the mean water table position at the site in 2006 (12.4 ± 12.1 cm), high variation around the mean water table value caused summer water table fluctuations to fall below the newly formed moss layer, which in turn rendered this layer not truly restored from an ecohydrological perspective.

4.3.2 Summer water deficit model

The average summer water deficit for Bois-des-Bel over three consecutive years was -64 mm (Table 4.3). This demonstrates that on average over a summer season, cumulative evapotranspiration exceeds cumulative precipitation at the site and as such the water table position is expected to drop over the summer months. The average depth to which the water table position drops over the summer months is a function of the summer water deficit represented by P-ET and the storage capacity of the new peat layer, represented by S_y.

Table 4.3: Summer* cumulative P (precipitation) and ET (evapotranspiration) values for the restored portion of Bois-des-Bel for 2000-2002.

	2000	**2001**	**2002**	**Mean**
P	220	254	210	228
ET	248	374	253	292
P-ET (mm)	-28	-120	-43	-64 ± 49
P (Jun – Aug) in relation to the long-term annual average	32%	37%	37%	-

* Day of the year, 138-243.

As the new peat layer develops, the ability to offset summer water deficit induced water table drop increases from initial conditions post-restoration, where no moss layer exists and water table changes are controlled only based on the lower storativity properties of the old peat layer, until a point where a thick enough moss layer has developed and summer deficit induced water table drop is contained within this layer. This would vary according to site specific peat properties of the old catotelm and the new moss layer, but also vary according to climatic conditions (P-ET), which places special attention to inter-annual climatic variability. S_y appears to remain relatively constant with increasing depth in the upper moss layer, and starts to decrease with depth below the surface once in the catotelm peat (Figure 4.8). Average S_y for the moss layer was 0.34 ± 0.04 and for the lower catotelm layer was 0.10 ± 0.02.

The water deficit model predicts that water table depth would be determined by the storativity properties of the old peat in a cutover peatland as well as by the evaporative demand in relation to precipitation input over the summer months. However, as a new peat layer is formed over the old peat, its higher storativity will aid in offsetting some of the evaporative demand until a point where it is thick enough to contain the water table fluctuations over the summer months. For example, if P-ET for a site is -60 mm, initially the cutover site would have a water table position of -60 cm as there is no new peat layer to moderate the water table drop in the catotelmic peat (S_y = 0.1) (Figure 4.9). By incorporating the values obtained for BDB into the summer water deficit model, the model predicts that a 19 cm layer of moss would be required to offset summer deficit induced water table drop at the site.

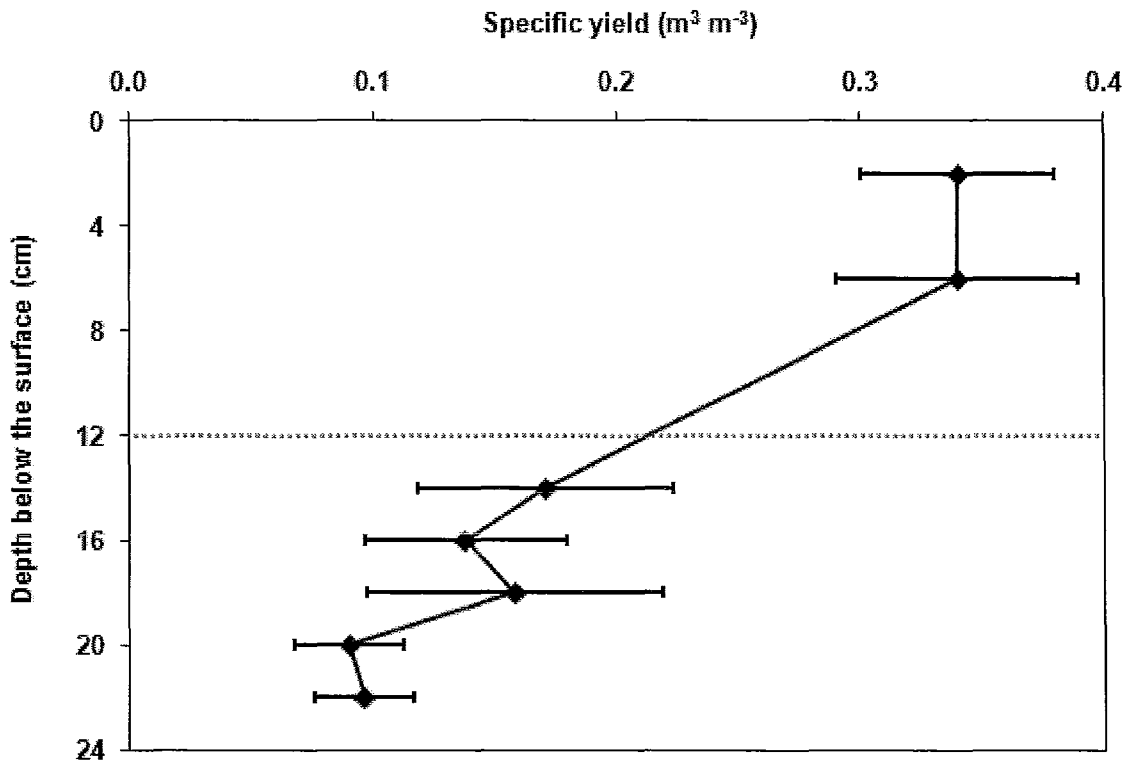

Figure 4.8: Specific yield values with depth in the peat profile for the restored site. Error bars represent the standard deviation around the mean value. Dashed line represents the transition between the upper new layer and the old catotelm.

