Investigating regeneration in a raised ombrotrophic bog after peat extraction

by

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Abstract

Burns Bog is a raised ombrotrophic bog in Delta, British Columbia and faced with myriad disturbances. This study is focused on the impact and restoration of peat extraction by the Atkins-Durbrow Hydropeat method. Depth to water table, relative abundance and distribution of vegetation, and the degree of peat decomposition at consistent-depth intervals were investigated to elucidate the status of passive and active ecological restoration in three fields previously harvested for peat approximately one decade apart and compared to a fourth unharvested field. Summary statistics, Redundancy Analysis, and regression were used to compare restoration status and trends in hydrology, vegetation composition, and peat accumulation. A lag period between cessation of harvest and implementation of restoration, coupled with rapid anthropogenic climate change, serve as impediments to restoration here. Intervention in the form of improved rainfall retention, assisted recolonization, and the introduction of nurse species are recommended to improve bog function and resiliency.

Keywords: Atkins-Durbrow Hydropeat method; Burns Bog; ditch blocking; ecological restoration; peat extraction; raised ombrotrophic bog

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List of Acronyms

ARP	Applied Research Project
CH_4	Methane
CO_2	Carbon dioxide
MSW	Municipal solid waste
MUAV	Multisensory unmanned aerial vehicle
PCA	Principal Components Analysis
RDA	Redundancy Analysis

Glossary

Acrotelm	The aerobic peat layer containing the fluctuating water table.
Aerenchyma	Spongy tissue found in some plant species that aids in respiration of water-logged roots.
Atkins-Durbrow Hydropeat method	A peat extraction technique involving felling trees and hosing peat, followed by pipeline transport to a local processing plant.
Capitulum	The top of a *Sphagnum* plant; plural: capitula.
Catotelm	The anaerobic peat layer.
Diplotelmic	A peatland comprised of two distinct layers (i.e., acrotelm and catotelm).
Ditch blocking	The damming of drainage ditches to reverse water table depression.
Drainage ditch	A ditch dug in conjunction with peat harvest to lower the water table position and ease extraction.
Ecological restoration	Activities aiding in the regeneration of a degraded ecosystem.
Ericaceae	A genus of heather species, including cranberries, blueberries, and salal.
Haplotelmic	A peatland with only one layer (i.e., the acrotelm has been removed, typically through peat extraction).
Hollow	A depression within a bog that is typically water-filled.
Hummock	A raised surface within a bog that is at least 20 cm above the surrounding area.
Keystone species	A species with a disproportionately large role in an ecosystem.
Mire	Peat-forming ecosystem.
Mire breathing	The swelling of peat to hold surplus water or compression upon drying to reduce the distance from ground surface to water table.
Paludification	*Sphagnum* propagation that occurs through horizontal spread.
Peat	Soil-like organic matter of varying decomposition levels that accumulates in saturated conditions when plant productivity outpaces decay.
Raised ombrotrophic bog	A category of peatland fed only by precipitation and distinguished by nutrient-poor, acidic sites, hummock-forming *Sphagnum* and Ericaceae species, and a surface that rests above groundwater.

Re-wetting	The reversal of drainage, resulting in a heightened water table position.
Sphagnum	A keystone genus of the bryophyte group and characteristic of raised ombrotrophic bogs. Species within this genus act as ecosystem engineers, releasing acid that lowers soil pH and releases nutrients. A major contributor to peat accumulation.
von Post Degree of Humification Scale	A ranking system from H1-H10 based on the physical changes of peat as it decomposes. H1 equates to undecomposed and H10 equates to fully decomposed peat.

Introduction

Overview

The Burns Bog Ecological Conservancy Area (hereafter referred to as Burns Bog) is 2,259 ha of land co-managed by Metro Vancouver and the City of Delta, within a 5,000+-year old raised bog and currently extending approximately 3,000 ha (Biggs, 1976; Metro Vancouver 2007, 2010; Dr. Sarah Howie, City of Delta, personal communication) (Figure 1). Near Vancouver, British Columbia and on the Fraser River delta, Burns Bog is unique for its estuarine environment, chemistry, biota (Hebda et al., 2000), and marine influence (Christen et al., 2016). Further, it is North America's largest undeveloped urban landmass (Delta, 2018) and western Canada's largest active (i.e. peat-accumulating) peatland within a dense urban centre (Christen et al., 2016). In 2012, Burns Bog and parts of the Fraser River estuary were designated a "Ramsar Wetland of International Importance," a title denoting national and international collaboration on informed conservation of wetlands and their inherent wealth of resources (Metro Vancouver, 2012). This designation aids in the protection of wetlands as they are under extreme pressure for development and resource extraction (Daigle and Gatreau-Daigle, 2001).

Figure 1. **Map demonstrating the placement of Burns Bog (red boundary) within the lower mainland, British Columbia and the location of the Vancovuer Landfill within the bog perimeter. Created with ArcGIS (ESRI, 2018).**

Historical disturbance has been extensive and includes drainage and removal of surface vegetation for peat harvesting in 40% of the remaining bog, conversion to agriculture, and isolation due to operation of a large regional landfill and adjacent urban and industrial development (Hebda et al., 2000; Howie et al., 2009; Metro Vancouver, 2010). These activities heavily altered hydrology, especially by lowering the position of the water table. Reversing this alteration (i.e. “rewetting”) is the first step in restoration (Rydin and Jeglum, 2013; Bess et al., 2014; Mackin et al., 2017a). Without intervention, drainage can have an irrevocable impact on peat structure, leading to the establishment of new vegetation communities (Haapalehto et al., 2011). In 2001, a pilot project was initiated in Burns Bog using weirs to control water outflow (Howie et al., 2009). Monitoring of the water table position and vegetation response began in 2005, and

Burns Bog is to be managed in the long term as a functional raised bog (Metro Vancouver, 2007). In 2007, the City of Delta ditch blocking program resumed and more than 450 weirs have been built to date in Burns Bog (Howie, pers. comm.)

Wetlands have an ecological role (i.e. habitat provision for plants and animals, carbon sequestration, and the cycling and storage of water) far greater than their surface area (Wheeler and Shaw, 1995; MacKenzie and Moran, 2004; Rydin and Jeglum, 2013). They are dynamic, shifting over time and enduring climatic changes over the long term (Daigle and Gatreau-Daigle, 2001).

Peatlands are a category of wetland marking 3% of the earth's surface, yet they represent the largest terrestrial carbon sink globally (IUCN, 2017). They form where soil saturation leads to anaerobic conditions, slowing decomposition and thus allowing carbon accumulation and subsequent peat formation (Ingram, 1982; Craft, 2016). Decomposition here is out-paced by plant growth (Grover and Baldock, 2013). As peat accumulates over millennia, carbon dioxide (CO_2) from the atmosphere is deposited via photosynthesis and stored in deep, water-saturated peat (Evans et al., 2014), comprising one-third of global soil carbon (Waddington et al., 2009). Anthropogenic pressures on these sizable carbon stores are plentiful and represent a potentially inordinate source of CO_2 and methane (CH_4) (Frolking et al., 2011; Evans et al., 2014). Further, as peatlands straddle the line of anaerobic and aerobic respiration, depending on position of the water table, their volume of emissions is highly mutable (Beer and Blodau, 2007).

Raised ombrotrophic bogs are a category of peatland fed only by precipitation and distinguished by nutrient-poor, acidic sites, hummock-forming *Sphagnum* and Ericaceae species, and a surface that rests above groundwater (Ingram, 1982; MacKenzie and Moran, 2004; Rydin and Jeglum, 2013). *Sphagnum* species are considered ecosystem engineers and a keystone genus in the restoration of bog systems because they establish conditions of low pH, enabling their competitive advantage over non-bog species (Craft, 2016; Mezaka et al., 2018). As discussed by Christen et al. (2014), Burns Bog is characteristic of a raised bog in the following ways: 1. the peat mound rests above the regional water table, 2. there is an internal water mound near the ground surface, 3. water is sourced solely from precipitation, resulting in low pH and nutrient levels, 4. it has a diplotelmic structure (discussed below), and 5. bog vegetation, especially *Sphagnum* and Ericaceae species predominate.

Ivanov (1953) pioneered the consideration of peatlands as comprised of two layers, followed by Romanov (1968) who asserted that the top layer is “active” while the layer below is “inert.” As described in Ingram (1978), “acrotelm” refers to the vertical region beginning at the mire soil surface and ending at a boundary of stark physical change. This layer experiences the greatest energy exchange with the surface. Below is the “catotelm,” a layer of considerably less physical and biological variation and energy exchange. Ingram refers to a bog with both layers intact as “diplotelmic” and a disturbed bog lacking the acrotelm as “haplotelmic.” Peat harvest typically removes or causes substantial disturbance to the acrotelm, resulting in haplotelmic status and exposing the catotelm (Gottlich and Kuntze, 1976). The acrotelm is generally 0.1-0.5 m thick and has a high hydraulic conductivity and decomposition rate (Clymo, 1984). Hydraulic conductivity refers to the transmission of liquid through a porous medium, specifically water through peat soil in this context (Grover and Baldock, 2013). Additionally, this layer contains the fluctuating water table, receives intermittent oxygen inputs as the water table drops, and contains a plethora of microfauna and microflora (Ingram, 1978). The catotelm has a considerably lower decomposition rate and hydraulic conductivity (Clymo, 1984), reduced permeability (Bu et al., 2019), consistent saturation by water, and negligible microbiodiversity (Ingram, 1978). It is now known that decomposition and ecological function are still present in this layer and it is not inert (Ingram, 1978; Clymo, 1984). Under normal conditions, about 90% of the acrotelm is lost to decomposition while the remaining 10% is added to the catotelm, with proportion of each influenced principally by temperature (Clymo, 1984). The boundary between the acrotelm and catotelm is largely determined by the degree of decay of vegetation (Wallet et al., 1992). A bog’s surface can also be considered as primary (i.e. resulting from natural peat accumulation) or secondary (i.e. the surface has been removed, for example by peat extraction, altering morphology and function) (Lindsay et al., 2014).

Morris et al. (2011) note that the diplotelmic model is a frequently cited but not rigorously tested hypothesis. These authors argue that it is overly simplistic, especially when applied to disturbed peatlands. Upon disturbance, abiotic and biotic processes are altered, potentially leading to violations of a single depth boundary. Positively though in this context, the authors note that research to validate the model has overrepresented raised bogs. They offer alternative hypotheses, including considering an additional transitional boundary between the catotelm and acrotelm (i.e. “mesotelm”), considering a continuous biogeochemical gradient, or considering hot and cold spots (i.e. areas within a peatland that are more or less productive than average). Taminskas et al. (2016) added to the growing understanding that peatlands, especially upon disturbance, do not have a single depth threshold but instead peat is vertically and horizontally heterogeneous, making comparison even between different regions of the same peatland challenging.

My Applied Research Project (ARP) fits within the Metro Vancouver Mission Statement for Burns Bog, which seeks to restore and maintain this raised bog ecosystem with evidence-based methods (Metro Vancouver, 2007). Specifically, my project evaluates recovery of hydrology, vegetation, and peat accumulation due to natural regeneration and active ecological restoration occurring in three fields previously harvested using the Atkins-Durbrow Hydropeat method. Ecological restoration activities thus far have focused on ditch blocking but have led to varied vegetation responses. Periodic monitoring and timely intervention are critical to the successful restoration of peatlands disturbed by harvesting (Wind-Mulder et al., 1996). This research is therefore valuable for intervention determined by preliminary results; it will inform future bog restoration policies and procedures and aid in tailoring future restoration efforts to area-specific conditions.

Carbon Sequestration

Peat harvest in Burns Bog ceased in the 1980s, and work in the decade prior estimated a remaining peat volume of 1.09x108 m^3 and a dry mass of almost 4,000,000 metric tons, totalling 2 Tg of carbon (Biggs, 1976). Globally, peatlands store at least 550 Gt of carbon, more than that stored cumulatively by all other vegetation types (IUCN, 2017). Bogs in their natural state sequester carbon most effectively, stored as saturated peat in the catotelm (Lindsay et al., 2014). Drainage for harvesting purposes exposes deeper, anaerobic peat to oxygen, increasing CO_2 and CH_4 emissions through oxidation (Christen et al., 2014).

Drainage and Peat Extraction

Despite the long history of peat harvesting, it is not considered sustainable due to the slow rate of peat accumulation and magnitude of disturbance on the landscape (Price et al., 2003). For example, the rate of peat decay decreases over time as microbes eliminate easily decomposed portions first, leaving a concentration of relatively decay-resistant older peat (Hogg, 1993; Bu et al., 2019). This means that harvest not only removes some of the carbon store but also reduces the rate of sequestration. Further, though harvest itself is damaging, drainage was often performed in conjunction, lowering the water table and easing peat harvest (Lindsay et al., 2014). Drainage causes peat to be oxidized to $CO_{2,}$ escaping the system and entering the atmosphere (Craft, 2016). Further, the remnant peat undergoes subsidence and subsequent compaction (Craft, 2016). Interestingly, drainage alone may maintain primary bog surface, while peat harvest results in a secondary surface (Lindsay et al., 2014). Restoration, however, must first address this hydrological alteration to regenerate an active (i.e. peat accumulating) surface (Craft, 2016). Some extraction methods have a smaller disturbance footprint than others. For example, the acrotelm transplant method couples extraction with restoration in the form of surface (i.e., acrotelm) replacement and has shown expedited regeneration of peat accumulation and carbon storage when compared to block cut and vacuum extraction methods. Since peat extraction is still occurring on a global scale, innovation of methods that reduce the carbon footprint of peat harvest and subsequent restoration are crucial in the face of climate change (Waddington et al., 2009).

Andersen et al. (2013) found microbial functional diversity differed between natural and harvested peatlands; harvested sites that had been restored had greater microbial diversity than both natural and harvested but not yet restored peatlands. The authors observed that harvest and drainage activities alter nutrient cycling. Because the majority of phosphorus is tightly bound in the acrotelm, extraction results in a P-limited environment. They predict that where natural revegetation occurs before active restoration, microbial activity is incited, and nutrient uptake increased. Ten years post-restoration, microbial functional diversity was greater than at reference sites, indicating restoration was not yet complete. The reduction in living biota leads to reduced nitrogen uptake, increasing nutrient availability to plants and potentially limiting carbon uptake upon revegetation. Therefore, the lag time between drainage and rewetting is important.

Peat harvesting generally involves removal of the acrotelm, reducing water storage and self-regulating capacity (Price et al., 2003). Permanent physical and chemical alterations result, especially when coupled with drainage (Price et al., 2003). This storage capacity is due to larger pores in the acrotelm compared to few, small pores in the catotelm (Price et al., 2003). The self-regulating capacity of a functional acrotelm goes by many names, but “mire breathing” will be used in this document. This function involves swelling to hold surplus water, thus increasing storage capacity for the dry season, and compressing these large pores in drought to reduce the distance between bog surface and water table (Ingram, 1983; Price, 2003; Howie and Hebda, 2018). “Mire” is synonymous with peatland and refers to any peat-forming ecosystem (Ingram, 1978).

Peat accumulation is a result of an anaerobic environment and drainage exposes this product to oxygen (Daigle and Gatreau-Daigle, 2001). Aerobic microbes now speed the decay of carbon previously tied up in this peat mass and considerable amounts of CO_2 are released to the atmosphere for up to two decades (Daigle and Gatreau-Daigle, 2001; Waddington et al., 2009). This now dry acrotelm also loses its peat-forming capacity, terminating inputs of biomass to the catotelm (Lindsay et al., 2014). Water table draw down is subsequently restricted by the catotelm’s low hydraulic conductivity and results in further pore collapse and consolidation near the drain (Lindsay et al., 2014). Drainage ditches can have impacts in a large area, even extending over an entire bog (Lindsay et al., 2014). Peat subsidence often results from drainage as oxygen further penetrates the acrotelm and compresses the peat below (Price et al., 2003; Lindsay et al., 2014). Importantly, active drainage ditches generally emit a bog’s largest proportion of CH_4 due to constant anaerobic conditions (Waddington et al., 2009), abundant labile carbon (i.e. rapidly broken down by soil microbes), and warm temperatures (Waddington and Day, 2007). Additionally, ditches may experience rapid runoff due to extreme rain events (Holden et al., 2006).

A variety of harvest methods were employed in Burns Bog, each resulting in different disturbance types, restoration timelines, and regeneration trajectories (Hebda et al., 2000; Price et al., 2003); therefore, this study was restricted to fields harvested using the Atkins-Durbrow Hydropeat method. This technique involved felling trees and hosing peat, resulting in a slurry transported via pipe to a local processing plant (Hebda et al., 2000 from Madrone Consultants Ltd., 2000). Fewer drainage ditches were required for this method, but the study area was still impacted by drainage and especially peat extraction (Hebda et al., 2000). A Burns Bog vegetation inventory in 2000 found average coverage of *Sphagnum* to be 30% on sites previously harvested with this method, versus an average of 4-7% for sites harvested by vacuum-mining and Western Peat Hydropeat methods and 16% by hand cutting methods (Hebda et al., 2000). Importantly, though, hydropeat methods resulted in major soil disturbance (Hebda et al., 2000). Restoration after Atkins-Durbrow extraction will differ slightly from that of a remnant that was vacuum-harvested bare, for example, as such a disturbance would likely result in a depleted seedbank (Waddington et al., 2009).

Restoration of Raised Bogs

In this document “ecological restoration” follows the definition provided by the Society for Ecological Restoration: “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” and “an ecosystem is recovered – and is restored – when it contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy” (SERI, 2004).

Depth to water table is a priority variable in peatland viability (Rydin and Jeglum, 2013). Raising the water table after drainage activities improves water chemistry (i.e. by decreasing pH and conductivity) (Wilson et al., 2011) and increases bog vegetation cover (e.g. Goud et al., 2018). Damming drainage ditches (i.e. “ditch blocking”) is a common practice used to raise the water table. Considerable variability in regeneration trajectories, however, demonstrates that depth to water table alone does not predict restoration results (Triisberg et al., 2014). Many studies report a rapid increase in water levels while resulting vegetation communities were highly varied (e.g. Bonsel and Sonneck, 2011; McCarter and Price, 2013).