Using the maximum and minimum P-ET values based on one standard deviation around the calculated mean for Bois-des-Bel (Table 4.3), the required thickness would range from 33 to 4 cm respectively. By applying the same logic to the storativity of the new peat layer and varying it one standard deviation above and below the mean value of 0.34, a range in thickness from 21 to 17 cm is obtained. It appears that larger inter-annual variability associated with environmental conditions is likely to have a more significant impact in determining the thickness required to offset summer deficit induced water table drop.

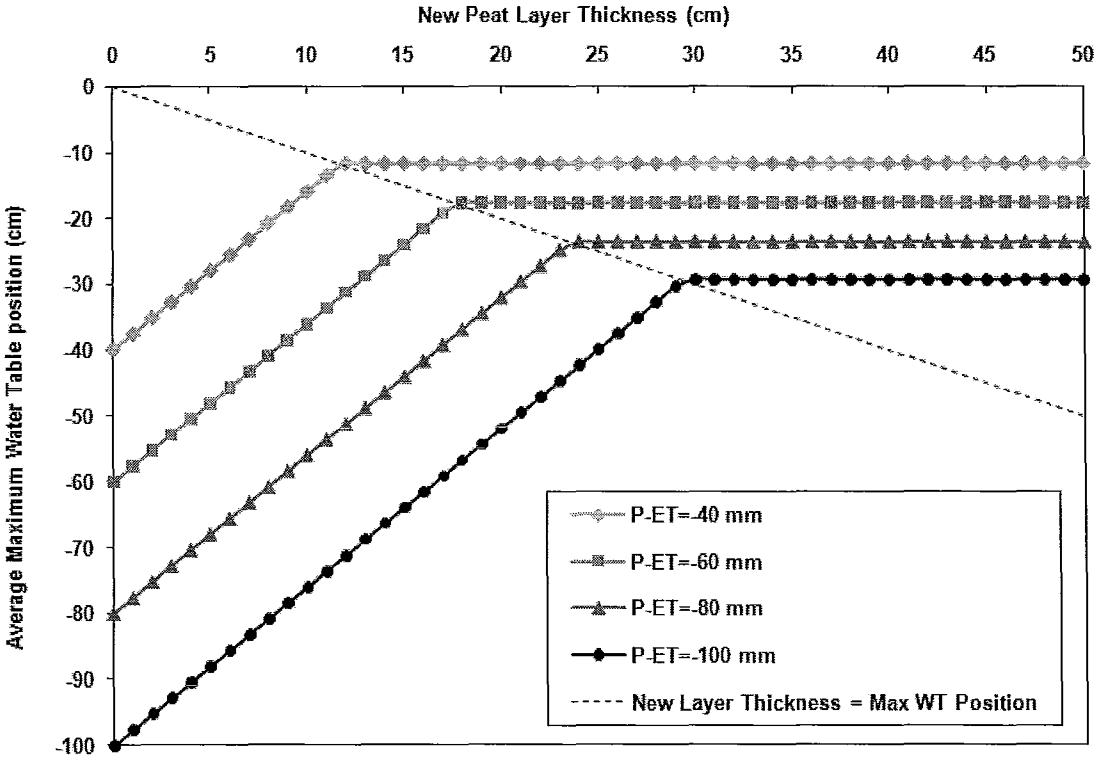

Figure 4.9: Lowest summer water table position at BDB as the moss layer grows. Dotted line represents the point at which lowest water table position equals moss layer thickness, an estimation of the thickness required for the layer to be considered restored.

4.3.3 Model for acrotelm development

Clymo's (1984) concept of a model for acrotelm growth was used and the model adapted and parameterized for BDB in order to estimate how long it would take to develop a 19 cm new peat layer at this site. Model parameters include average dry biomass production, which was calculated based on mean NPP over the last five years of data available (2002 – 2005) (p_a = 0.38 kg m^{-2} yr^{-1}), bulk density of the upper peat layer (average between the upper 0-4 cm and 4-8 cm layers, 22 kg m^{-3}) (see Chapter 3), and the annual decomposition rate for *Sphagnum* in the newly formed layer (0.07 yr^{-1}, assuming that *Sphagnum,* other mosses and ericaceous/other vascular constitute approximately the same proportion of the produced mass and taking into account the individual decay rate for each of these species - see table 4.2). The acrotelm growth model predicts that initially, the mass of accumulated peat rapidly increases however this rate seems to decrease with time as the slope of the curve decreases (Figure 4.10). The model predicts that the mean thickness of the acrotelm layer would reach the 19 cm acrotelm threshold determined in the previous section 17 years post-restoration (Figure 4.10). When comparing model results to observed mean thickness values for the years of 2003, 2005 and 2007, the model appears to overestimate biomass accumulation in the initial years post-restoration (2003 and 2005), and underestimate it for the latter year (2007).

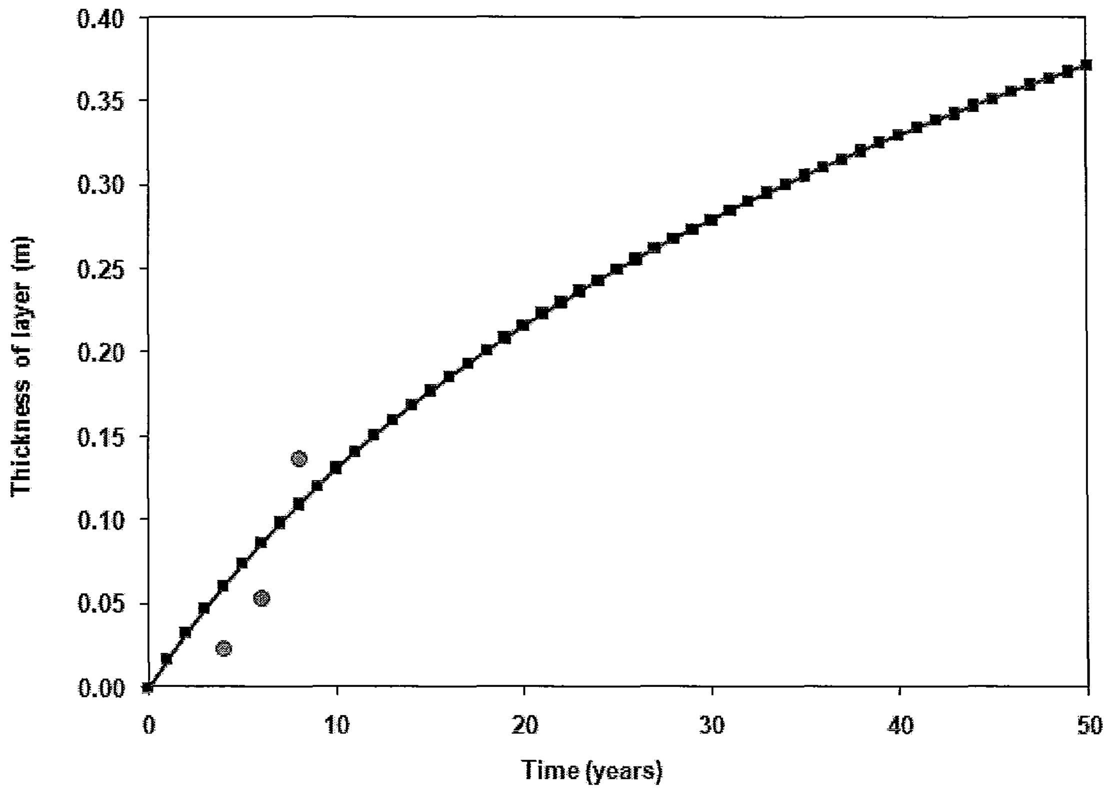

Figure 4.10: Estimated thickness of the new layer at the restored site over time using Clymo's (1984) model and site specific parameters. Grey dots represent actual mean thickness 4, 6 and 8 years (2003, 2005 and 2007) post-restoration.

The model was run for three scenarios of biomass production to examine the model sensitivity to this parameter, given its inter-annual variability over the initial six years post-restoration (Figure 4.11). The following scenarios were considered: (1) average NPP over the last five years of data available, $p_a = 0.38$ kg m^{-2} yr^{-1}, (2) average NPP over six years post-restoration, $p_a = 0.28$ kg m^{-2} yr^{-1} and (3) NPP for the last year of data available, $p_a = 0.68$ kg m^{-2} yr^{-1}. The model demonstrated a better fit to observed results for the 0.28 or 0.38 production values, as the 0.68 scenario greatly overestimated the thickness of the new layer for all data points. Using the 0.28 and 0.38 kg m^{-2} yr^{-1} p_a values, the decay rate of 0.07 for the new layer or 0.09 for only new *Sphagnum* decomposing in the new layer, and bulk density values of 19 kg m^{-3} (upper portion of new layer), 22 kg m^{-3} (mean) or 24 kg m^{-3} (lower portion of new layer), the time to form a 19 cm new peat layer ranged from 14 to 37 years post-restoration (Table 4.4).