Restoration success is marked by a heightened water table, a decrease in tree cover, and an increased proportion of bog vegetation (Hebda et al., 2000; Bonsel and Sonneck, 2011; Howie, 2013). These factors are expected to occur in step-wise fashion, with achievement of one preceding the next. Specifically, the cover of *Sphagnum* is important, both for its ability to engineer a low-pH environment in which it has a competitive advantage and its predominant contribution to peat formation (Andrus, 1986; Rydin and Jeglum, 2013).

The actively growing acrotelm layer is crucial for water retention (i.e. both storage capacity and elasticity suitable for mire breathing) and responsible for most lateral water flow (Rydin and Jeglum, 2013; Mackin et al., 2017a; Price, 2003). A functioning acrotelm is evidenced by resiliency to both high and low precipitation years (Bonsel and Sonneck, 2011) and may be delayed relative to water table rehabilitation (Howie et al., 2009; Howie and Hebda, 2018). Without sufficient retention of water, *Sphagnum* spp. are not favoured and vegetation communities change (Howie et al., 2009). The vegetation able to colonize an area with a depressed water table can access deep water via their roots and thus exacerbate this drying through increased evapotranspiration (Lindsay et al., 2014), resulting in a positive feedback loop.

McCarter and Price (2013) utilized ditch blocking as well as constructed bunds (i.e. raised walls that aid in water retention) and *Sphagnum* transplants in Bois-des-Bel bog in Quebec. They noted that although a new layer of *Sphagnum* spp. established, connectivity between the new and remnant peat layers was limited, suggesting hydrological processes (i.e. water retention and transfer) were still altered 10 years post-restoration. Despite a higher water table, fluctuations were restricted to the remnant layer, resulting in drier peat on the surface. Evaluation of restoration progress within a few years of implementation is critical to enable early intervention where necessary (Gonzalez et al., 2014b).

Restoration Efforts in Burns Bog

A large ditch blocking program in Burns Bog was initiated by the City of Delta in 2001. (Howie, pers. comm.). Summer students blocked interior ditches in 2007 and 2008, followed by a contractor in 2010 and annually thereafter (Howie, pers. comm.). This study focused on ditch blocking (i.e. both by the City of Delta and by beavers) and the passive regeneration occurring since cessation of harvest on water table height, vegetation present, and peat accumulation post peat-extraction. Preliminary monitoring (i.e. prior to this study) noted the establishment of a layer 0.1-0.2 m thick of active *Sphagnum* in some peat-harvested fields, warranting further investigation of this observation elsewhere (Howie and Hebda, 2018). Low capillarity of living *Sphagnum* increases difficulty of water uptake from the catotelm (i.e. the underlying layer of somewhat decomposed peat) unless the water table is shallow enough to maintain peat saturation (Boelter, 1964). Dry conditions prevent capillary movement of water to the bog surface, potentially leading to desiccation and mortality of *Sphagnum* spp. (Rydin and Jeglum, 2013). Initial monitoring of peripheral and interior ditch damming in Burns Bog indicated a quick response in the water table within a radius of approximately 20 m (Howie et al., 2009).

Additional restoration efforts include an underground wall pilot project, initiated in 2018 to assess whether sheet piling (i.e. a perimeter wall) can reduce the quantity of water lost to adjacent agricultural ditches (Howie, pers. comm.). Monitoring of depth to water table both at this location and throughout the bog is ongoing (Howie, pers. comm.). Additionally, both Metro Vancouver and the City of Delta are collaborating on two tree seedling removal pilot projects in recently burned areas to facilitate *Sphagnum* recolonization and inhibit tree encroachment (Howie, pers. comm.). Finally, invasive species encroachment is monitored and actively managed by Metro Vancouver (Howie, pers. comm.).

Additional Threats to Burns Bog

Raised bogs are sensitive ecosystems due to their harsh conditions (i.e. consistently moist, low pH and nutrient content), which result in highly specialized biota (Dyderski et al., 2016). Fire and drought constitute the major modern day and active threats to Burns Bog (Hebda et al., 2000). Further, raised bogs in general are susceptible to climate change, and this is especially true for bogs at their southern extent, such as Burns Bog (Howie and Hebda, 2018). Water storage will be inadequate in dry summer conditions if precipitation in winter increases but is not retained sufficiently (Hebda et al., 2000; Howie et al., 2009).

Burns Bog has been facing drought since 2015, evidenced by a lower than typical summer water table position (Howie, pers. comm.). According to global climate models, a 3°C temperature increase is anticipated in the Metro Vancouver region by the 2050s (Metro Vancouver, 2016). A climate projection report for Metro Vancouver forecasts a 5% increase in precipitation; however, this is largely expected during the already wet fall months (especially during extreme events) while summer drought is extended (Metro Vancouver, 2016). The number of summer days is expected to double by 2050 (i.e. 22 day current average to 55 days) and triple by 2080 (i.e. 79 days). Summer precipitation is projected to decrease by 20%, increasing the consecutive dry days from 21 to 26. A precipitation decrease is also expected in September, thus extending the length of the dry season. In 2018, high fire risk restrictions in Burns Bog continued well into September. Therefore, storage of fall season precipitation will be crucial to maintain bog-like conditions during summer droughts.

Northern hemisphere peatlands are more vulnerable to drought due to ecohydrological feedbacks between depth to water table and vegetation composition (Goud et al., 2018). Their role as long-term carbon sinks will be increasingly crucial in the face of anthropogenic climate change. Further, some vegetation species in Burns Bog are at the southern limit of their range, including cloudberry (*Rubus chamaemorus*), bog rosemary (*Andromeda polifolia*), crowberry (*Empetrum nigrum*), and velvet leaf blueberry (*Vaccinium myrtilloides*) (Hebda et al., 2000). Finally, multiple waterfowl species found here are at their range limit, and rare insects have been identified in wet regions of the bog and are thus threatened by drought (Hebda et al., 2000).

Historically, fire in Burns Bog was less of a concern. In fact, fire creates landscape heterogeneity and prevents woody species encroachment (Bellamy, n.d.; Hebda et al., 2000). Currently, however, fire is a major threat due to a number of factors, including drier conditions, greater proportion of early successional stages, increased human activity in the bog (Hebda et al., 2000), and a multitude of infrastructure, including a nearby natural gas pipeline, power lines, highways running through the bog and along the periphery, and various industrial activities nearby (SER, 2019). Indirect feedbacks due to anthropogenic climate change are projected to increase fire frequency (Frolking et al., 2011). Strict fire access restrictions and protocols are in place and only one fire has occurred in the last decade; in 2016 a large fire occurred in the north eastern edge along Highway 17 (Delta, 2016; SER, 2019). Fires also occurred in 2005 and 2007 (Howie, pers. comm.). Fire damage can be minimized by sufficient restoration of the water table, especially if *Sphagnum* is not eliminated from the surface (Bellamy, n.d.; Lindsay et al., 2014).

The Vancouver Landfill, in operation since 1966, is located in the southwest portion of Burns Bog (Figure 1). This 320 hectare landfill is slated to close in 2037 and receives a maximum of 750,000 tonnes of municipal solid waste (MSW) per year (City of Vancouver, 2018). In 2017, the landfill received 609,892 tonnes MSW, 68% of the region's MSW, and 126,513 tonnes of demolition waste (City of Vancouver, 2018). The landfill overlays impermeable clayey-silt, a stratum of compressed peat, and finally a layer of demolition materials (City of Vancouver, 2018). A double ditch system collects leachate, which is directed to the Annacis Island Wastewater Treatment Plant and 50 monitoring stations ensure the containment of leachate, surface water, and groundwater (City of Vancouver, 2018). Prior to infrastructure upgrades in the 1970s, leachate was directed towards the Fraser River (Dr. James Atwater, UBC Civil and Environmental Engineering Department, pers. comm.).

Vegetation

Ombrotrophic raised bogs are characterized by *Sphagnum* mosses, Ericaceae shrubs, and sedges due to harsh nutrient poor, acidic, and water-logged soils (Potvin et al., 2015). These plant functional groups are crucial to bog function (Rydin and Jeglum, 2013). *Sphagnum* spp. best indicate restoration success due to their function in peat accumulation and ability to acidify their environment (Andrus, 1986; Craft, 2016). In fact, *Sphagnum* can influence succession in wetlands through this capacity for water retention and acidification (Andrus, 1986). Further, *Sphagnum* spp. engineer their environment through paludification (i.e. propagation through horizontal spread) (Andrus, 1986; Daigle and Gatreau-Daigle, 2001). Vegetation distribution is largely determined by depth to water table (Moore et al., 2002), though proximity to source propagules, peat quality, and degree of humification can influence recolonization by *Sphagnum* (Mulligan and Gignac, 2001). Drainage must be reversed for *Sphagnum* recolonization to be successful (Smolders et al., 2003).

Numerous rewetting restoration projects have resulted in increased plant cover; however, recolonization by novel vegetation communities that differ from reference conditions sometimes occurs (Bonsel and Sonneck, 2011; McCarter and Price, 2013; Triisberg et al., 2014). This can result in an impediment to acrotelm establishment (Bonsel and Sonneck, 2011) as active surface vegetation alone does not indicate a healthy bog (Clymo, 1984). It is therefore important to ensure raised bog restoration is following a desired ecological trajectory, with a diverse and functional *Sphagnum* community and active peat accumulation. Consequently, evaluation of bog restoration progress should investigate vegetation change over time, with consideration of percent cover of *Sphagnum* spp. as a key indicator (Haapalehto et al., 2011). Howie et al. (2009)

notes that monitoring at Burns Bog will continue until it can be demonstrated that Burns Bog is on a trajectory towards a sustainable raised bog state.

Ecosystem Services

“Ecosystem services” is a term commonly used to describe the economic benefit various ecosystem processes provide to humans; however, quantifying these values remains challenging (Evans et al., 2014). This translation of ecological function to economic value is inherently imperfect, but in its absence, economic and policy decisions operate on incomplete or outdated science (Evans et al., 2014).

Functioning peatlands (i.e. peat-accumulating) afford services including flood regulation, wildlife habitat, and especially carbon sequestration and storage (Bonn et al., 2014). Evaluation of ecosystem services tends to ignore cumulative and synergistic impacts from multiple disturbances (Evans et al., 2014), like those affecting Burns Bog. Evans et al. (2014) found that UK peatlands in or near dense urban regions, like Burns Bog, were also those with the greatest shortcomings in ecosystem services and should be prioritized for restoration. The restoration of ecosystem services in disturbed peatlands is difficult after drainage because of considerable between-site variation (Renou-Wilson et al., 2019). Rewetting disturbed peatlands is complex and reducing greenhouse gas emissions may be more straightforward and rapid than is revegetation by bryophytes, including *Sphagnum* spp. (Renou-Wilson et al., 2019). Fluxes of CO_2 and CH_4 are site specific and differ widely across rewet peatlands, influenced especially by climate, nutrient levels, time since restoration, and vegetation present (Renou-Wilson et al., 2019).

Over multiple centuries, undamaged wetlands exert a net cooling effect as CO_2 is sequestered (Hemes et al., 2019). Peatland restoration is generally lauded for its capacity to mitigate anthropogenic climate change; however, rigorous testing of this assertion is required to understand both the trajectory and timeline (Hemes et al., 2019). Degraded and restoring wetlands, for example, have a very different carbon budget compared to their pristine counterparts (Artz et al., 2013) as these novel systems have unique hydrological conditions and historical uses (Hemes et al., 2019). The carbon budget in these systems is influenced by their pre-restoration condition, hydrological outcome of restoration efforts, and current stage of regeneration (Artz et al., 2013).

Couwenberg et al. (2011) investigated the utility of vegetation in predicting greenhouse gas fluxes. They asserted that long-term hydrological regime, assimilation of gases into plant tissue, and the abundance of aerenchymous plants can be gleaned from the vegetation present. Aerenchyma is spongy tissue found in some, especially aquatic, species and acts as a shunt, transporting deeper gases up to the atmosphere. The authors concluded that vegetation mapping, coupled with the plants' individual moisture requirements, reveal general water level trends. These trends are a strong proxy for greenhouse gas fluxes; where water levels were high without matching emissions, aerenchymous plants were in low abundance, and gas transportation thus limited.

Saturated soils are a major source of CH_4 (Evans et al., 2014) via methanogenesis (Christen et al., 2014). Anaerobic conditions, organic matter abundant in labile carbon, and warm temperatures increase the productivity of anaerobic microbes resulting in considerable CH_4 production (Price, 2003). CH_4 has a global warming potential of 28-36 over 100 years compared to the same mass of CO_2 (USEPA, n.d.) as CH_4 has a higher radiative efficiency but shorter lifespan in the atmosphere (Artz et al., 2013). CH_4 moves through the peat mass via three pathways: diffusion, in solution in interstitial water, and upward flow via bubbles (i.e. ebullition) (Clymo, 1984).

Restoration activities can strongly influence emissions of CO_2 and CH_4 (Christen et al., 2014) and extensive care should be taken to reduce these emissions where possible. Further research is likely required to decide best management practices in rewetting, wetland creation, and any restoration activities resulting in considerable greenhouse gas emissions. Efforts should be made to restore highly emitting wetlands as these systems tend to serve as a net carbon source (Christen et al., 2014). For example, where the acrotelm is lacking, restoration can be targeted to return capacity for peat accumulation and self-regulation (Christen et al., 2014). Rewet harvested fields become a considerable CH_4 source when the water table is within the top 0.2 m, as in parts of Burns Bog, therefore peatland ditch blocking is implemented and monitored carefully to moderate CH_4 emissions (Christen et al., 2014; Evans et al., 2014). Sulfur deposition, too, may assist in the suppression of CH_4 emissions due to competition between methanogenic microbes and sulphate-reducing bacteria (Evans et al., 2014).

Restoring a suitable hydrological regime and vegetation composition to subsided peat can reduce emissions and improve carbon sequestration capacity (Hemes et al., 2019).

According to radiative forcing and greenhouse gas warming potential models, Hemes et al. (2019) predicted that, on average, restored wetlands establish a positive greenhouse gas effect within 50 years post-restoration and serve as a net sink of atmospheric carbon roughly 100 years post-restoration. As oxygen now reaches this organic-rich peat, relatively large CO_2 emissions result, leading to subsidence and peat compression (Hemes et al., 2019). Also, the relatively high productivity compared to slow plant litter decay in these systems enables carbon accretion (Andrus, 1986; Rydin and Jeglum, 2013). By protecting and restoring wetlands, carbon storage is maximized (Hemes et al., 2019).

Research Objectives

This ARP has the overarching goal of comparing fields harvested for peat in 1948, 1957, or 1966 to an unharvested area with the expectation that, as the 1948 field was harvested longest ago, it will be most similar to the unharvested field.

Objective 1: Evaluate rewetting efforts to determine if the water table position in the site harvested longest ago (i.e. 1948) is most similar to the unharvested reference site

> Action 1.1: Plot and analyze available monthly depth to water table data from the City of Delta and collected since 2005
>
> Action 1.2: Determine current status of hydrological restoration and if there is a difference between harvested fields and the unharvested reference site
>
> Action 1.3: Compare to literature reference conditions to determine if further hydrological management is required

Objective 2: Determine if vegetation composition in the 1948 field is most similar to the unharvested reference site

> Action 2.1: In each of the four study areas, identify vegetation to species or genus and estimate percent cover in 20 1-m^2 plots systematically and randomly placed along a 300-m transect

Action 2.2: Determine if there is a difference in vegetation communities and percent cover between harvested fields and unharvested reference site to evaluate impact of harvest and restoration efforts on vegetation

Action 2.3: Compare to literature reference conditions to evaluate restoration of a peat-accumulating vegetation community

Objective 3: Determine if peat decomposition and accumulation activity in the 1948 field is most similar to the unharvested reference site

Action 3.1: At each plot, extract a peat core in the top 0.5 m and determine degree of humification every 0.1 m

Action 3.2: Compare within- and between-site variation to determine if peat profile is as expected and if fields differ in peat decomposition or accumulation

Action 3.3: Compare to literature reference conditions to evaluate restoration of a functional acrotelm layer

Action 3.4: For each field, perform investigative peat coring at plot 1, 10, and 20 to estimate maximum peat depth and provide recommendations for future research

Objective 4: Investigate the importance of microtopography and peat humification in explaining vegetation distribution

Action 4.1: Compare vegetation communities and degree of humification

Action 4.2: Compare vegetation communities and microtopography type

Action 4.3: Compare distribution of microtopography types

Action 4.4: Complete constrained multivariate analysis to investigate effect of environmental variables on vegetation composition

Methods

Study Area

A variety of harvest methods were employed in Burns Bog, resulting in different disturbance types, restoration timelines, and regeneration trajectories (Hebda et al., 2000; Price et al., 2003). Study fields were limited to those harvested by the Atkins-Durbrow Hydropeat method (Figure 2). Fields in Burns Bog harvested with this method have shown increased *Sphagnum* recolonization but greater soil disturbance (Hebda et al., 2000). All study fields are located on Triggs soils with slow drainage, considerable water-holding capacity, and a pH of approximately 4 (Biggs, 1976). Burns Bog is located within the Coastal Douglas-fir moist maritime (CDFmm) biogeoclimatic subzone (CFCG, n.d.), experiencing a modified maritime climate with wet, mild winters, dry, warm summers, and a growing season of approximately 230 days (Biggs, 1976). There is an average annual precipitation surplus of 200 mm over evapotranspiration and a deficit period from April to September (Hebda et al., 2000).

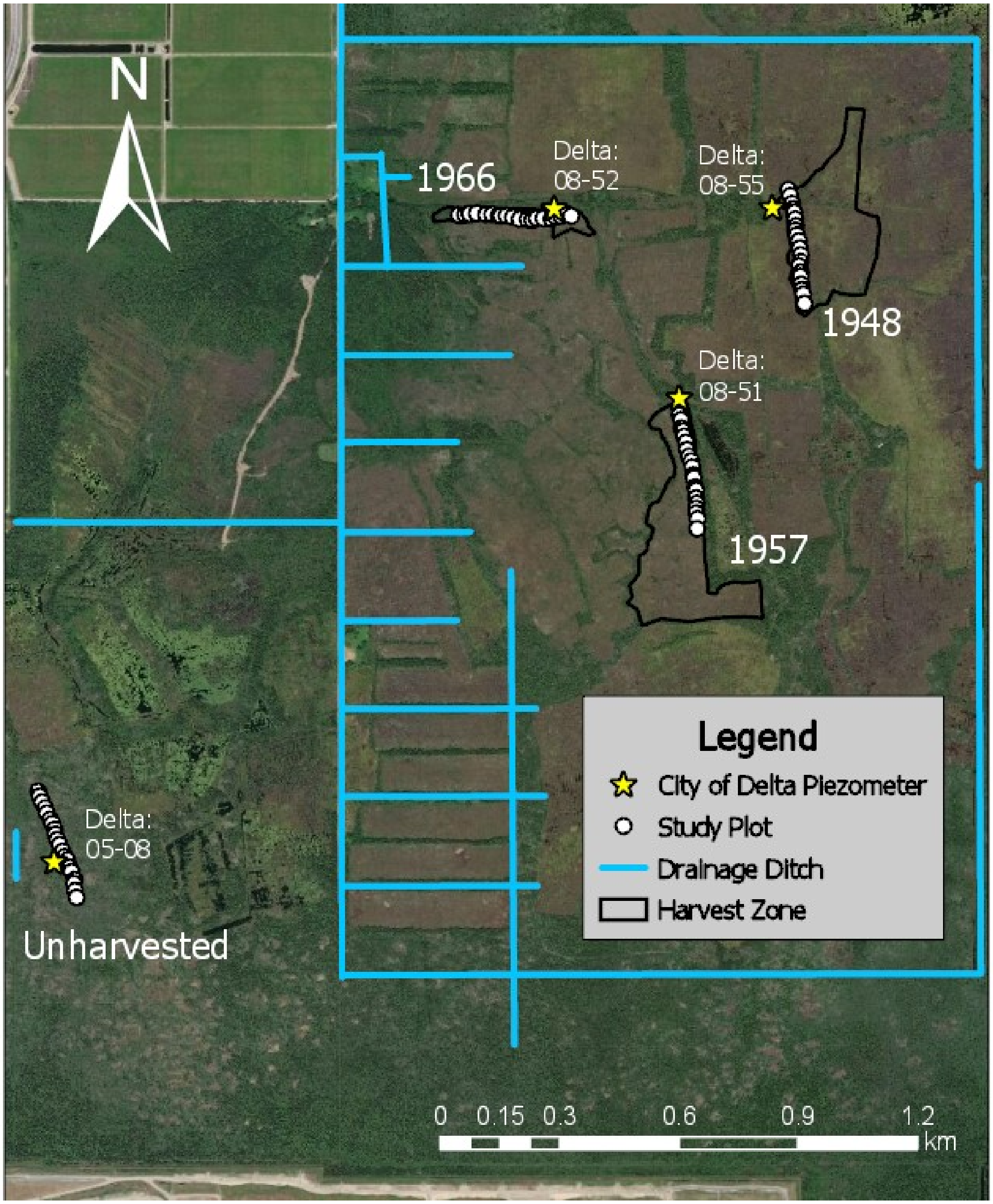


Figure 2. Study area within Burns Bog Ecological Conservancy Area with labels according to year of harvest or 'unharvested' and City of Delta piezometer. Drainage ditch and piezometer location data are the property of City of Delta. Created with ArcGIS (ESRI, 2018).

Study Site Selection

Harvest study sites were selected based on five factors:

1. Use of Atkins-Durbrow Hydropeat method, each harvested in a different decade
2. Presence of a City of Delta piezometer, enabling collection of my own depth to water table measurements
3. A long-term (i.e. at least 10 year) history of depth to water table monitoring data as successful rewetting is the first step in restoration
4. Similar vegetation type (i.e. "White Beak-rush (*Rhyncospora alba*) – *Sphagnum*" vegetation community type) according to a comprehensive ecosystem review (Hebda et al., 2000), indicating minimal variation in physical characteristics contributing to vegetation composition (Figure 3) and a trajectory toward the historical vegetation composition (i.e. "Pine-*Sphagnum* - Low Shrub") (Howie, 2004)
5. Fields in close enough proximity to each other to further minimize variation in abiotic influence

The reference area was selected as it was not harvested for peat. Additionally, upon purchase of the bog, this area has served as a reference site for researchers, enabling more rigorous investigation through data sharing. Hebda et al. (2000) found 14 differing plant communities in Burns Bog, of which only two were deemed suitable *Sphagnum* donor sites by Howie (2004) based on vegetation community composition, sufficient field size, and adequate resilience to harvest (i.e. areas least sensitive to harvest), severely limiting the availability of reference areas.

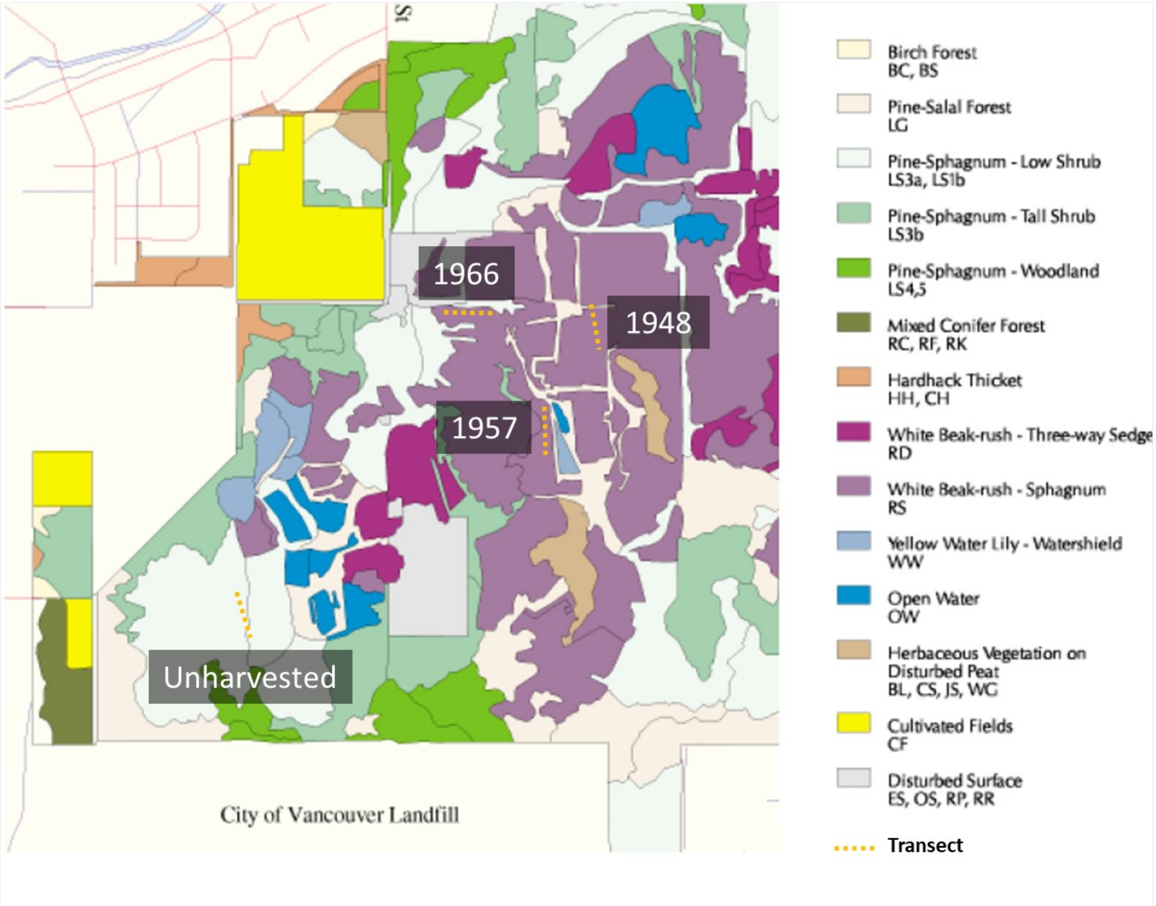


Figure 3. **Distribution of vegetation types in study area with doted yellow lines indicating study transects. Harvested fields are categorized as "White Beak-rush – *Sphagnum*" and the unharvested site is categorized as "Pine-*Sphagnum*-Low Shrub." Modified from Hebda et al. (2000).**

Depth to Water Table

Restoring hydrological function is the first step in restoration of a raised bog (Rydin and Jeglum, 2013). Measurements of depth to water table have been collected in Burns Bog since 2005. These data, property of the City of Delta, were provided for the harvested fields from 2008 to 2016 and for the reference site from 2005 to 2018. In July 2018 I collected depth to water table measurements once at each piezometer for comparison purposes as measurements in the harvested fields ceased in 2016. These data were analyzed to elucidate rewetting results to date in the three harvested fields and enable comparison of hydrological patterns between harvested sites and the unharvested reference area.

Vegetation

Vegetation is the primary response of interest in this study as successful bog restoration after peat removal hinges on reestablishment of a *Sphagnum* community capable of peat accumulation and sufficient mire breathing (Lindsay et al., 2014).

A 300-m transect was laid out on each field with 20 1 m by 1 m quadrat sample plots. A survey plot was randomly placed once every 15 m, followed by a 5-m buffer to ensure this minimum distance between plots (Haapalehto et al., 2011). This systematic, randomized approach aimed to represent both the patchiness of vegetation in Burns Bog (Howie, 2013), while providing suitable coverage and an acceptable distance between plots (Haapelehto et al., 2011). Hummocks and hollows tend to feature different vegetation communities, especially *Sphagnum* spp. (Triisberg et al., 2014; Mezaka et al., 2018; Taminskas et al., 2018); however, as the purpose of this study is to represent the regenerating fields as a whole, no effort was made to select for one topography type over another.

To reduce sampling bias that could result from selecting hummocks and hollows in the field, quadrat spacing was determined before entering the field. At each plot, topographic form was assessed as complete hummock, partial hummock if the hummock sloped down to lawn level within the quadrat, hummock-hollow combination if the hummock sloped to hollow level within the quadrat, or other for quadrats on complete hollows or lawns as it was difficult to differentiate the two. By incorporating randomization and quadrat spacing prior to entering the field, the issue in harvested sites of fewer available hummocks was addressed.

Vegetation was identified to species whenever possible and genus when necessary and percent cover estimated using a 1 m x 1 m quadrat following methods described in Anderson (1986). Percent cover was also analyzed by nine functional categories: vascular shrubs (i.e. Labrador tea (*Ledum groenlandicum*), bog laurel (*Kalmia microphylla* ssp. *occidentalis*), bog rosemary, bog blueberry (*Vaccicium uliginosum*), velvet leaf blueberry, hybrid blueberry, bog cranberry (*Oxycoccus palustris*), crowberry, and salal (*Gaultheria shallon*)), vascular herbs (i.e. cloudberry, sundews (*Drosera* spp.)), sedges (i.e. white beak-rush, *Eriophorum* spp., and three-way sedge (*Dulichium arundinaceum*)), *Sphagnum* spp., non-*Sphagnum* bryophytes (i.e. *Dicranum* or

Campylopus spp. and *Heterocladium* spp.), lichen (i.e. reindeer (*Cladina* spp.) and cup (*Cladonia* spp.) lichens), trees (i.e. Lodgepole pine (*Pinus contorta*) was the only tree observed within quadrats, though at least western hemlock (*Tsuga heterophylla*) and birch (*Betula* spp.) were observed in the area), bare ground or unvegetated pools of water, and ferns (i.e. bracken fern (*Pteridium aquilinum*)).

Peat Humification

Vegetation on the bog surface alone cannot denote the health of a peatland, necessitating the investigation of peat characteristics (Clymo, 1984; Lindsay et al., 2014). These characteristics influence both hydrology and vegetation communities (Howie, 2013; McCarter and Price, 2013; Howie and Hebda 2018) and can reveal trends in post-harvest peat accumulation. Peat extraction and subsequent in-field analysis using the von Post Degree of Humification Scale were performed using the methods outlined in Howie (2013). In regions like Canada with extensive peatlands, the von Post test is the most appropriate method to determine the degree of peat humification (Craft, 2016).

Peat decomposition occurs in three steps; first, organic matter is lost from the peat mass via gas, solution, and soil microinvertebrate activity, followed by physical structure break down. Finally, there is a chemical change, partly due to microorganisms (Clymo, 1984). The von Post Degree of Humification Scale assigns a rank from H1-H10 based on these physical changes, with H1 equating to undecomposed and H10 equating to fully decomposed peat (Bu et al., 2019). The rate of decay is influenced by vegetation species and plant organ present as well as chemical composition, meaning that vegetation composition and productivity, as well as water table position, influence the decomposition profile (i.e. the vertical soil component from bog surface to underlying sediment (Clymo, 1984; Grover and Baldock, 2013).

Although the von Post scale may appear subjective, experience in the field was straight-forward and this method is widely used in peatland research (Grover and Baldock, 2013; Craft, 2016). Multiple parameters (e.g. colour and proportion of available water, degree of decomposition of plant roots, and characteristics of the peat that escapes through the fingers) are given for each step in the scale and consensus was immediate in almost every case. Further, von Post data have been collected in Burns Bog by a number of researchers and are available for within-site comparison. They, too, found the methodology to be straight-forward and consistent (e.g. Howie, 2013; Exler, 2015).

One 0.5-m deep core was extracted per plot using an Eijkelkamp peat sampler, and degree of humification determined at 0.1, 0.2, 0.3, 0.4, and 0.5 m (following the depth intervals of Price and Schlotzhauer, (1999) and others) using the von Post Degree of Humification Scale (Eijkelkamp, n.d.; Peat and Peatlands, n.d.; Pennock et al., 2015). Determining degree of peat decomposition at set intervals will streamline comparison of pre- and post-harvested peat with the unharvested reference field. Investigative cores were taken at plot 1, 10, and 20 of each field to compare peat humification at depth and to determine maximum peat depth. Care was taken not to stand in the direct area of coring to avoid peat compression (Grover and Baldock, 2013).

Ideally, this coring would determine the acrotelm depth at each site. The upper boundary of this layer is the closed surface of moss capitula (Ingram, 1978). The lower boundary is more difficult to delineate, and somewhat subjective based on criteria used (Romanov, 1968). To some, it is delineated simply by water table position; however, for this study demarcation was considered as the boundary between rapid change in degree of decomposition and relative stability of this metric (Ingram, 1978; Money and Wheeler, 1999).

Statistical Analysis

Vegetation composition and peat humification results were presented with summary statistics and 95% confidence intervals. Due to the interrelated nature of these parameters (i.e. vegetation present both affects and is affected by water table position, peat humification, and topography type), a multivariate approach to statistical analysis was also used. Clustering analysis was used to reveal trends in vegetation distribution, followed by Redundancy Analysis (RDA) and regression to investigate important interactions.

Hierarchical clustering and k means clustering provide a characterization analysis and were used to investigate differences in vegetative composition between plots, separate from the field to which they belong. In both cases, clusters of plots are determined based on similarities in vegetation distribution. Hierarchical clustering is a bottom-up approach, beginning by grouping the two most similar plots. Conversely, k means clustering is a top-down approach, with every clustering step followed by an update to the algorithm (i.e. it is iterative), resulting in k clusters. This allows comparison of permutations with varying numbers of clusters and determination of the optimal number (k) based on differentiation of clusters and optimal cubic clustering criterion (CCC). As both methods are calculated differently, these approaches enable different visualization of dissimilarities in vegetation composition. In each case, field placement was removed from the computation, followed by colour- and symbol-coding by field to visualize site variation. Both analyses were completed in JMP 13 (JMP, 2018) with the ward grouping method and standardization of variables used for hierarchical clustering

RDA and regression were used to understand the degree of variability that can be explained by each variable in question and performed in R (R Core Team, 2018). This aids in elucidation of indicator parameters in the restoration of Burns Bog. Principal Components Analysis (PCA) and multiple regression were performed together through RDA, a multivariate vegetation response analysis constrained by environmental variables (i.e. topography type and von Post classification in this case). Prior to performing RDA, vegetation percent cover data were transformed using the Hellinger transformation to account for multiple rare species (Borcard et al., 2018). The resulting R^2 value was adjusted using Ezekiel (1930) to account for inflating explained variance due to the inclusion of multiple explanatory variables. A permutation test was then performed to verify the model and assess significance. Parametric tests were avoided as they assume normal distributions, standardized response variables, and normal error distribution and this is not the case for community composition data (Borcard et al., 2018).

Results

Hydrology

The City of Delta has collected monthly depth to water table measurements using shallow piezometers in each harvested field from 2008 to 2016 and in the unharvested reference area from 2005 to present (Figure 4). In July 2018 I collected measurements using these piezometers (Table 1). Measurements I collected fit within the expected trends explained by the City of Delta data (Figure 4). Analysis of both data sets demonstrates seasonal water table patterns and highlights water deficit periods. Additional dry season depth to water table measurements were desired but not possible in August due to fire access restrictions and in September due to a limitation in equipment availability. Figure 5 provides precipitation trends using data collected at the Environment and Climate Change Canada Burns Bog weather station (EC, 2019). This enables comparison of water table fluctuations to local precipitation patterns.