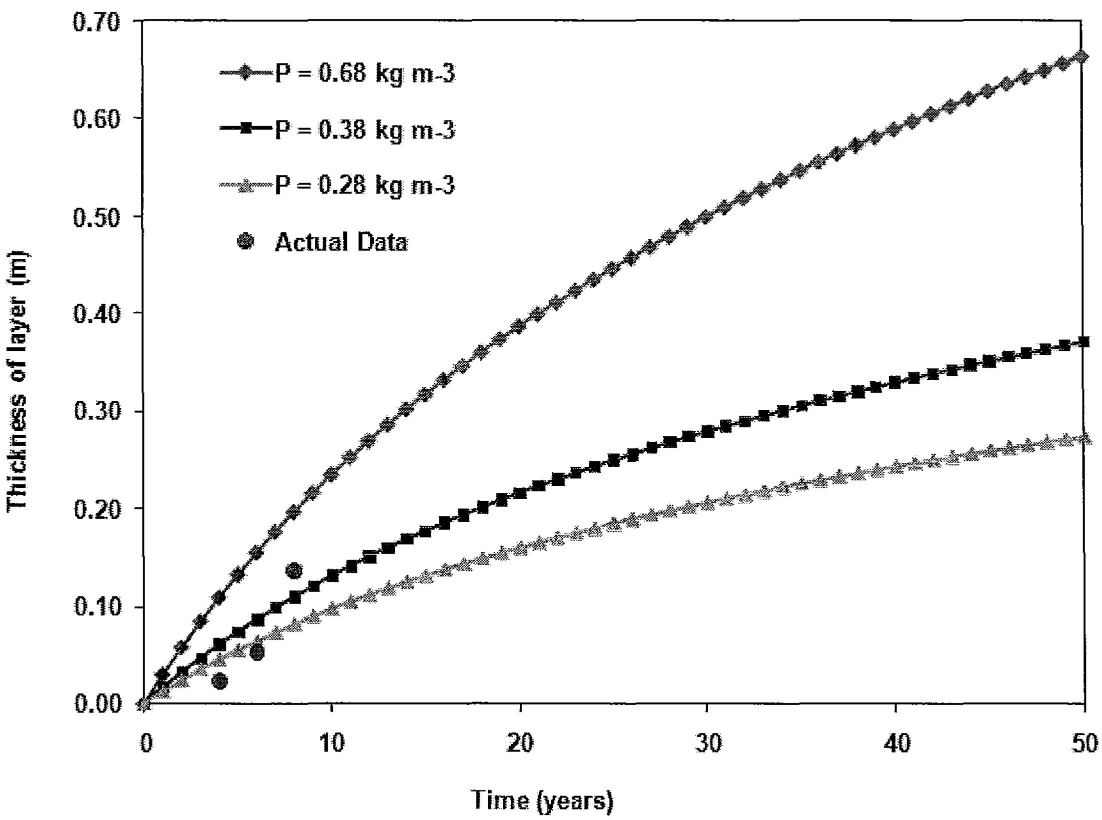

Figure 4.11: Three scenarios of estimated thickness of the new layer at the restored site over time using Clymo's (1984) model and site specific parameters. Grey dots represent actual mean thickness 4, 6 and 8 years (2003, 2005 and 2007) post-restoration.

Table 4.4: Estimation of the number of years to produce a 19 cm new peat layer based on combinations of values for production, decay and bulk density of the new peat layer at Bois-des-Bel.

Bulk Density ($kg\ m^{-3}$)	**Decay** (%)	**Production** ($kg\ m^{-2}\ yr^{-1}$)	**Number of Years**
19	0.07	0.28	21
19	0.07	0.38	14
19	0.09	0.28	25
19	0.09	0.38	16
22	0.07	0.28	27
22	0.07	0.38	17
22	0.09	0.28	32
22	0.09	0.38	19
24	0.07	0.28	31
24	0.07	0.38	19
24	0.09	0.28	37
24	0.09	0.38	22

4.3.4 GIS analysis and spatial prediction

In 2007, approximately 23% of the restored peatland was above the 19 cm threshold thickness (Figure 4.12). Coupling the 2007 thickness data by cell to the acrotelm growth model, an estimation of the locations that would be above the threshold thickness in 2016, 17 years post-restoration was obtained. In 2016, it is estimated that ~57% of the site would have a thickness of or higher than 19 cm (Figure 4.13), an indication that peatland ecosystem restoration from a carbon accumulation and hydrologic perspective may be achieved in the medium-term. At this point, it can be suggested that over 50% of the site would be restored from an ecohydrological perspective and the site likely underway to full ecosystem restoration.

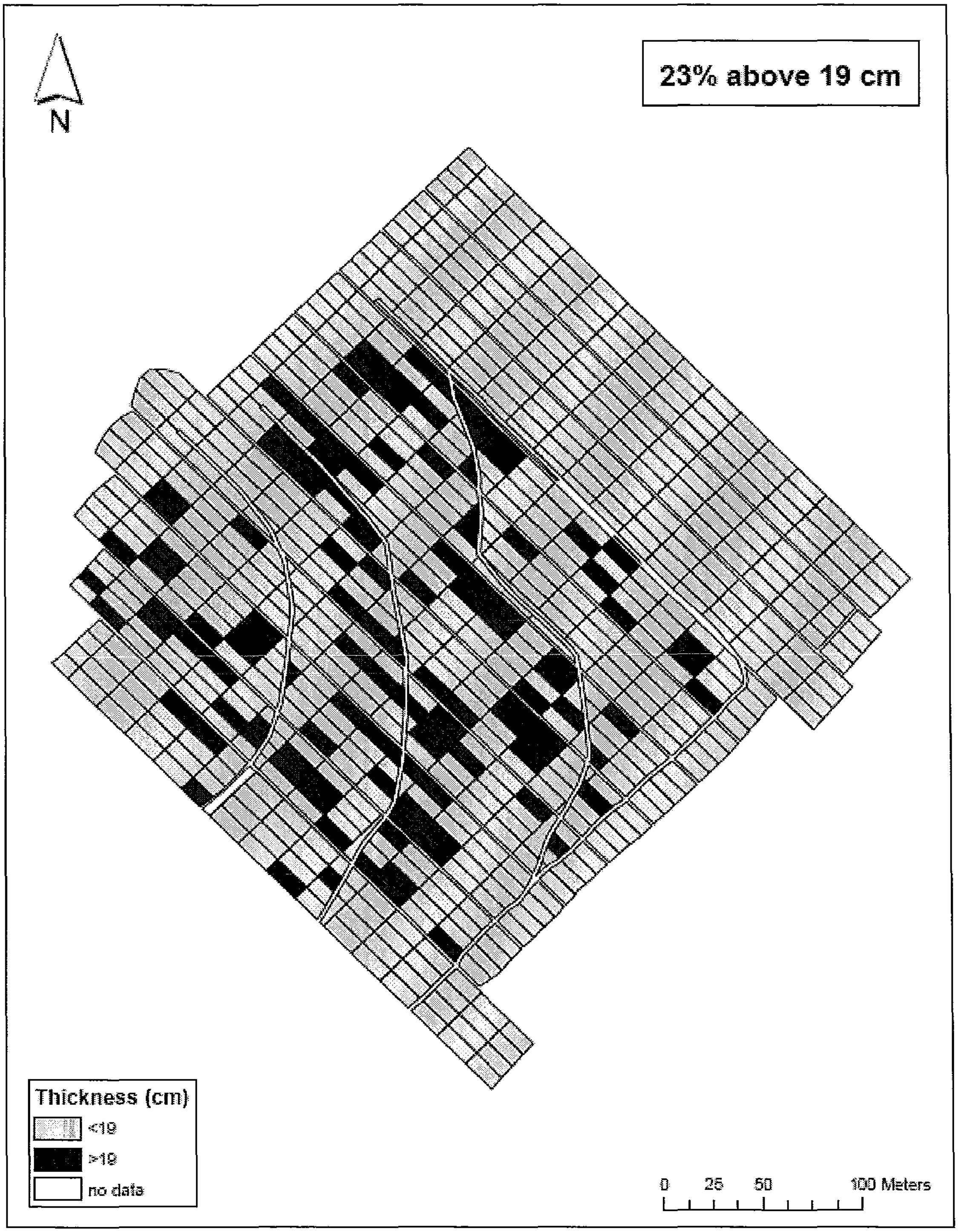

Figure 4.12: Locations above the 19 cm threshold in 2007, 8 years post-restoration at Bois-des-Bel.