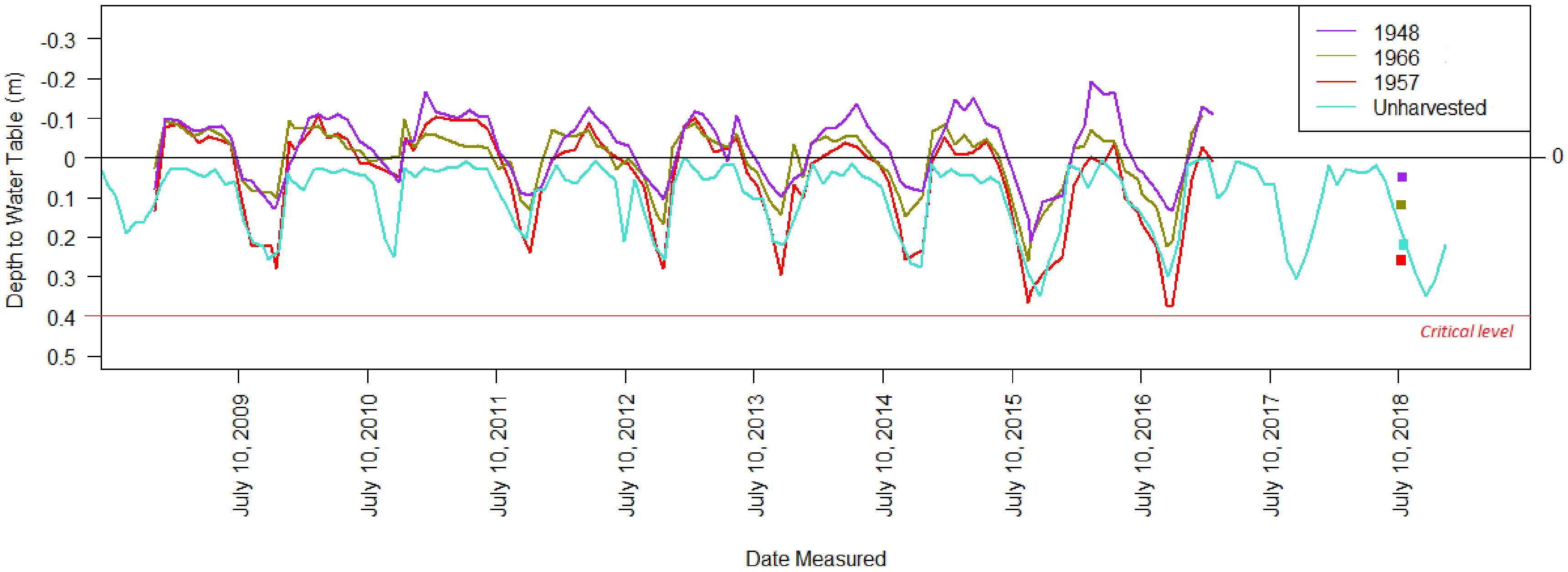


Figure 4. Monthly depth to water table measurements collected in Burns Bog by the City of Delta. Measurements I collected in July 2018 are represented by solid squares. Negative values indicate flooding. The critical level at 0.4 m represents the threshold for bog vegetation growth (Verry, 1988). Data are property of the City of Delta, used with permission.

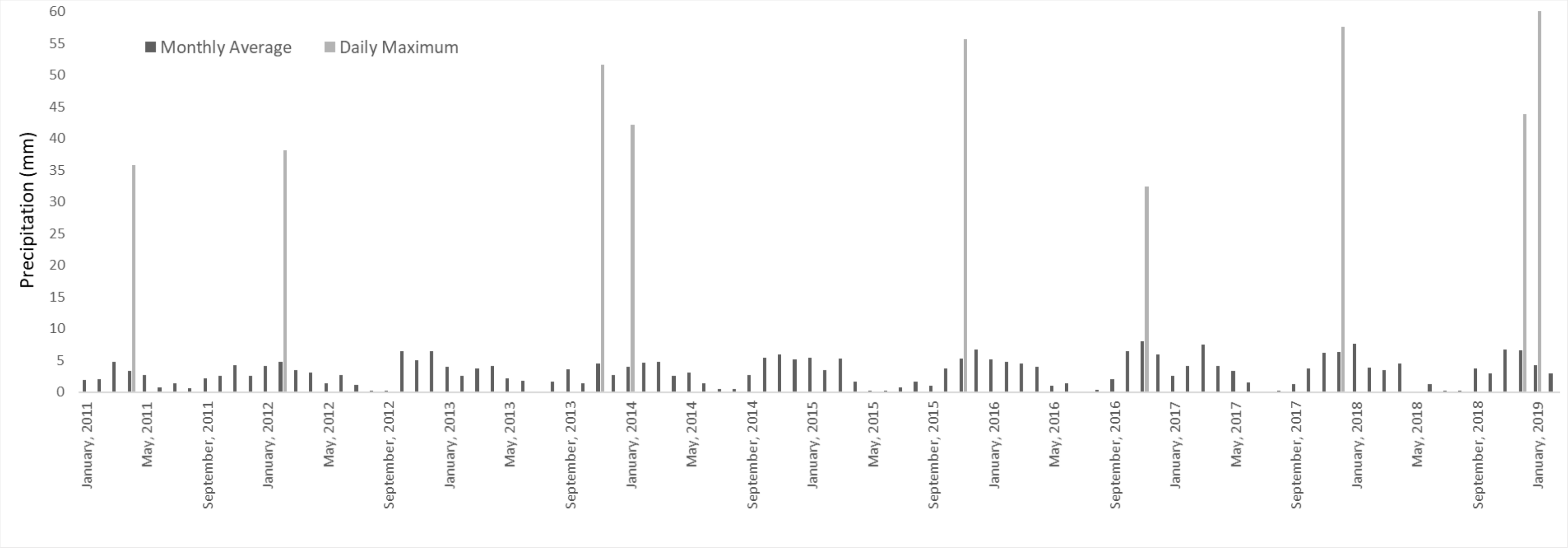


Figure 5. **Average monthly precipitation and annual maximum precipitation in a single day at the Environment and Climate Change Canada Burns Bog weather station (EC, 2019).**

Table 1. Depth to water table measurements collected at Burns Bog in July 2018 using City of Delta's shallow piezometers.

Harvest Year	Depth to Water Table (m)	Date Measured (2018)
1966	0.12	July 19
1948	0.05	July 22
1957	0.26	July 19
Unharvested	0.22	July 26

Depth to water table fluctuated more in harvested fields than in the unharvested reference where flooding was rare (Figure 4). As expected, water table position was highest in the 1948 field and flooding was common. Over the monitoring period, water table depressed in both the 1957 and unharvested fields, with both strongly approaching the 0.4 m threshold for bog vegetation growth (Verry, 1988) by 2015. This is a generally accepted threshold raised bog summer water table position used by peatland scientists (Price et al., 2003). Over this period, all but the 1948 field appear to experience surface drying. Maximum precipitation in a single day has increased in recent years, with 30-60 mm annual highs (Figure 5). Local average monthly precipitation tends to be greatest between November and March with a sharp decline between April and October. In some years, the wet season begins in September, but this timing appears to vary annually. Table 2 lists average summer (i.e. over July and August) depth to water table to investigate annual variation in seasonal low water table positions. Emphasis is placed where water table was at least 0.3 m deep for comparison to the 0.4 m threshold. Annual low values are presented for comparison and occurred in September on a number of years (Table 2). In 2012 the annual low water table position in all four sites was observed in late September (i.e. September 22 in harvested fields and September 26 in the unharvested field) and precipitation in September 2012 was almost zero (Figure 5). At the unharvested site, annual low water table was observed in September in 2009, 2011, and 2014 as well. In 2009 and 2011, again the annual low was observed in September at all sites. The 1948 field saw an additional annual low in September 2014 and 2016, and at the 1957 field in 2016. In the remaining years with annual low water table positions occurring in September, precipitation during this month had already increased considerably (Figure 5).

Table 2. **Average summer depth to water table (m) at Burns Bog in July and August, 2018 and annual low positions. Data are the property of City of Delta and collected at their shallow piezometers. Bold text indicates depths to water table of at least 0.3 m.**

		Field			
Year		**1948**	**1957**	**1966**	**Unharvested**
2005	Low				0.27
	Summer Average				
2006	Low				0.28
	Summer Average				0.23
2007	Low				0.20
	Summer Average				0.16
2008	Low				0.19
	Summer Average				0.18
2009	Low	0.13	0.28	0.10	0.26
	Summer Average	0.06	0.22	0.08	0.22
2010	Low	0.06	0.05	0.01	0.25
	Summer Average	*Not measured in July or August*			0.23
2011	Low	0.10	0.24	0.13	0.21
	Summer Average	0.05	0.12	0.06	0.16
2012	Low	0.11	0.28	0.17	0.26
	Summer Average	0.05	0.07	0.05	0.14
2013	Low	0.10	**0.30**	0.15	0.18
	Summer Average	0.08	0.23	0.13	0.22
2014	Low	0.09	0.26	0.15	0.28
	Summer Average	0.07	0.22	0.13	0.22
2015	Low	0.21	**0.37**	0.26	**0.35**
	Summer Average	0.18	**0.35**	0.23	**0.33**
2016	Low	0.14	**0.38**	0.23	**0.30**
	Summer Average	0.10	**0.30**	0.17	0.24
2017	Low				**0.31**
	Summer Average				0.28
2018	Low				**0.35**
	Summer Average				**0.32**

Vegetation

Plant community composition and abundance are depicted in Figure 6 (95% confidence intervals, t-distribution, n-1=19 df). Labrador tea distribution is relatively consistent among sites, with the exception of low abundance in the 1948 field. Bog laurel, bog rosemary, and bog blueberry, too, are in low abundance in this field, but less consistent across all fields. Considerably more bog blueberry was observed at the unharvested site compared to the harvested fields. The 1948 field had the highest relative abundance of *Sphagnum* spp., white beak-rush, bare ground, and unvegetated pools. The unharvested field had no observed round-leaved (*Drosera rotundifolia*) or English sundew (*Drosera anglica*) in study plots, the only observation of bracken fern, and the lowest coverage of *Sphagnum* spp. and other anticipated bog vegetation species including bog cranberry, bog laurel, and bog rosemary.

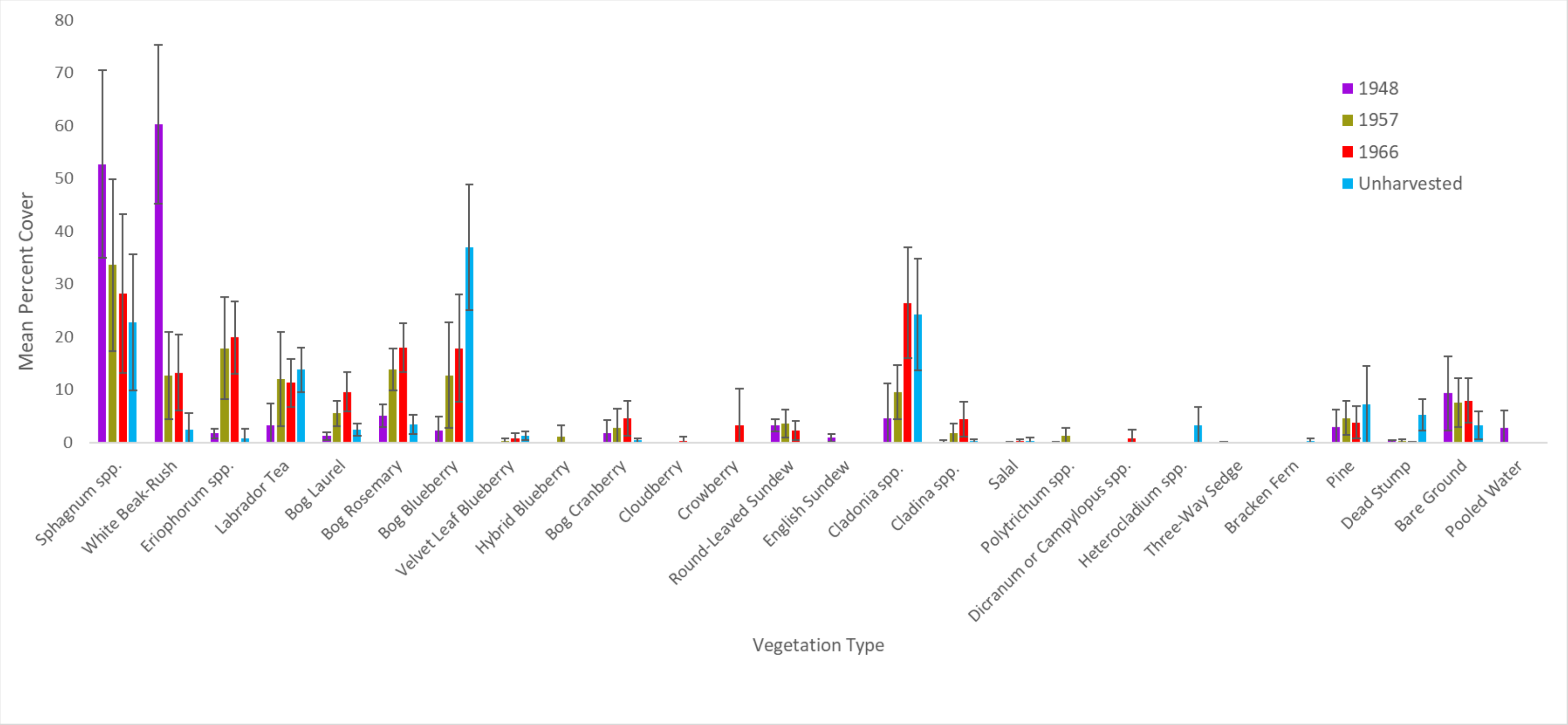


Figure 6. **Mean vegetation percent cover in Burns Bog in June and July, 2018 (n = 20 quadrats per field, error bars represent 95% confidence intervals).**

Lodgepole pine was most abundant at the unharvested site but varied little between fields. Salal, a species that can be invasive in disturbed bog conditions (Hebda et al., 2000) was rare in study plots, occurring only at the 1966 and unharvested fields. Lichen coverage was lowest at the 1948 field. Reindeer lichen was most widespread at the 1966 field, while cup lichen was most abundant at the 1966 and unharvested fields.

Eriophorum spp. including Chamisso’s cotton-grass (*Eriophorum chamissonis*), narrow-leaved cotton-grass (*Eriophorum angustifolium*), and tawny cotton-grass (*Eriophorum virginicum*) were observed. These species flower at different times of year, making identification to species was difficult. Tawny cotton-grass is an invasive species in Burns Bog, so every effort was made to delineate the species; however, to avoid implications of misidentification in statistical analysis and conclusions, *Eriophorum* spp. are considered at the genus level in this document. Vegetation composition was also analyzed by functional category (Figure 7; t-distribution, n-1=19 df).

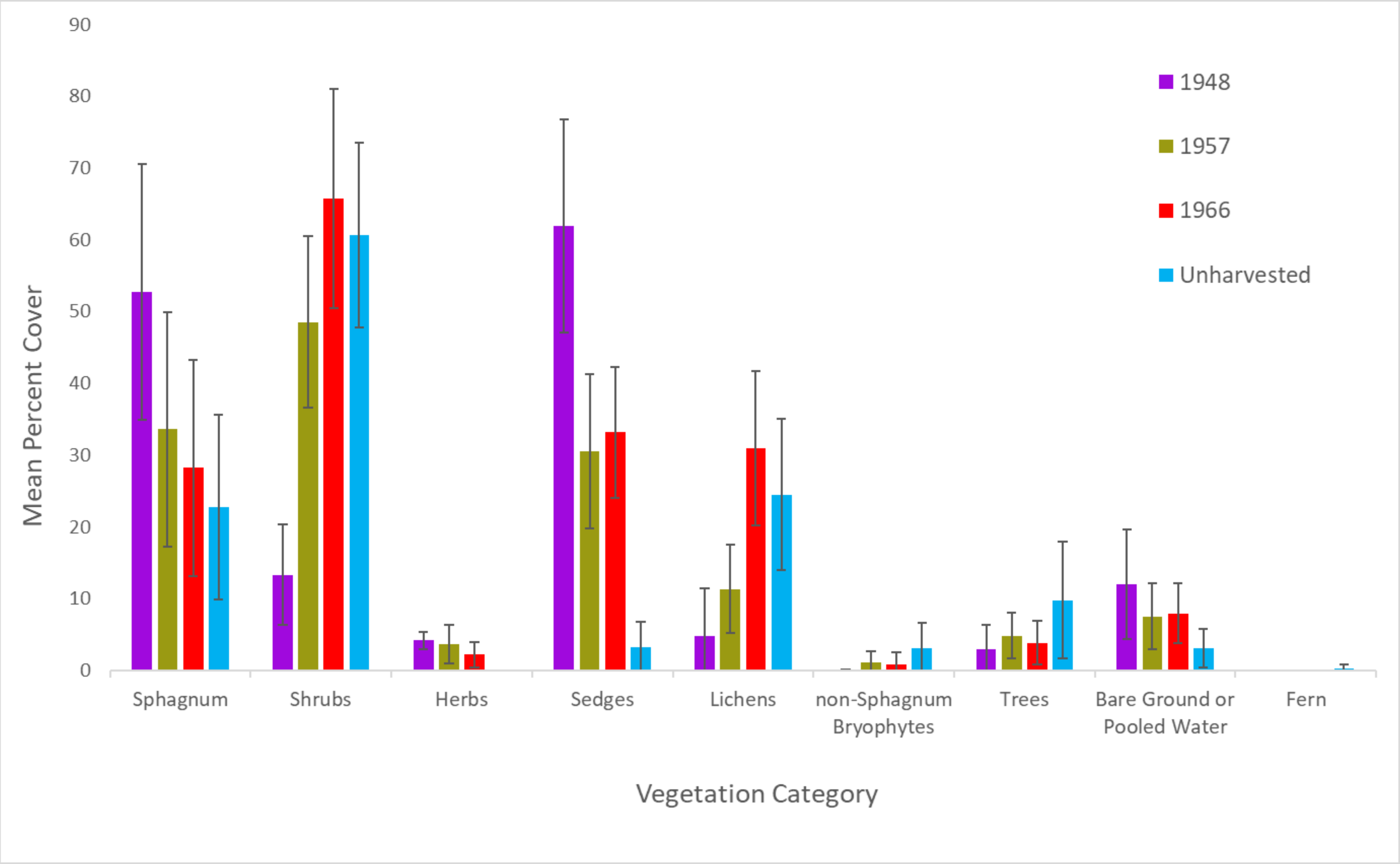


Figure 7. **Mean vegetation survey percent cover by category of each site (n = 20 quadrats per field, error bars represent 95% confidence intervals).**

The 1948 field saw the lowest cover of vascular shrubs and lichens and the greatest cover by pooled water, sedges, and *Sphagnum* spp. (Figure 7). The unharvested site had almost zero vascular herbs or sedges and little bare ground. The 1957 and 1966 fields have relatively similar vegetation composition, except in term of lichen coverage, which is considerably greater at the 1966 field (evidenced by no overlap of 95% confidence intervals). Generally large confidence intervals indicate relatively high variation in a number of these vegetation categories.

Clustering Analysis

Hierarchical Clustering Analysis

Hierarchical clustering of vegetation data enables visualization of within field variation (Figures 8 and 9). Figure 8 is read from left to right, with each plot couple along the left-hand margin being the most similar. This bottom-up incremental similarity merging results in a dendogram (Figure 8) and constellation plot (Figure 9), providing different visualizations of the same clustering. This approach resulted in eight clusters based on vegetation dissimilarity, indicating that vegetation alone does not adequately describe site variability and highlights the utility of constrained multivariate analysis to demonstrate the influence of environmental variables.

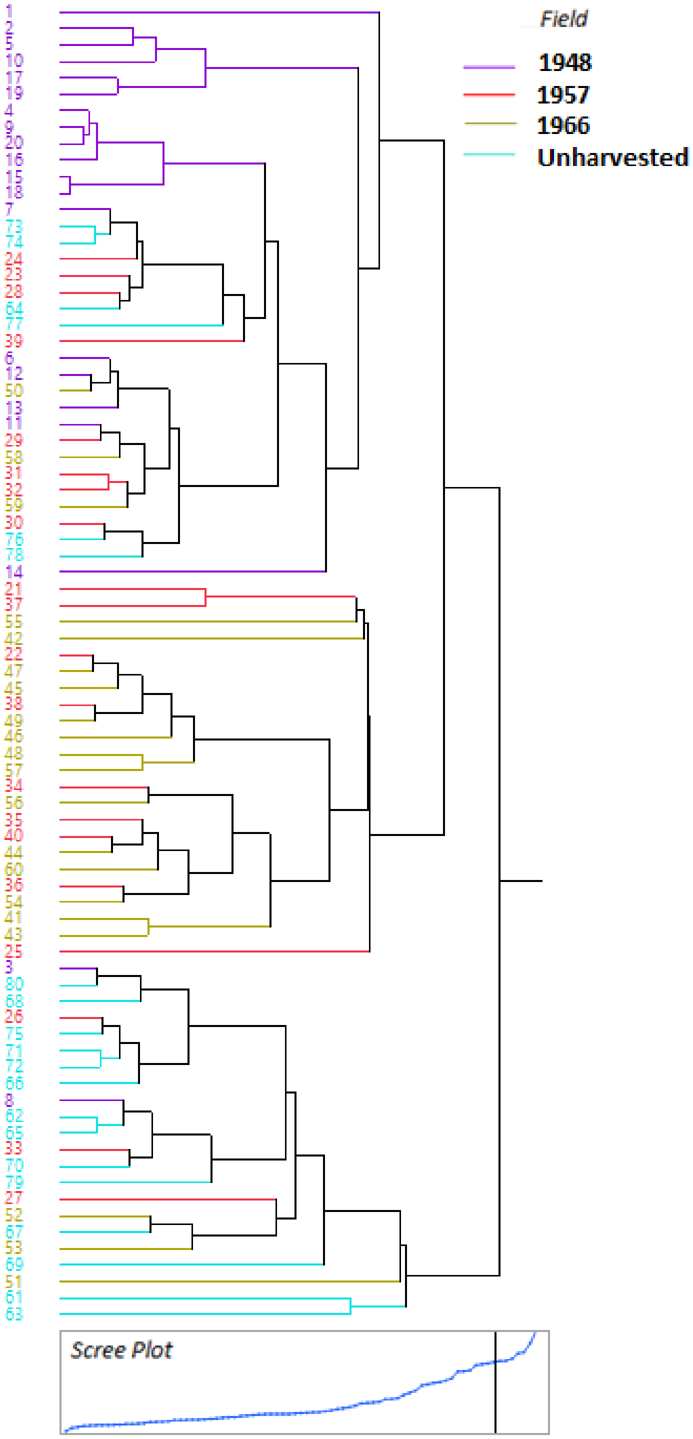


Figure 8. Hierarchical clustering dendogram demonstrating the progression of dissimilarity in vegetation in Burns Bog in June and July, 2018 (n = 20 per field). The scree plot indicates the optimal location of tree cut results in 8 clusters. Numbers along left side indicate plot number (i.e. 1948 field is represented by numbers 1-20, 1957 field by 21-40, 1966 field by 41-60, and unharvested field by 61-80).

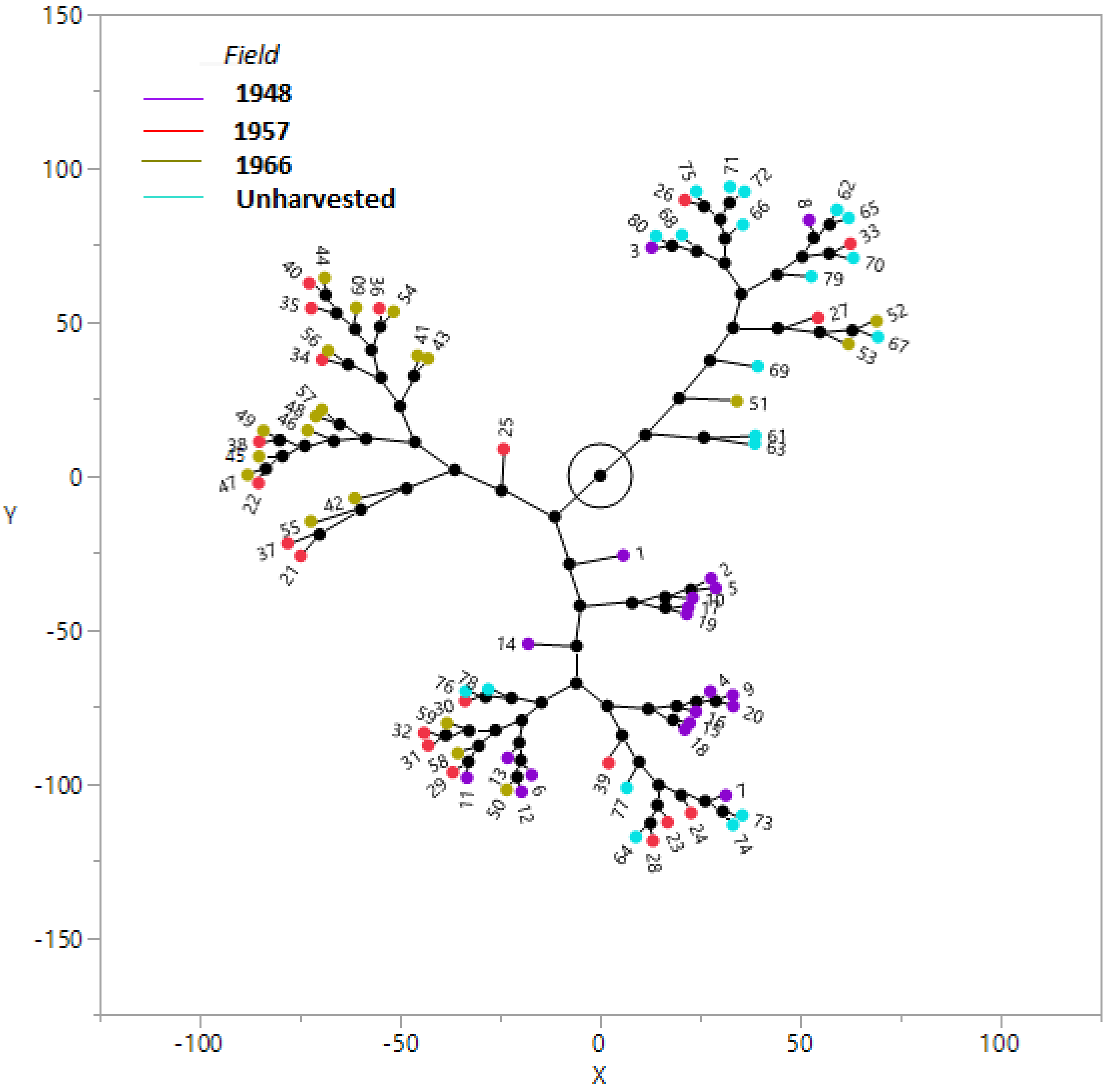


Figure 9. **Constellation plot demonstrating distance (i.e. dissimilarility) of vegetation in Burns Bog in June and July, 2018 between hierarchical clusters (n = 20 per field, circle represents tree root).**

Figures 8 and 9 provide different visualizations of the same hierarchical clustering. Vegetation at the 1948 and unharvested fields are relatively homogenous while the 1957 field shows the most variation in vegetation composition, dispersing within the other groupings. This likely results from the greater diversity seen at the 1957 field as most species observed in this study were seen at least in this field (Figure 6). For example, *Polytrichum* spp. were most abundant at this field and rare otherwise.

K Means Clustering Analysis

Cluster 1 was comprised of 26 plots and predominated especially by white beak-rush (mean cover = 57.5%), *Drosera* spp. (mean cover = 4.51%), *Sphagnum* spp. (mean cover = 47.0%), bare ground (mean cover = 11.3%), and pooled water (mean cover = 2.1%) (Figure 10). Cluster 2 was comprised of 24 plots and predominated especially by Labrador team (mean cover = 18.7%), bog blueberry (mean cover = 42.4%), velvet leaf blueberry (mean cover = 1.9%), pine (mean cover = 8.7%), and *Cladonia* spp. (mean cover = 25.2%). Cluster 3 was composed of 30 plots and predominated especially by bog laurel (mean cover = 9.3%), bog rosemary (mean cover = 16.2%), bog cranberry (mean cover = 3.9%), *Eriophorum* spp. (mean cover = 23.6%), and *Cladina* spp. (mean cover = 4.0%).

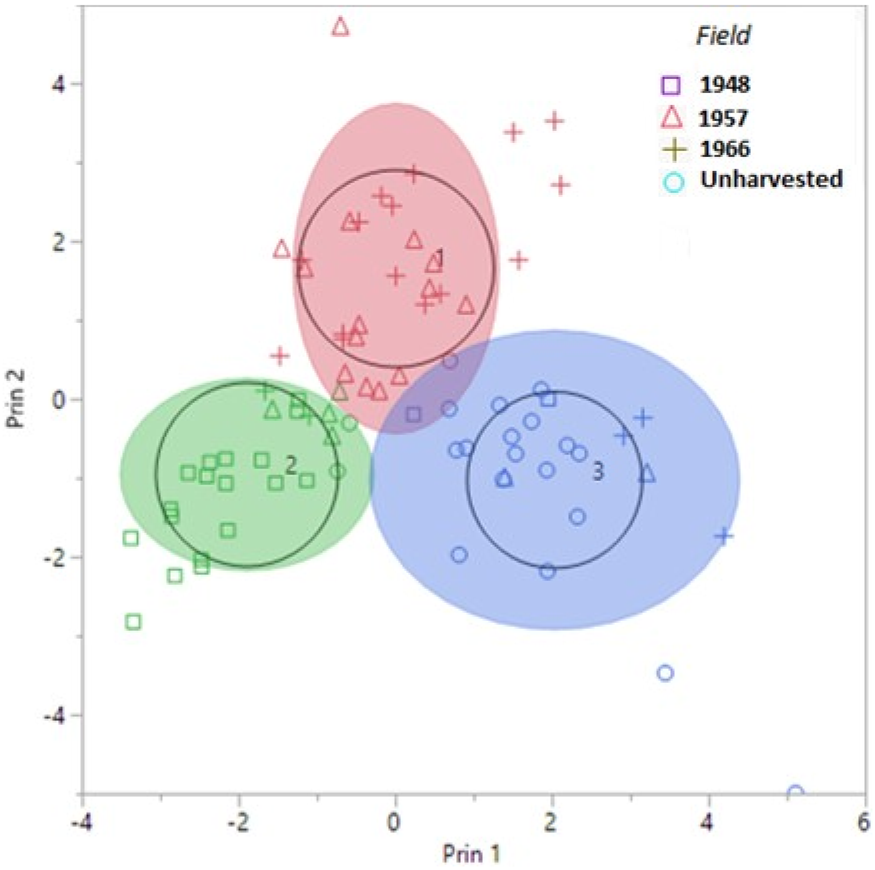


Figure 10. **K means clustering biplot demonstrating optimized three clusters based on similarities in the distribution of vegetation in Burns Bog in June and July, 2018 (n = 20 per field, shaded area represents 90% of observations in a given cluster).**

The three clusters generated by k means clustering seem to align with the complementary hierarchical clustering. Though the analysis determined eight optimal clusters, three prominent clusters are evident in Figure 9, comparable to Figure 10 and further verifying this classification.

Peat Humification

It is expected that degree of humification will increase with depth in peatlands (Barrett and Matmough, 2015). This was true in the 1948 and 1966 fields, while the 1957 field remained relatively consistent with depth, decreasing only slightly, and the unharvested field showed an unpredictable fluctuation (Figure 11, t-distribution, n-1=19 df). Comparison with depth to water table results seem consistent as the 1948 and 1966 fields showed an expected or acceptable depth to water table as well as predicted humification trends, versus the 1957 and unharvested fields, which show a deeper water table and unanticipated humification profile.

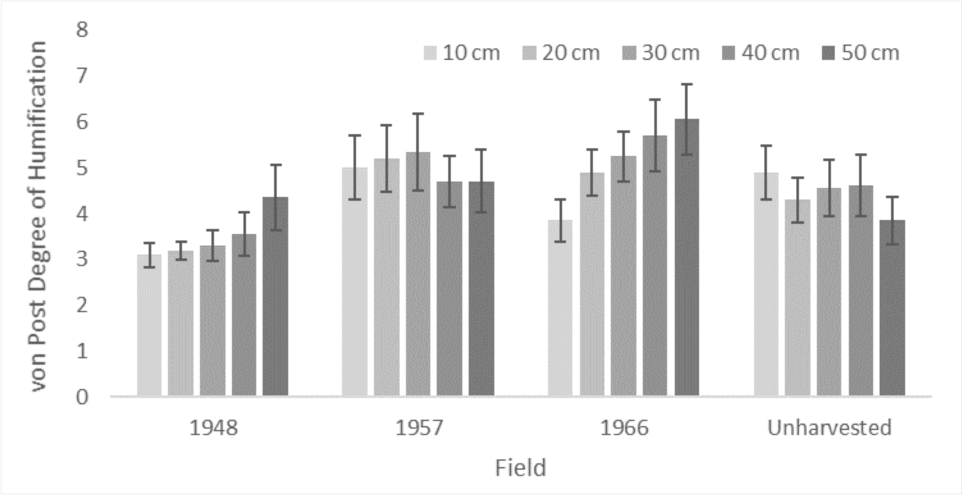
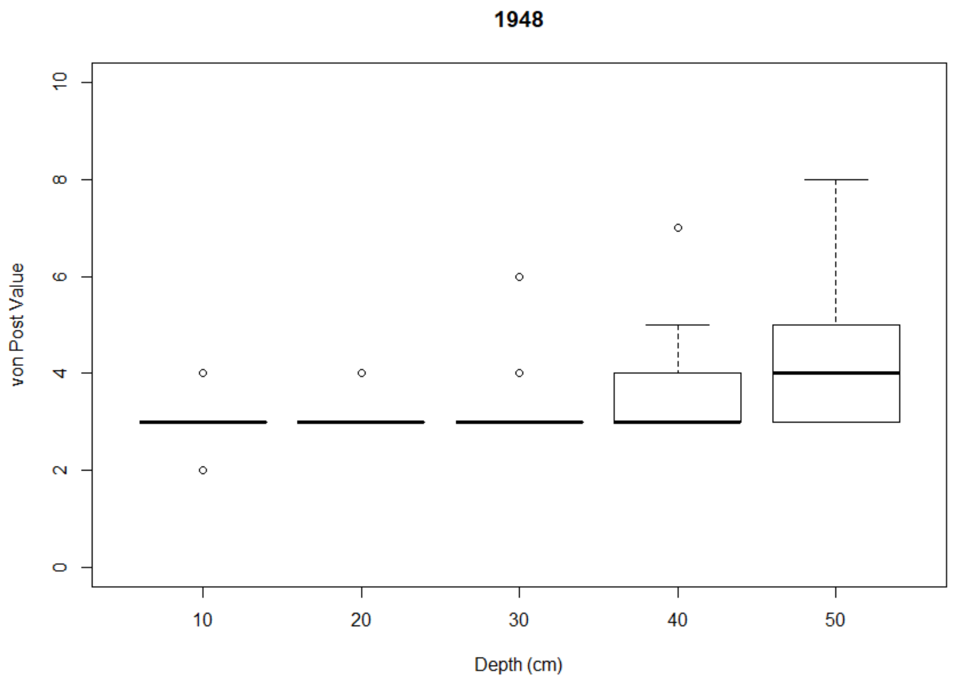


Figure 11. **Mean degree of peat humification in Burns Bog in June and July, 2018 (n = 20 per field, error bars represent 95% confidence intervals).**

H2 was the lowest level of decomposition observed and was seen at 0.1 m peat depth at the 1948 field (Appendix A, Table A-1, Figure A-1). H10 represents fully decomposed peat and was only recorded at the 1966 field at both 0.4 and 0.5 m depth. The variation of degree of humification values by depth and by field is best visualized by boxplots as the range of values is evident (Figure 12). The 1948 field is relatively homogenous with depth and consistently different from the other fields, as expected, but not most similar to the unharvested reference site (see Discussion).