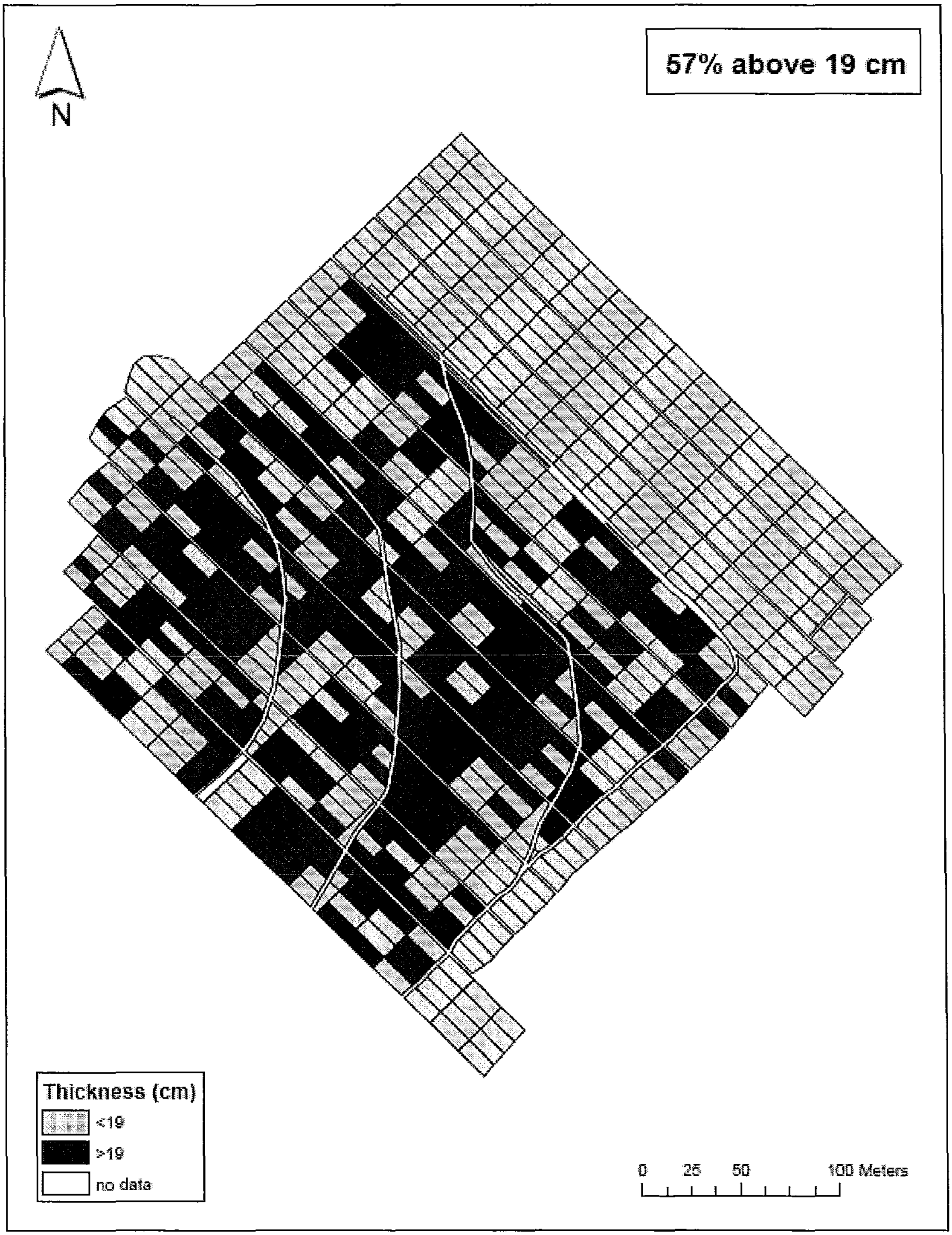

Figure 4.13: Locations estimated to be at or above the 19 cm threshold 17 years post-restoration at Bois-des-Bel.

4.4 Discussion

4.4.1 Development of a new peat layer

Peatlands represent significant sinks for carbon, with an estimated 455 Pg of carbon stored in peatland soils which represents almost one-third of the total global soil carbon (Gorham, 1991). Peatland ecosystem function can be, therefore, defined by the long-term accumulation of carbon in the soil as peat. This is a result of high and stable water levels, which enable the slow decay of plant matter and transfer for storage in the catotelm. A healthy acrotelm is what links together these ecohydrological processes operating in a natural peatland. It ensures that water table levels can be maintained high and above the catotelm even during periods of high water deficit such as the summer months. In this manner, the successful restoration of a cutover peatland will be defined by the reestablishment of a functional acrotelm. In the short term, the return of key moss species such as *Sphagnum* is the main goal of peatland restoration, and for this the hydrology of cutover sites must be manipulated to ensure adequate conditions are met for these non-vascular plants to grow (Gorham and Rochefort, 2003). In the longer term, it is necessary that the biodiversity, trophic organization of plants and animals, and the productivity, decomposition and biogeochemical cycles characteristic of these systems be re-established (Gorham and Rochefort, 2003).