1948
von Post Value
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2
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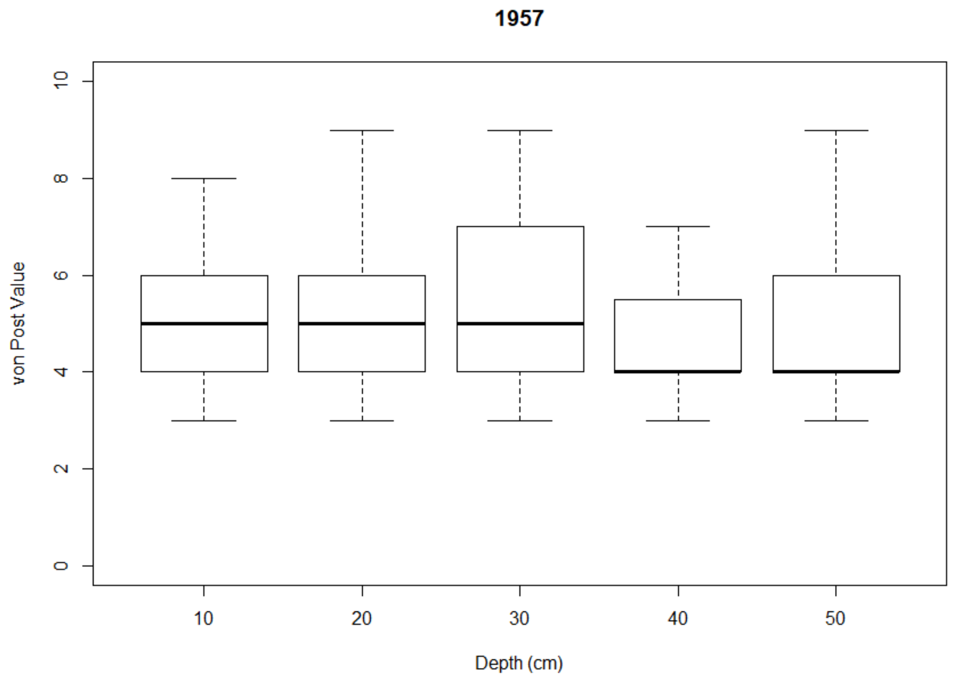
6
8
10
10
20
30
40
50
Depth (cm)


1957
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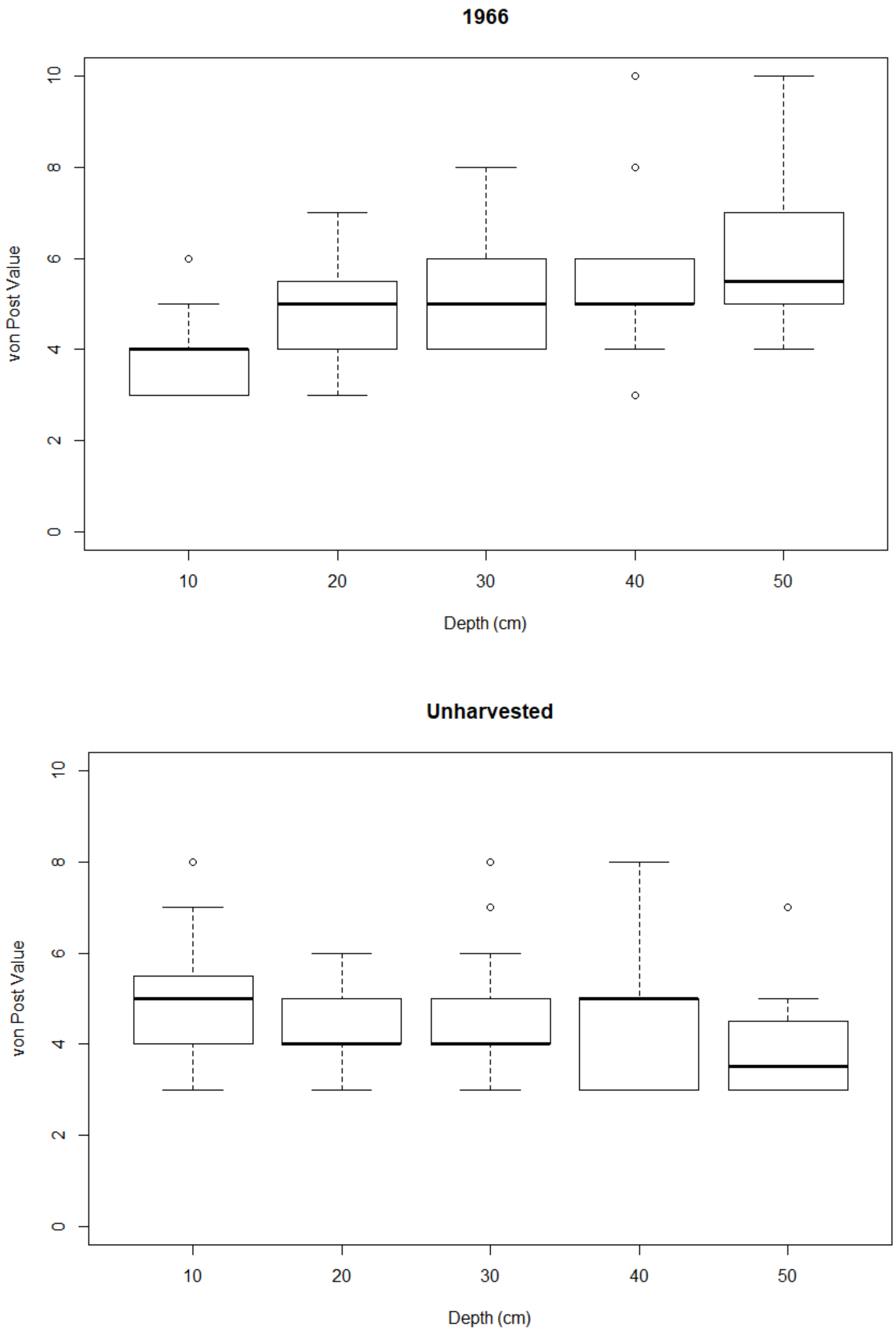

Figure 12. **Boxplots presenting degree of peat humification at specified depthsof three harvested and one unharvested field in Burns Bog in June and July, 2018 (n = 20 extracted peat cores per field).**

Investigative coring at the start, mid-point, and end of each transect suggested that maximum peat depth is greatest at the unharvested site at approximately 5 m, and approximately 3 m at the 1948 field, and 4 m at the 1957 and 1966 fields (Appendix A, Table A-2). It appears that at the 1948 and 1957 fields, degree of humification increased with depth before decreasing at 2 m (Appendix A, Figure A-2). At both the 1966 and unharvested fields, values fluctuated before strongly increasing at 2 m. This aspect was purely investigative and with only three sampling locations per field is not rigorous without a larger and more representative sampling design.

Microtopography

Table 3 presents the microtopography type of each plot. The unharvested field had the greatest number of hummocks (10), followed by the 1957 field (7). The 1948 and 1966 fields had a relatively small number of hummocks (3 each). The 1948 field was largely comprised of lawn, pooled water, and *Sphagnum* ponds. No pooled water was observed at the unharvested field.

Table 3. Number of plots belonging to each microtopography category in Burns Bog in June and July, 2018. (n = 20 per field)

Field	Hummock	Incomplete Hummock	Hummock-Hollow Combination	Hollow, Lawn, or Pool
1948	3	1	3	13
1957	7	3	2	8
1966	3	5	1	11
Unharvested	10	7	0	3

Influence of Environmental Variables

Figure 13 presents the RDA computation including vegetation percent cover and its topographic and humification constraints. Microtopography was again considered in categories of complete hummock, partial hummock, hummock-hollow combination, and other (e.g. complete hollow, lawn, pool). The adjusted R^2 was 0.21, indicating that the constraining variables in this computation account for 21% of the variation in vegetation percent cover. The first two axes explain 18% of the variation in the model (RDA axis 1 explains 14%, permutation test with 999 runs: F = 19.56, P = 0.001; RDA axis 2 explains 4%, permutation test with 999 runs: F = 5.21, P = 0.034). It is expected for these numbers to be small due the noise inherent in ecological data (i.e. random error or unmeasured and unpredictable variation) (Borcard et al., 2018). Additionally, without including water table data in this analysis, the proportion of variation explained will necessarily be small. We can therefore reject the null hypothesis of no relationship between the vegetation composition data and the matrix of explanatory variables.

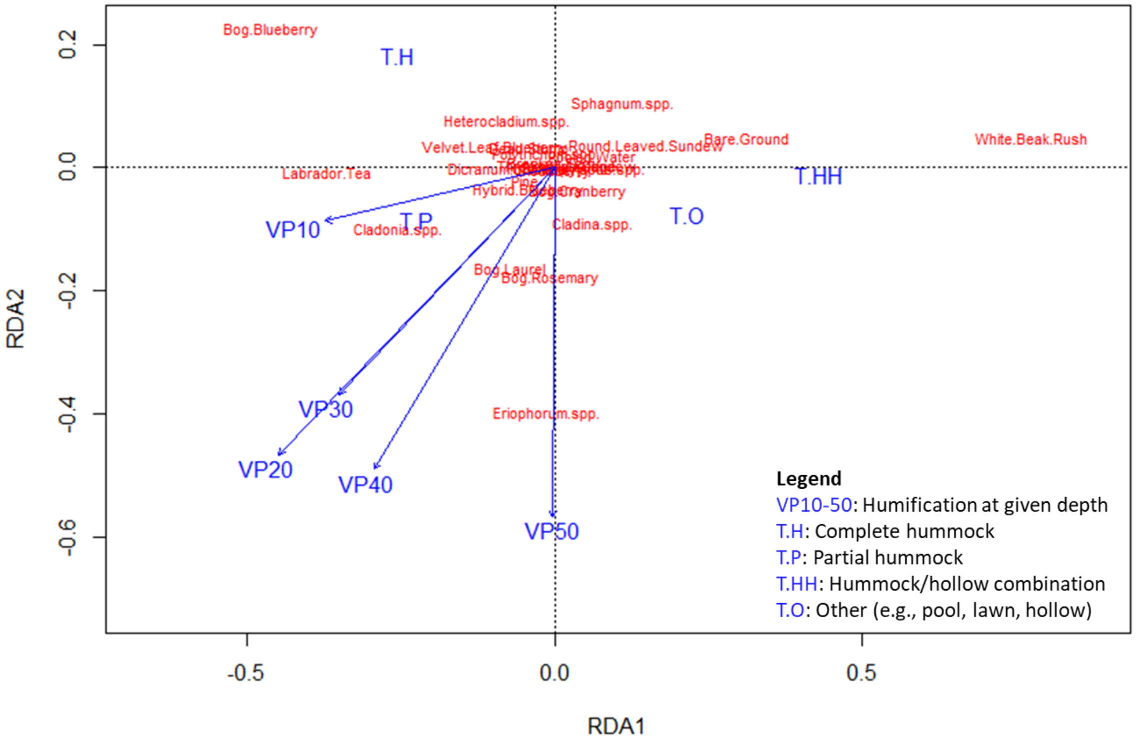

Figure 13. **RDA with PCA on vegetation percent cover and constrained by topography type and degree of humification in Burns Bog in June and July, 2018 (n = 20 per field). The x and y axes represent principle components one and two, respectively.**

Figure 13 demonstrates that, in this study, degree of humification had a larger influence on vegetation percent cover than did topography type. The distribution of *Eriophorum* spp., bog laurel, bog rosemary, lichens (i.e. *Cladonia* spp. and *Cladina* spp.), and Labrador tea was positively correlated with more strongly humified peat. Conversely, *Sphagnum* coverage was somewhat negatively correlated with strongly humified peat. Bog blueberry was positively correlated with hummocks, while white beak-rush was positively correlated with hummock-hollow combinations.

Linear regression was completed to compare *Sphagnum* distribution to topography type and degree of humification near the peat surface and the strongest results are presented below (Figures 14-16) (Eliminated analyses in Appendix A, Figures A-3 and A-4). Importantly, statistical significance (i.e., p-value ≤ 0.05) was not found in the majority of these comparisons.

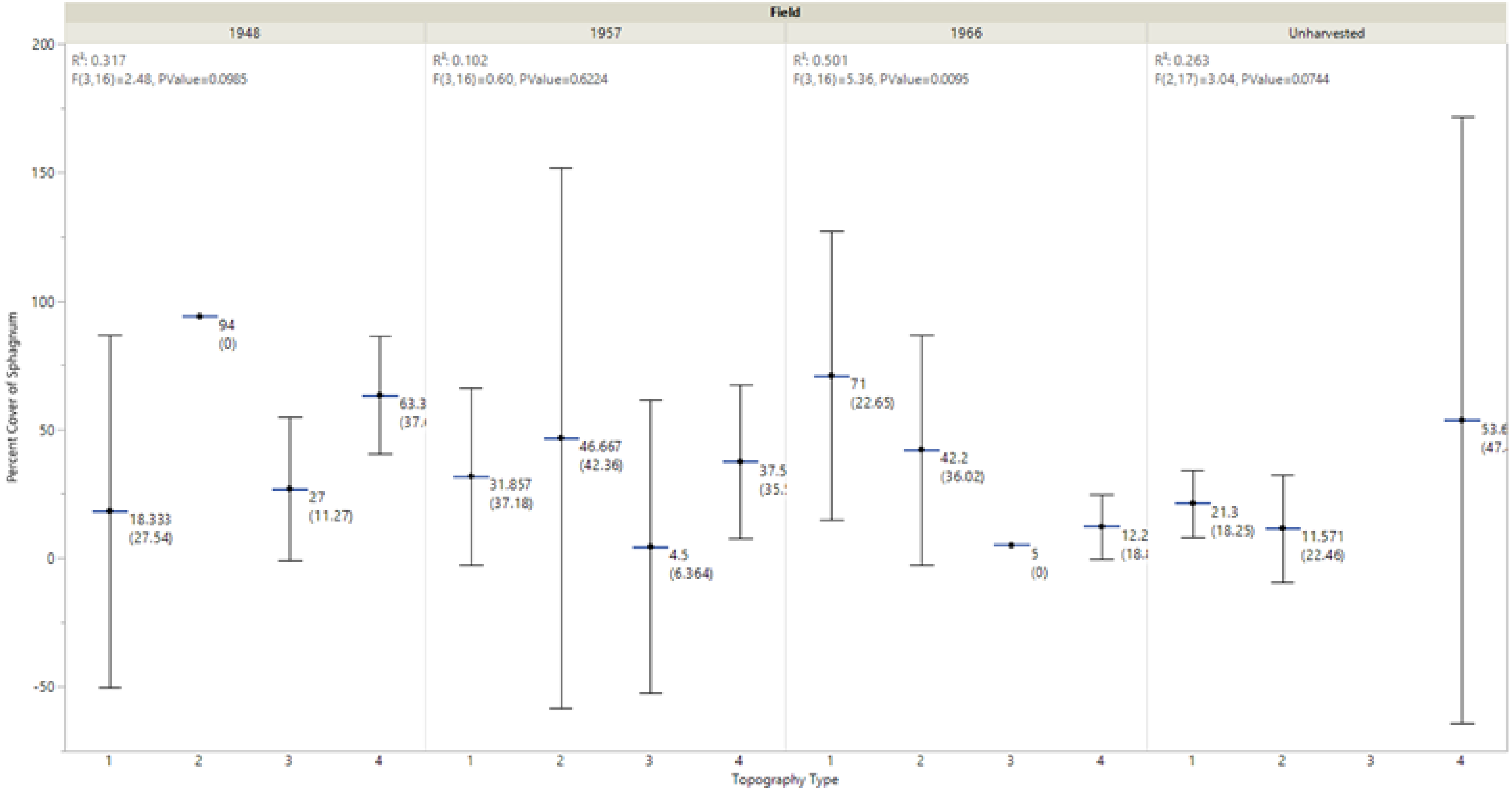

Figure 14. Linear regression comparing *Sphagnum* coverage to topography type in Burns Bog in June and July, 2018. Topography type 1 refers to a complete hummock, 2 refers to an incomplete hummock, 3 to a hummock-hollow combination, and 4 to to other (i.e. lawn, complete hollow, wet depression etc.) (n = 20 per field, unequal variances, 95% confidence intervals).

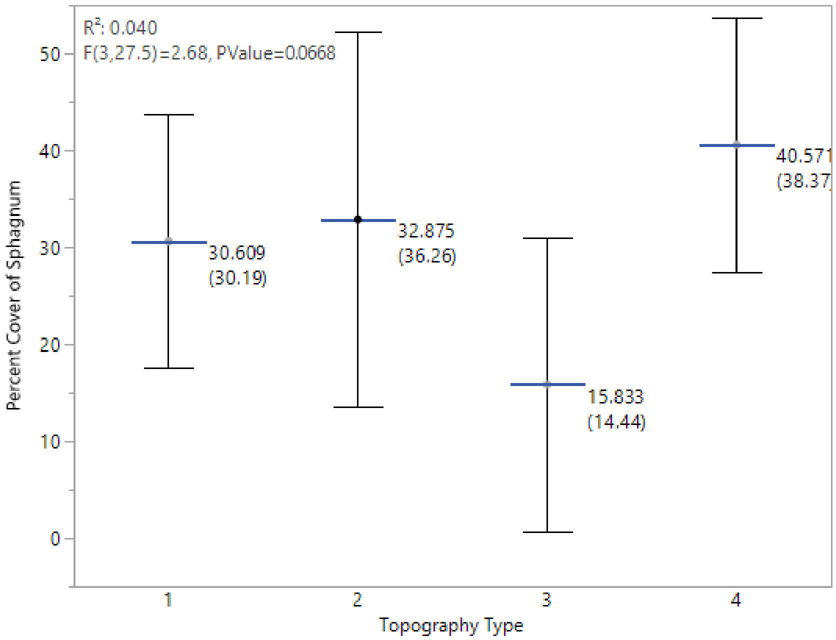

Figure 15. **Linear regression comparing *Sphagnum* coverage to topography type in Burns Bog in June and July, 2018. Topography type 1 refers to a complete hummock, 2 refers to an incomplete hummock, 3 to a hummock-hollow combination, and 4 to to other (i.e. lawn, complete hollow, wet depression etc.) (n = 20 per field, unequal variances, 95% confidence intervals).**

Topography type did not strongly influence the distribution of *Sphagnum* in this study (Figures 14 and 15). This was an unexpected finding of the RDA that was subsequently verified with regression (Figure 13). Across the study, *Sphagnum* coverage was greatest on hummocks and low depressions, while hummock-hollow combinations saw the smallest coverage by *Sphagnum*, though not quite statistically significant ($R^2 = 0.04$, $p = 0.067$) (Figure 15). The strongest correlation between topography type and *Sphagnum* coverage was seen at the 1966 field ($R^2 = 0.5$, $p = 0.0095$) (Figure 14). Regardless of high R^2 values for the remaining comparisons, p-values were not quite statistically significant.