Sphagnum moss has successfully returned to grow at the restored portion of Bois-des-Bel due to active restoration. Estimates of biomass productivity for natural peatlands vary considerably, but results from this study are comparable to findings from other studies on

Sphagnum dominated bogs. One estimate of productivity of adjacent pools, lawns and hummocks on the same bog (*Sphagnum* dominated) obtained 490, 380 and 240 g m^{-2} yr^{-1} respectively (Clymo and Reddaway, 1971). Moore et al. (2002) measured *Sphagnum* growth at the Mer Bleue bog over two years using the crank wire method, and report that average *Sphagnum* NPP was 140, 210 and 225 g m^{-2} yr^{-1} for a hummock, hollow and fen sites respectively. The authors also determined that aboveground biomass for the bog, composed mainly of shrubs and *Sphagnum* was 587 g m^{-2}. McNeil and Waddington (2003) used crank wire measurements for *Sphagnum capillifolium* and estimated that NPP was 207 g m^{-2} yr^{-1} (May-October) whereas estimated NPP from gas exchange was 75 g m^{-2} yr^{-1}. Total aboveground biomass has been reported to range from 109 to 7740 g m^{-2} and NPP from 158 to 755 g m^{-2} yr^{-1} for bogs (Moore et al., 2002). Waddington et al. (2003) determined biomass accumulation and mean *Sphagnum* net production rate using the destructive sampling method for a restored peatland in Québec. Mean *Sphagnum* net production rate ranged from 222 ± 11 to 341 ± 17 g m^{-2} yr^{-1} for *S. capillifolium* and *S. fuscum* respectively, but it is important to notice that these were *Sphagnum* dominated fully established plots. The results obtained in the present study are comparable to values reported in studies for other restored peatlands, and within the range reported for natural systems by Moore et al. (2002). In this manner, despite expected annual variations in production, the restored site has successfully accumulated biomass at rates comparable to natural ecosystems, which is an indication that active restoration has been successful in initialing biomass production and accumulation in cutover peatlands.

New peat layer thickness was significantly higher for the restored portion of the peatland in comparison to the cutover portion, demonstrating the importance of restoration in the re-establishment of a new moss carpet. In addition, it appears that despite some spatial variation in thickness observed throughout the site in the initial years post restoration, after 8 years there were no significant differences between the four zones at the restored portion of the site, despite the fact that zone 1 was restored a year later than the other 3 zones. Growth of the new peat layer as estimated by thickness for this study produced comparable but higher estimates than a literature value for a restored peatland. Waddington et al. (2003) calculated an average thickness increase over 5 growing seasons in restored plots ranging from 1.1 to 1.4 cm yr^{-1}. Restored Bois-des-Bel had an average increase of 1.7 cm yr^{-1} over six years post-restoration. However, the plots studied by Waddington et al. (2003) were 100% *Sphagnum* fully established moss colonies, while in this study we examined ~500 locations throughout the entire site, ranging over a variety of conditions.

New peat decay was higher when placed in the acrotelm layer than in the catotelm layer, which is expected due to aerobic vs. anaerobic conditions found in these layers respectively. Decay rate for the natural site with new peat decomposing in the acrotelm (9.8 ± 4.8 %) is comparable to the same conditions for the restored site *Sphagnum* (9.1 ± 2.7 %) and *Sphagnum* with ericaceous (8.1 ± 2.6 %) surfaces and higher than decay for the unrestored condition (old peat decomposing in the catotelm, 5.7 ± 2.1 %), which is an indication that restoration is successful in returning decay to values more similar to those found in a natural site. The estimate of annual *Sphagnum* decomposition in the upper moss layer found in this study (9.11 ± 0.64 %) falls within the reported range for restored and natural peatlands. Waddington et al. (2003) found that mean loss over two years for *S. capillifolium* and *S. fuscum* was 17.1 ± 1.1% and 13.1 ± 0.7% respectively for a restored peatland. Rochefort et al. (1990) reported 16% loss over two years for S. *fuscum*. S. *capillifolium* annual mass loss was found to be 16% (Clymo, 1965). Mean seasonal decomposition for *S. capillifolium* was 9.1 ± 6.2 % in the revegetated Cacouna bog, Québec (McNeil and Waddington, 2003). An important factor to note is that decomposition of mosses is not generally linear with time (Johnson and Damman, 1993). Waddington et al. (2003) estimated that roughly two-thirds of the loss took place in the first year, and the last portion in the second year in a two year decomposition experiment.

4.4.2 Predicting restoration success

Peat extraction leads to the removal of the upper living moss layer and exposure of the old catotelm peat, which greatly alters the ecohydrological properties and functions of these systems. As previously mentioned, the re-establishment of the diplotelmic structure of a peatland is essential for restoration. It is not until a new acrotelm layer overlying the old catotelmic peat has developed that a peatland can be considered functionally restored. In this manner, peatland restoration focuses on returning hydrological conditions more suitable for *Sphagnum* re-establishment. As *Sphagnum* vegetation grows and forms a living new peat layer, its properties aid in further regulating the hydrological conditions of the site and possibly leading to the eventual restoration of a peatland once this layer is deep enough to moderate water table fluctuations (Price and Whitehead, 2001).

Results show that seven years post-restoration, the newly formed layer was not sufficiently thick (13.6 ± 6.5 cm in thickness) to contain mean water table fluctuations at the site (mean -12.4 ± 12.1 cm, minimum -42.6 cm in 2006), deeming it not truly restored from an ecohydrological perspective. Rochefort et al. (2003) estimated that a stable and high water table could be established about a decade after restoration. By definition, an acrotelm encompasses the full range of water table fluctuation (Price et al., 2003) and therefore its depth must be defined as the lowest water table position achieved in the dry summer months. In peatlands, change in water table position in the summer is determined by cumulative water deficit. The water deficit model estimates the relative change in water table position given cumulative water inputs (P) and outputs (ET) since runoff can be considered negligible in the summer months (Price, 1996), and the water storage capacity of the peat layer. S_y values obtained for the old layer (0.1 ± 0.02) and the new peat layer (0.34 ± 0.04) are within ranges reported in the peatland literature for the catotelm and acrotelm respectively (Price et al., 2003; Price, 1992; Price, 1996; Price and Schlotzhauer, 1999). Van Seters and Price (2001) found similar S_y values between a natural nearby site and the Cacouna bog (revegetated site) (0.11 and 0.08 respectively) for the 5-10 cm section. To our knowledge it has not been examined how S_y for a restored peatland changes as the new peat layer grows. The water deficit model assumes that the system re-saturates itself post-snow melt and therefore does not take into account microtopography. Nonetheless, the model is a simple and useful tool for predicting the thickness of a new peat layer required to encompass water table fluctuations over the summer. The water deficit model also demonstrates that climate has a strong control on the re-establishment of an acrotelm. Inter-annual variability in precipitation and evapotranspiration are common, and therefore it would be valuable to obtain these measurements for a longer period in a restored peatland for a better estimate of how these would impact the predicted mean thickness value. Natural peatlands have acrotelms that typically range between 10 to 50 cm thick (Clymo, 1984). The estimated required minimum thickness of an acrotelm for Bois-des-Bel (~20 cm) falls within this range for natural systems.

Clymo's acrotelm model provides an ecological view of acrotelm development based on input (production) and output (decay). Production is assumed to be constant in the model, which as was shown in this study is not the case especially in the initial years post-restoration. Also, the initial rate of decay is assumed to remain constant with time, which is unlikely to hold true as the hydrological conditions of the site improve with time post-restoration. One of the main criticisms towards Clymo's Bog Growth Model view is that it does not consider the interactions between peat accumulation and hydrological conditions (Belyea and Baird, 2006). Despite its limitations, this model and Clymo's concepts have been widely used in the field and they provides a simplistic but realistic view of the interactions between the two main ecological processes taking place in the re-establishment of an acrotelm. Van der Schaff (1999) used a similar approach to estimate acrotelm growth for a bog in Ireland with the main difference being that two other factors were incorporated: compression in the layer and a decay ratio that was dependent on pre-established water table depths and fluctuations scenarios. Results comparable to Van der Schaaf's were obtained in this study. The author also suggested that as an acrotelm is re-establishing, it is possible that development may speed up after a slow start, as conditions for accumulation change. The different production scenarios for which the acrotelm model in this study was run demonstrate that the most realistic scenario was the average of the last four years of data. It is apparent that for the initial years post-restoration production does not provide a representation of a longer term mean value, as the system recovers itself from harsh hydrological conditions imposed by harvesting and abandonment. However, as mentioned earlier, restoration is indeed successful in returning the peatland to production values which are comparable to those of natural systems only 8 years post-restoration. Coupling the GIS database for thickness at the site to model predictions, we have determined that 17 years post-restoration, 57% of the site would be above the 19 cm threshold limit and suggest that this is an indication that the site would be underway to full ecosystem restoration.

Also, it is important to note that microtopography plays a significant role in determining acrotelm thickness. Hummocks will have thicker acrotelms than hollows simply because the depth to the water table in hummock microforms will be greater than water table depth in hollows. It is therefore important that future studies include an examination of microform specific rates of production and decomposition in order to assess micro-site specific required acrotelm thickness. This would also aid in providing a better understanding relating acrotelm formation to microform development in peatlands. Despite its simplicity, the acrotelm model provides a reasonable estimate of the thickness of the newly forming layer with time, as model results fall close to field measured thickness at Bois-des-Bel. In addition, the predicted time it would take for ecosystem restoration at Bois-des-Bel falls close to values previously estimated, for example Rochefort et al. (2003) stated that the return of a restored peatland to a peat accumulating system in 20-30 years would be acceptable but the authors were are unsure if this was realistic. Model results show that this would be possible to achieve in ~20 years post-restoration, and provide basis for further research to develop on the presented suggestions. The results from this study confirm what has been previously suggested by others, for example Rochefort et al. (2003) suggested that a number of significant characteristics of *Sphagnum* bogs can be re-established in 3-5 years, a stable and high water table in about a decade, and a functional ecosystem that accumulates peat in 30 years.

4.4.3 Conclusion and implications for restoration

Peatland restoration research has shown great developments in the past decade and currently we have a sufficient understanding of how to manipulate the hydrology and ecology of cutover peatlands (e.g. Rochefort et al., 2003), improving the conditions of these sites and consequently minimizing anthropogenic impacts with respect to carbon losses as CO_2, even returning sites to net sinks for CO_2 in the short-term (Greenwood, 2005). *Sphagnum* has been successfully re-introduced as a result of restoration, and it appears that some characteristic properties of these sites such as production and decay can be returned in the short-term. However, it is important for researchers to also focus on better understanding restoration from a peatland development perspective. Little is known about how the hydrological and physical properties of the growing moss layer changes with time, and more importantly how this relates to acrotelm development. Restoration will not have reached its main objective until an acrotelm has developed, and currently little is known about the dynamics of the ecohydrological processes operating in the re-establishment of the diplotelmic nature of restored peatlands. The simple hydrological and peat formation models utilized in this study are optimistic and suggest that this may be achieved in the medium-term, through the development of a thick enough new peat layer able to encompass water table fluctuations, restoring the ability for long-term accumulation of carbon as peat; however this would greatly depend on site specific peat hydrophysical properties as well as climatic controls.