Linear regression illustrated that *Sphagnum* distribution was not strongly influenced by topography type (Figures 15 and 16). When constrained by field, R^2 values were larger, indicating that site characteristics are influencing this distribution, as expected. Across all fields, *Sphagnum* percent cover was greater off hummocks (Figure 14). When constrained by site there was little variation between fields and only at the 1966 field was there a statistically significant correlation ($R^2 = 0.5$, $p = 0.0095$) (Figure 16). Here *Sphagnum* coverage was highest on hummocks.

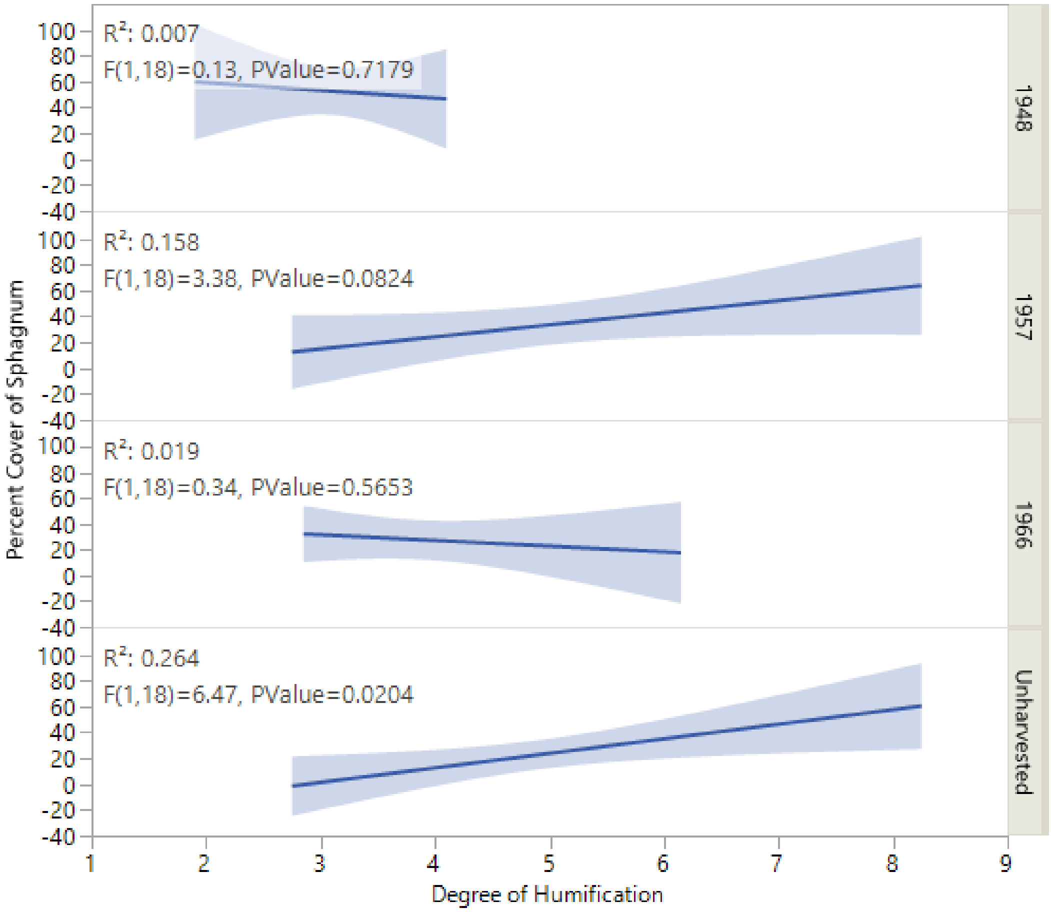

Figure 16. **Linear regression comparing humification at 0.1 m depth to *Sphagnum* coverage in Burns Bog in June and July, 2018. A von Post score of 1 represents undecomposed and 10 represents completely decomposed. (n = 20 per field, shading represents 95% confidence interval).**

RDA highlighted a stronger influence of degree of humification than topography type on vegetation percent cover. The only statistically significant correlation between humification and *Sphagnum* coverage was found at the unharvested field (R^2 = 0.264, p = 0.02) (Figure 16).

Linear model testing (i.e. multiple regression) was performed to simultaneously investigate the influence of multiple covariates (i.e. topography type ['complete hummock' versus 'other'], degree of humification at each depth, and field) on percent cover of specific species of interest. This analysis was performed to complement and verify RDA and quantify the correlation between specific interactions of note. There were very few significant findings from this analysis. However, multiple regression was able to predict cover by *Eriophorum* spp. was strongly influenced by the degree of humification at each depth. With the Bonferonni correction for multiple hypothesis testing reducing the significance level to 0.0167 (i.e. 0.5/3 predictor variables), the model was statistically significant at each peat depth tested (df = 76) (Table 4).

Table 4. Results of multiple regression model predicting the cover of *Eriophorum* spp. with predictor variables topography type, field, and degree of humification every 0.1 m in the top 0.5 m of peat collected in Burns Bog in June and July, 2018. Statistically significant results are in bold (p-value ≤ 0.0167)

	0.1 m	**0.2 m**	**0.3 m**	**0.4 m**	**0.5 m**
Adjusted R^2	0.28	0.25	0.21	0.21	0.25
Model P-value	**3.01e-06**	**1.41e-05**	**1.10e-4**	**1.00e-04**	**1.33e-05**
Species Estimate	3.59	-1.11	2.71	0.85	-7.54
			Predictor Estimate (p-value)		
Topography	-4.18 (0.22)	-6.78 (0.056)	-7.43 (0.045)	-4.64 (0.20)	-5.78 (0.10)
Field	-6.43 (**5.45e-05**)	-3.35 (0.019)	-3.15 (0.032)	-2.71 (0.066)	-0.63 (0.69)
Degree of Humification	5.64 (**1.00e-05**)	4.89 (**5.27e-05**)	3.77 (**4.95e-04**)	3.74 (**4.50e-04**)	4.40 (**4.92e-05**)

At 0.1 m peat depth, multiple regression explains 28% of the variation in cover by *Eriophorum* spp., which was negatively correlated with field and positively correlated with degree of humification (Table 4). Cover by this genus was highly varied, with low relative abundance in the 1948 and unharvested fields, and high relative abundance in the 1957 and 1966 fields. Multiple regression with the remaining peat depths did not indicate the same influence level by field. The results of this analysis revealed that cover by *Eriophorum* spp. and degree of humification are correlated.

Discussion

It is not feasible to expect complete ecological restoration as per the SERI (2004) definition as a longer timescale is required. The desired system of ombrotrophic raised bog is a climax ecosystem and thus requires time to re-establish, with more than one pathway possible (Money and Wheeler, 1999). The lag time in this study of ~40-60 years between cessation of harvest and implementation of restoration further implicates this trajectory. Price et al. (2003) note that restoration in this strict sense is likely only possible in the long term, ideally with recovery of some functionality in the short term.

Initially, it was expected that the 1948 field would be most similar to the reference area since it was harvested longest ago. However, this field was largely homogenous, which is not indicative of successful restoration (e.g. Bonsel and Sonneck, 2011). Though increasing time since harvest was expected to correspond with increased restoration success, the opposite might be true due to the lag between cessation of peat harvest and implementation of restoration activities. For example, Price (1997) investigated restoration in a Quebec bog-poor fen peatland and observed that time between disturbance and restoration is possibly the most important factor in ecological restoration following harvest. Further, investigation of site conditions at the unharvested field indicate that it is disturbed and does not function as a model site for comparison (see Study Limitations).

Investigation of the interactions of hydrology, vegetation composition, and peat accumulation and decomposition is crucial to understanding the restoration trajectory of a peatland. Potvin et al. (2015) investigated the impact both of a changing depth to water table and plant functional groups on vegetation cover, aboveground plant production, and peat subsidence. They found that lowering water table position (as per the projected impact of anthropogenic climate change) altered vegetation composition, productivity, and peat subsidence. By conducting a manipulation experiment, they were able to delineate the independent and cumulative influence of water table position and plant functional group. They found that water table position alone had the strongest influence on cover of *Sphagnum* and Ericaceae species. Additionally, *Polytrichum* spp. cover was influenced considerably by both depth to water table and plant functional groups. Finally, microtopography was influenced by both plant functional group and water table position by impacting subsidence and accumulation of peat. Of note, they observed that in the absence of sedges, shrubs responded even more positively to decreases in water table position. Whether the cause is release from competition or otherwise is unclear, and further research on this interaction is suggested. Through investigating the water table position, vegetation present, peat humification, and distribution of microtopography types in three harvested fields of Burns Bog, the restoration status and trajectory is better understood and early intervention made possible.

Hydrology

In an undisturbed raised bog, organic matter amasses in the form of peat, raising the water table via water storage in the acrotelm and hydrologically detaching *Sphagnum* peat from the surrounding system (Price et al., 2003). Even though the Atkins-Durbrow method saw the installation of fewer drainage ditches than other methods in Burns Bog (Hebda et al., 2000), drainage impacts still apply here. Even peat extraction alone removes the water storage capacity of the acrotelm and causes water table depression (Price et al., 2003), and is further intensified by concurrent drainage to ease peat harvest (Lindsay et al., 2014). Drainage and ditch blocking both alter hydrology in a similar manner to climate change, but with a more drastic and sudden change (Frolking et al., 2011).

Saturation over the long term prevents oxidation and reduces carbon loss from peat (Price, 2003). Damaged raised bogs typically exhibit considerable variability in water table position (Money and Wheeler, 1999). The decrease in hydraulic conductivity with peat drying is likely part of the self-regulation mechanism as it limits lateral seepage (Price, 2003; Howie and Hebda, 2018). Upon drying and consolidation, the largest pores (present closest to the surface) collapse first and drastically lower hydraulic conductivity, reducing lateral water loss (Price, 2003). By changing peat volume through mire breathing, a bog reduces hydraulic conductivity and lateral seepage with it, contributing to acrotelm function (Price, 2003). Price (2003) observed a decrease in hydraulic conductivity of two orders of magnitude with only a 1% decrease in peat volume. Upon rewetting, hydraulic conductivity increased, likely due to subsequent saturation of large pores.

Depth to water table measurements in this study illustrate that the water table is most restored in the 1948 field (Figure 4). Both here and at the 1966 field, the dry season position is not low enough to cause concern. Seasonal lows in the 1957 field, however, do approach the 0.4 m critical level and water table position has dropped in recent years, indicating that bog vegetation growth may be inhibited (Verry, 1988). Water table position at the unharvested site has also dropped in recent years and approaches this threshold. In 2014, other researchers in Burns Bog also observed a depth to water table range of 0.1 to -0.3 m (with negative measurements indicating flooding) in the area around the 1948 field and 0.14 to 0.31 m in the area around the unharvested reference area (Christen et al., 2016). As this group used the same piezometers and City of Delta data it is expected that results agree, but as their study covered a larger area, it indicates that fields in my research are representative of the larger area. As both the 1957 and unharvested fields approach the 0.4 m threshold during drought years, they are candidates for further hydrological restoration (Table 2). The 1957 field experienced annual water table depth lows of at least 0.3 m in 2013, 2015, and 2016, at which point monitoring unfortunately ceased. The unharvested field experienced the same annual lows from 2015 to present. Even though water table position is shallower at the 1948 and 1966 fields, the lowering trend in recent years applies here too, suggesting climatic, rather than site differences are responsible. Fluctuations in water table position were most pronounced at the harvested sites, as expected. Amplitude and frequency of water table oscillations is increased by peat harvest and drainage as the water storage and regulation capacity of the acrotelm is eliminated (Price et al., 2003). These fluctuations then alter redox potential and subsequently nutrient and carbon cycling (Artz, 2009). Other research covering a larger portion of Burns Bog also found this trend (Howie and Hebda, 2018). Proximity to drainage ditches is illustrated in Figure 2, but their varying influence cannot be quantified. The close proximity of the unharvested site to a small drainage ditch potentially contributed to disturbance observed here.

The yearly low water table position occurred in September, 2009, 2011, and 2012 in all fields (Table 2). In 2012, this annual low coincided with almost no precipitation in September (Figure 5). In 2009 and 2011 though, September saw a considerable increase in precipitation, potentially indicating issues with absorption after drought. For example, lichens and bryophytes have a hydrophobic surface after desiccation (Tessier and Boisvert, 1999; Howie and Hebda, 2018). As lichens are most abundant at the 1957 and unharvested fields, they may serve as an impediment to water absorption at the start of the wet season (Figure 7). Regional climate modeling predicts decreased precipitation in September, lengthening the summer drought period (Metro Vancouver, 2016) and further necessitating hydrological management and acrotelm restoration. Monitoring of depth to water table at the piezometers on harvested fields should resume due to the trend at the 1957 field towards the 0.4 m critical threshold and the observation of increasing drought severity (Figure 4). Though City of Delta measurements in harvested sites ceased in 2016, measurements I collected in 2018 suggest that these trends are still occurring. I recommend resuming measurements at these sites to enable monitoring and early intervention (Table 1).

Howie and Hebda (2018) investigated mire breathing in Burns Bog and observed a smaller depth to water table in harvested compared to unharvested sites in summer drought (0.24 m mean maximum depth compared to 0.30 m). This is expected as the piezometers used in this study were a subset of all those used in Howie and Hebda (2018), though my research was restricted to three fields harvested with the Atkins-Durbrow Hydropeat method and one reference site that was considered unharvested and undisturbed (Howie and Hebda, 2018; Howie, pers. comm.). Additionally, Howie and Hebda (2018) observed that under wet conditions, depth to water table was less at harvested sites than unharvested (- 0.072 m compared to - 0.04 m). At unharvested fields, the authors also observed most extensive mire breathing where conditions were wettest and water table most stable. Conversely, at harvested sites they observed that mire breathing had the greatest amplitude where water table fluctuations were greatest and depth to water table smallest. The authors note that in Burns Bog, an unharvested area might have been irrevocably altered by drainage, resulting in decreased or eliminated peat elasticity and mire breathing capacity. They conclude that minimizing the lag between harvest and restoration intervention is crucial to regeneration and suggest measuring this property as an indicator of success. For example, the species capable of colonizing a drained peatland are also those with a deep root system, enabling deeper access to water and thus further lowering the water table through evaportranspiration (Lindsay et al., 2014). This finding is corroborated by Price (1997), who observed that the length of time between disturbance and restoration is most important. This finding is very relevant to this study as most successful restoration was expected in the 1948 field, but this factor could lead to the opposite in reality and thus adds to the rationale for assisting with revegetation of this field (see Recommendations).

Ditch blocking can restore the quantity of available water; however, storage capacity is still lacking if a functioning acrotelm was not been restored (Price et al., 2003). Additionally, ditch blocking can result in oversaturation and increased surface water and resultant overland flow (i.e. runoff) (Menberu et al., 2018). This could see more water lost during heavy rainfall events. Previously drained peat typically absorbs precipitation more rapidly than undisturbed peat, but also loses it very rapidly due to inferior storage (Menberu et al., 2018). As the interior of Burns Bog is considered to have a restored water table (Howie, pers. comm.), acrotelm establishment is now crucial to climate resiliency as further water table elevation is unlikely without increased summer precipitation. This is further complicated by the observation that extended summer drought can result in pore water pressures below the critical limit for uptake by plants of -100 mb (Price et al., 2003) (further discussed in *Spagnum* Distribution). Extended drought contributes to seasonal subsidence resulting from water table drawdown, further reducing water storage capacity (Price and Schlotzhauer, 1999) with additional pore collapse decreasing hydraulic conductivity and inhibiting water transport to *Sphagnum* (Price et al., 2003). Retention of fall precipitation is thus crucial to withstand summer precipitation lows (Price et al., 2003) in the face of predicted decreases in the region (Metro Vancouver, 2016). Without increased retention, the water table may drop further with a changing climate prior to successful acrotelm reestablishment. Surface recontouring to create terraces or polders within bunds is one option to increase water retention (Price et al., 2003). A less invasive option is the encouragement of nurse species like *Eriorophorum* spp. to shade recolonizing *Sphagnum* and decrease water lost to evapotranspiration via plant use and evaporation due to radiation (Price et al., 2003) (see Nurse Species).

As hydrology was not incorporated into multivariate analysis, we cannot glean its contribution to the bigger picture. It is accepted that depth to water table has a strong influence on cover of *Sphagnum* and Ericaceae (Potvin et al., 2014). In future, perhaps soil moisture can be measured at each plot (in addition to depth to water table measurements) to enable comparison of hydrology within multivariate statistical analysis (Price, 2003; Potvin et al., 2014). Continuous measurements, as opposed to the current monthly manual measurements, could aid in elucidating fine-scale temporal variation as observed at various peat depths in other studies (e.g. Price, 2003).

Influence of Depth to Water Table on Greenhouse Gas Emissions

A major motivation behind peatland restoration is maximization of carbon sequestration function as a climate change mitigation strategy. Ditch blocking and subsequent rewetting, though, can cause a temporary increase in CH_4 emissions as the proportion of submerged soil increases (Evans et al., 2014). The extent and duration of this increase is determined by water table position, drainage history, site management (Maljanen et al., 2010), vegetation present, and thickness of both saturated and unsaturated peat (Bubier, 1995). High summer temperatures increase CH_4 production and cause ebullition of dissolved gases into bubbles (Price, 2003). This extracts pore water, drying soil and exposing it to further consolidation as summer drought depresses the water table (Price, 2003). Rewetted bogs act as major CH_4 emitters in the first 5-20 years (Strack and Waddington, 2012). After a number of decades, greenhouse gas sink function is generally restored (Strack and Waddington, 2012). CH_4 emissions increase linearly up to a depth to water table of 0.2 m; when depth to water table is greater than 0.2 m, CH_4 emissions are negligible (Couwenberg et al., 2011; Frolking et al., 2011; Evans et al., 2014). As water table position at the 1948 field reaches this height in the wet season, considerable CH_4 emissions are predicted (Figure 4).

Christen et al. (2016) investigated greenhouse gas emissions in Burns Bog, including in the two vegetation types investigated in this study (i.e. “White beak-rush – *Sphagnum*” at harvested sites and “Pine–*Sphagnum* – Low Shrub” at unharvested site). The authors found that CH_4 was the predominant greenhouse gas emitted at a number of sites within Burns Bog, with the 1948 field included in the category of highest emitters. Within Burns Bog, they observed that CH_4 emissions were reduced almost three-fold below the 0.2 m water table depth emission threshold. In 2014 they noted that while Burns Bog is a large CH_4 emitter in these sites, it is not a strong CO_2 sink and predict restoration of this function after 10-20 years (Christen et al., 2014). They note that seasonal water table management may reduce these emissions in the high CH_4 production 5-20-year rewetting time period (Christen et al., 2014).