CHAPTER 5: SUMMARY

5.1 Conclusion

Pristine peatlands store approximately one-third of the world's soil carbon through the long-term accumulation of carbon as peat (Gorham, 1991). In Canada and Europe, peatlands are exploited for peat fuel and horticultural peat, which has an impact on the hydrologic conditions and carbon balance of these ecosystems. The uppermost layer of peat (acrotelm) that is characterized by a high porosity, specific yield and saturated hydraulic conductivity is removed during extraction. This alters local hydrological conditions resulting in a generally lower and 'flashier' water table position, limiting water availability for peatland vegetation (Price et al., 2003). This is particularly damaging to *Sphagnum*, one of the dominant peat-forming species in peatlands, since *Sphagnum* is a non-vascular plant that requires a constant supply of water (Price, 1997). Recent advances in peatland restoration techniques (e.g., Rochefort, 2000) have succeeded in the re-vegetation of *Sphagnum* moss on previously cutover surfaces. The main goals of this study were to examine the development of a new peat layer post-restoration as well as its ecohydrological properties, as this is essential for assessing peatland restoration. Results presented in this thesis provide evidence that restoration has assisted in the return of a *Sphagnum* moss layer at the site; however, its ecohydrological properties differ from those of a nearby natural site. Model results from this thesis are encouraging and suggest that restoration could be achieved in the medium term, when an acrotelm is re-established at Bois-des-Bel.

The properties of the new peat layer differed significantly between the sites, especially for the lower (8-12 cm) layer. Lower samples for the natural and naturally revegetated sites had a bulk density of 43 ± 5 kg m^{-3} and 41 ± 11 kg m^{-3} respectively, almost twice as high as the value for lower samples from the restored site (24 ± 4 kg m^{-3}). *Sphagnum rubellum* capitula density (ρ_C) was significantly higher ($p < 0.05$) for the restored peatland (28726 # m^{-2}) compared to the natural site (26050 # m^{-2}). Waddington et al. (2003) suggested that the higher capitula density observed in a restored peatland was a mechanism to cope with harsher hydrological conditions initially present at restored peatlands, allowing mats to keep high moisture contents due to tighter arrangement. However, this is the opposite of what was observed in the moisture retention laboratory experiment. Residual moisture content at 200 mb (-200 cm in soil tension) (θ_r) was significantly lower ($p < 0.05$) for the restored site in comparison to the natural and naturally revegetated sites for the lower samples (8-12 cm). This suggests that *Sphagnum rubellum* in a natural peatland is able to hold on to more moisture under increasing soil-tension than the same species growing in a restored site likely due to its higher bulk density and relatively more decomposed state. Moisture retention capacity of the new moss layer at the restored site will likely increase with time, as decomposition and compaction in this layer increase and carbohydrate allocation strategies potentially change with time.

This study showed that restoration techniques were successful in returning *Sphagnum* moss to the site. The new moss layer thickness increased from 2003 to 2007 at the restored site. Mean thickness increased from 2.3 ± 1.7 cm in 2003 to 5.3 ± 4.6 cm in

2005. In 2007, eight years post-restoration, mean thickness was 13.6 ± 6.5 cm and appeared evenly distributed throughout the restored site. For the cutover (unrestored) portion of the peatland, the mean values for 2003, 2005 and 2007 were significantly lower than the mean values for restored portion of the site for each year ($p < 0.001$ for all years), with mean thicknesses of 0.22 ± 0.69 cm, 0 cm and 0.81 ± 1.22 cm for the three years respectively. Accumulated new peat layer biomass at the restored site increased over the six years following restoration, ranging from 47 ± 43 g m^{-2} in 2000 to 1692 ± 932 g m^{-2} in 2005. In contrast, the cutover (unrestored) portion of the site showed higher biomass accumulation for ericaceous vegetation, but lower accumulation of biomass for other mosses, other vascular and *Sphagnum* in comparison to the restored site. The return of a continuous moss layer at Bois-des-Bel is promising; however, the ecohydrological balance of the site must be restored in order for peatland restoration to be successful. True peatland restoration will not be achieved until the diplotelmic nature of the peatland is restored and the newly forming layer becomes an acrotelm, able to contain water table fluctuations and return the peat accumulating function of the peatland. While it has been demonstrated that standard peatland restoration practices can facilitate the return of hydrological and ecological conditions that more closely resemble those of pristine peatlands (Shantz and Price 2006; Greenwood, 2005), it may take many years post-restoration for an acrotelm to develop and consequently, as it has been hypothesized, for the carbon exchange functions and hydrology of the site to be considered fully restored (McNeil and Waddington, 2003). The simple water deficit model and Clymo's (1984) acrotelm formation model utilized in this study are optimistic and suggest that the development of a new acrotelm may be achieved in the medium-term (17 years) at the Bois-des-Bel peatland, through the development of a thick enough (19 cm) new peat layer able to encompass water table fluctuations and restore the ability for long-term accumulation of carbon as peat at the site. However, the time required to develop an acrotelm at other restoration sites would greatly depend on site specific peat hydrophysical properties as well as differences in regional climate. Nevertheless, this study has demonstrated that it is possible to restore a peatland post extraction from an ecohydrological perspective.

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