Vegetation present also influences the production and release of CH_4. In flooding conditions, emissions are increased by the presence of young vegetation and especially those with an abundance of aerenchymous tissue (Christen et al., 2016). Aerenchyma acts as a shunt for passing CH_4 from saturated deep peat to the atmosphere (Couwenberg et al., 2011). Sedges especially contain this tissue, explaining their persistence in saturated hollow and lawn conditions (Potvin et al., 2015) and these were most prevalent at the 1948 field (Figure 7). The combination of water table position in the top 0.2 m of peat and the abundance of aerenchymous species here make it a strong emitter and a candidate for further restoration intervention (See Aquatic *Sphagnum*).

CO_2 emissions increase linearly with depths to water table up to 0.5 m; CO_2 sink function is maintained with water table positions above 0.065 m and reduced when depth to water table is below this threshold (Couwenberg et al., 2011). Rewetting can enable CO_2 sequestration within three years as peat oxidation is reduced and vegetation, including *Eriophorum* spp., recolonizes (Tuittila et al., 1999). Greater control of water table position during rewetting should be investigated in peatland restoration projects to better understand this mechanism and reduce greenhouse gas emissions (Christen et al., 2016), with acknowledgement of the considerable challenge a 3,000 ha such as Burns Bog poses.

Vegetation

Vegetation can serve as an indicator of environmental health but takes a number of years to reflect changes (Couwenberg et al., 2011). Minimally invasive ditch blocking activities can redirect the revegetation trajectory towards a more desirable assemblage than that of passive revegetation without restoration (Gonzalez et al., 2014a). It is expected that without intervention, recolonization will occur within the first years after cessation of harvest and revegetated principally by vascular vegetation (Gonzalez et al., 2014a). Bog restoration should see a decrease in trees and non-bog species and an increase in nurse species (e.g. *Polytrichum* spp.) (Price et al., 2003) and *Sphagnum* as water tables rise and vegetation shifts to a more bog-like community (Hebda et al., 2000). Gonzalez et al. (2014a) cite an increase in dry-intolerant species, including *Eriophorum* spp. and *Polytrichum strictum,* and a decrease of dry-tolerating species, including *Cladina rangiferina,* as evidence of the effectiveness of rewetting. Renou-Wilson et al. (2019) evaluated rewetting efforts in two Irish raised bogs with success indicated again by the presence of dry-intolerant species and also a water table position at or near the surface for most of the study.

Sphagnum are generally the predominant vegetation in ombrotrophic peatlands and are crucial to the water budget (Robroek et al., 2007b). Species die off tends to occur more quickly than colonization (Couwenberg et al., 2011) and tree die off after rewetting typically indicates restoration success (Hebda et al., 2000); however, very few trees were present in the harvested study sites and in the absence of pre-restoration baseline data, this is not an ideal indicator. Further, in natural bogs, tree encroachment is slow and generally creates positive landscape heterogeneity and microclimates (Gunnarsson et al., 2002). Lower water tables resulting from drainage encourage tree encroachment, further perpetuated by their increased water usage through evapotranspiration, thus increasing water table draw down (Dyderkshi et al., 2016). Additionally, increasing shade inhibits shade-intolerant bog species like *Polytrichum* spp. (Price et al., 2003). Therefore, monitoring tree encroachment in dry areas is advised. Without rewetting, tall and dense Ericaceae colonize first and *Sphagnum* coverage may remain below 10% (Gonzalez et al., 2014a). Ditch blocking is most effective when applied immediately after abandonment, otherwise Ericaceae shrubs can thrive, with 65% cover indicating arrival at an alternative stable state (Gonzalez et al., 2014a). This level of coverage results in litter amounts too excessive for *Sphagnum* growth due to the shade cast (Gonzalez et al. 2014a) and was observed at the 1966 and unharvested fields, potentially contributing to low observed S*phagnum* coverage here (Figure 7).

Sphagnum coverage was most extensive in the 1948 field (Figure 6). As this site was harvested longest ago, this was expected. However, a very shallow water table (Figure 4) and a lack of topographic variation (Table 3) hint at a potential alternative stable state and disrupted restoration here (see Aquatic *Sphagnum*). Christen et al. (2016) also noted that this area of Burns Bog is characterized by a small number of hummocks, colonized by typical bog shrub species, and a prevalence of depressions with incomplete vegetation cover. Interestingly, the unharvested reference site had the lowest observed relative abundance of *Sphagnum* and, when coupled with the deeper water table and more observed drought-tolerant vegetation here, this field is problematic as a reference site (see Study Limitations). In response to a low or fluctuating water table, passive revegetation is typically by dry heath vegetation and only patches, if any, of *Sphagnum* (Money and Wheeler, 1999), which corroborates this observed distribution.

Increased Ericaceae abundance is sometimes used to indicate healthy peatland, but often the opposite can be true (Lindsay et al., 2014). This group contributes little to peat-formation and tends to thrive after drainage (Lindsay et al. 2014). Some species in this group are more positively regarded than others, for example salal is sometimes considered invasive within Burns Bog due to its rapid spread (Hebda et al., 2000). It therefore makes sense that bog blueberry and salal were most prevalent at the sites with the greatest depth to water table (i.e. the 1957 and unharvested fields) (Figure 6). Bog blueberry is also a relatively shallowly rooted and productive species that has similar access to nutrients as *Sphagnum* (Malmer et al., 1994)*,* potentially outcompeting this species at the unharvested field.

Labrador tea was most prevalent at the unharvested site while bog laurel was least abundant here and at the 1948 site (Figure 6). Velvet leaf blueberry is a common bog species but was found in low abundance across the study. Moore et al. (2002) found Labrador tea and velvet leaf blueberry did not show a consistent trend in productivity with depth to water table, while a species similar to bog laurel, *Kalmia angustifolia,* was most productive where water table position was 0.5-0.6 m or 0.3-0.4 m below the surface.

Cloudberry is aerenchyma-rich and has a deep rooting system, enabling uptake from anaerobic layers (Malmer et al., 1994) but was observed in very small abundance and only at the 1966 field (Figure 6). It was also seen at the 1957 field but did not occur within any study plots.

There are typically more vascular plants on hummocks than hollows (Malmer et al., 1994), for example dwarf Ericaceae shrubs are typically restricted to hollows and *Eriorphorum* spp. sedges are abundant on lawns and slight hummocks (Hogg, 1993). Generally, hummock shrubs have long-lived above ground biomass and adventitious roots (Malmer et al., 1994). Others have observed a more variable relationship between sedges (including *Eriophorum* spp.) and water table position, likely also influenced by competition from other vascular species and bryophytes (Potvin et al., 2015), even noting the common presence of *Eriorphorum vaginatum* on hummocks (Malmer et al., 1994).

Sedges (comprised of white beak-rush, *Eriophorum* spp., and three-way sedge in this study) were most abundant at the 1948 field and least abundant at the unharvested site (Figure 7). This is likely a result of water table position as Moore et al. (2002) observed that sedge productivity (*Carex* spp. and *Eriophorum* spp. in their study) strongly increased with water table positions above 0.3 m (Moore et al., 2002). White beak-rush was present in moderate abundance at all sites except the 1948 field where it was the predominant species (Figure 6). The 1957 field has the greatest abundance of lichens and Ericaceae shrubs, indicating insufficient moisture here (Wallen et al., 1992; Lindsay et al., 2014).

Sparse lawns predominated by sedges are common in peatlands (Malmer et al., 1994). In this study, these areas were predominated by white beak-rush and *Eriophorum* species. Vegetation in these areas tends to be highly aerenchymous and deep rooted (Malmer et al. 1994). Bog rosemary and bog cranberry are also commonly found in these zones (Malmer et al., 1994). Lawn and depressions were most common at the 1948 field (Table 3).

Clustering analysis resulted in three general categories of vegetation distribution (Figures 8-10). Clustering analyses reveal that collective vegetation differs the most at the unharvested site, especially evidenced by the root placement of Figure 9. The observation of three major clusters and relative similarity in vegetation distribution at both the 1957 and 1966 fields verifies the same observation from Figure 7. *Sphagnum* spp.*, Drosera* spp.*,* and white beak-rush predominate cluster one, and this is largely determined by study plots in the 1948 field (Figure 6). These species require the shallow water table observed at this field (Figure 4). Ericaceae, including Labrador tea and bog blueberry, predominate cluster two, largely comprised of plots at the unharvested field. Here the depressed water table and advanced successional stage due to lack of peat extraction likely influence this observation. Small, herbaceous species, including bog rosemary and bog cranberry, and also *Eriophorum* spp. predominate cluster three. The water table position varies between these two fields (i.e., the 1957 field approaches the moisture threshold that inhibits bog vegetation growth), so the observation of similar vegetation was surprising. Interestingly, Gonzalez et al. (2013) found that cover of vegetation species that indicate restoration success varied only minorly between different restoration outcomes, making evaluation of restoration status difficult. The opposite was observed in this study, evidenced by these distinct clusters computed by opposing methods.

Across the study, hummock-hollow combinations saw the smallest coverage by *Sphagnum*, though shy of statistical significance (p = 0.067) (Figure 15). This is likely due to the disproportionate abundance of this topography type at the 1948 field while it is rare elsewhere. When computed by site, this topography type at this field did not see a considerable increase in *Sphagnum* coverage, supporting the recommendation for intervention here. The findings of this study suggest that the very shallow water table here is not hospitable to a diverse *Sphagnum* community or colonization has been limited by proximity to source terrestrial *Sphagnum* propagules. Nurse species including *Eriophorum* spp. and *Polytrichum* spp. are also low here, potentially warranting the manual introduction of both nurse species and hummock-forming *Sphagnum* spp..

Bogs are typically seen as stable, perhaps due to the slow growth rate of *Sphagnum*. However, Gunnarsson et al. (2002) investigated vegetation changes over 40 years in a natural Swedish bog and found dynamic changes in vegetation composition. For example, sections with the lowest pH saw a decrease over the study as well as a concurrent increase in certain *Sphagnum* spp. The abundance of stunted shrubs and hummock moss species also increased. This is a rare study that provides insight into natural patterns in bog development, despite its characterization as a climax ecosystem (Money and Wheeler, 1999). This implies reference conditions in bogs are dynamic and thus comparison to literature values is inappropriate and an ideal threshold proportion of *Sphagnum* spp. does not exist. Long-term monitoring of *Sphagnum* growth is thus preferable to comparison to reference conditions alone. Monitoring of the rate of *Sphagnum* growth in some unharvested areas of Burns Bog is already conducted by Metro Vancouver and may be appropriate now in harvested fields to monitor the restoration trajectory. Gonzalez et al. (2014a) also caution the comparisons between rewet and unrestored sites and instead suggest that regeneration of function should serve as the metric of success.

Years are required before ditch blocking regenerates bog hydrology, worsened when remaining peat is strongly humified as this reduces water storage (Price et al., 2003). Harvest method, sufficient available moisture, pH, light conditions, nutrient availability, and the proximity to source populations of bog vegetation propagules are the primary factors in determining the restoration trajectory of harvested bogs (Money and Wheeler, 1999; Daigle and Gatreau-Daigle, 2001; Dyderski et al., 2016). The lag in Burns Bog between cessation of harvest and implementation of ditch blocking efforts (i.e. ~40-60 years) likely resulted in a highly decomposed peat mass from oxidation and early colonization by drought-tolerant species (Daigle and Gatreau-Daigle, 2001). Though ditch blocking did not begin until the 21st century, beaver activity in Burns Bog began to reverse drainage shortly after cessation of harvest, but to an unmeasured extent (Howie, pers. comm.). Revegetation likely competed with and inhibited recolonization by *Sphagnum* (Lindsay et al., 2014). Post peat extraction and abandonment, remnant peat loses its self-regulation function, further depressing water table position and compressing water storage pores (Price et al., 2003). As aerobic microbes establish due to the now oxidative conditions, *Sphagnum* recolonization by propagules can be further inhibited (Price et al., 2003). These exposed peat remnants can also be nutrient bare, as opposed to a functioning natural acrotelm with cycling resulting in an ideal proportion of available nutrients (Money and Wheeler, 1999).

Andersen et al. (2013) determined that some trees and shrubs can circumvent bulk density, moisture, and degree of humification of peat by manipulating soil microbe diversity, at least close to the surface. This is of note because they observed that natural and unrestored harvested peatlands are more similar in terms of microbial composition than restored harvested peatlands.

According to RDA, 21% of the variation in vegetation distribution was explained by topography type and degree of humification in the top 0.5 m of peat (Figure 13). Incorporating soil moisture in future analysis would likely increase the proportion of variation explained by environmental variables. Gonzalez et al. (2014a) also found through RDA that environmental variables did not explain a large percentage of the variation in vegetation recolonization (i.e. 14%) in their investigation of bog rewetting. Stronger humification was positively correlated with increased abundance of Labrador tea, *Cladonia* spp., bog laurel, bog rosemary, and *Eriophorum* spp. and negatively correlated with *Sphagnum* spp. and white beak-rush. This is also influenced by depth to water table as the shallower water table of the 1948 field results in a larger proportion of submerged (and thus anaerobic) peat (Figure 4).

***Sphagnum* Distribution**

Changing abundance and distribution of *Sphagnum* spp. influences bog functionality, necessitating a species approach to assessing restoration status (Robroek et al., 2007b). Identification of *Sphagnum* to species was desired but not possible in this study due to the inherent complexity and resources required. The species of *Sphagnum* present also influences peat humification (Hogg, 1993). Due to their vascular tissue, trees, shrubs and, herbaceous species can achieve a height advantage and shade bryophytes, while *Sphagnum* capitula enable uptake of water and nutrients across the whole plant surface (Malmer et al., 1994; Dyderski et al., 2016). Therefore, under moist conditions*,* *Sphagnum* can outcompete vascular species, and vice versa in drought. Herbaceous species like *Drosera* spp. and bog cranberry have relatively shallow roots and thus rely on the water holding capacity of adjacent *Sphagnum* (Malmer et al., 1994). The presence of *Drosera* spp. indicates a suitable raised-bog hydrological regime, including low pH conditions (Renou-Wilson et al., 2019).

Malmer et al. (1994) noted that moderate shading of *Sphagnum* (i.e. not resulting in more than a 50% reduction in photosynthetic activity) tends to give rise to a looser moss carpet. While this increases surface area, it can serve as an impediment to water transport. Malmer et al. importantly note, however, that with increased vascular productivity, *Sphagnum* may be limited more by above-ground litter than directly by the plant itself. This is worth noting as considerably more litter was observed on hummocks in the unharvested field compared to the harvested fields. This is likely due to the greater proportion of trees, especially older trees, at the unharvested site (Figure 7). Additionally, the harvest method investigated by this study (i.e. Atkins-Durbrow Hydropeat) involved the removal of trees and surface vegetation, resulting in an earlier successional stage. Therefore, the trees at the unharvested site and their needle fall is likely adding to the shade cast on *Sphagnum* here (Lindsay et al., 2014).

Large depths to water table can also increase soil N and P, which can be taken up by vascular plants (Malmer et al., 1994). The influence of *Sphagnum* on vascular vegetation, rather than vice versa, is more important for peat function (Malmer et al., 1994). Capitulum water content determines photosynthetic ability (Moore et al., 2002). Both excessive and insufficient water content can decrease photosynthesis and thus productivity (Schipperges and Rydin, 1998), but vary by species in response to drying (Andrus, 1986). Therefore, identifying the *Sphagnum* spp. present can elucidate response to increased wet season flooding and dry season drought.

Sphagnum spp. differ in their resistance to desiccation, another important factor in *Sphagnum* distribution in restoration (Andrus, 1986). For example, a species like *S. nemoreum* grows in a dense and compact colony, and thus is more resistant to desiccation (Andrus, 1986). Further, this species engineers its own ideal environment by slowly contributing to hummock formation (Andrus, 1986). In general, hummock *Sphagnum* spp. can withstand this greater distance to the water table because of their superior internal water transport system (Robroek et al., 2007b). Hummock species can typically survive on hollows better than vice versa (Robroek et al., 2007b).

Sphagnum is a geologically young genus that evolved alongside vascular vegetation, in competition for light, nutrients, and water (van Breemen, 1995). Further, this genus can propagate clonally, resulting in a large, long-lived patch (van Breemen, 1995). This reproductive form necessitates phenotypic plasticity to withstand environmental change. For example, Robroek et al. (2007b) investigated the persistence of hollow *Sphagnum* spp. on hummocks and determined that lateral hummock water transport (LHWT) and precipitation determine their success here. The higher water content of hummock species benefits adjacent hollow species, enabling their persistence. This transport is only relevant on high hummocks and after heavy rain. Therefore, heavy precipitation in Metro Vancouver would enable these hollow species to survive on hummocks in the fall and winter but likely not in the dry summer months (Metro Vancouver, 2016).

Schipperges and Rydin (1998) assessed the photosynthetic response of five *Sphagnum* spp. to various intervals of desiccation in the lab. They found that after a period of desiccation, photosynthetic capacity is maintained only if capitula water content does not fall below 10%. They observed that connection between capitulum and plant base is the key to survival. Completely desiccated samples did not recover the capacity for photosynthesis. As expected, *Sphagnum* spp. differed in their ability to resist desiccation, with hummock species showing more tolerance due to greater capillary water uptake. Importantly, species that grew more closely together withstood dry conditions longer than larger, more isolated species, but not indefinitely. Therefore, knowledge of the species diversity, distribution, and growth habit of *Sphagnum* spp. in Burns Bog would elucidate resilience to reductions in summer precipitation.

Sphagnum has both chlorophyllous and hyaline cells. Hyaline cells comprise about 80% of a *Sphagnum* plant, with chlorophyllous cells positioned between, and are responsible for water and nutrient transport due to their hollow pores (van Breemen, 1995). Leaves can live for years, at which point shade cast from vertical growth causes mortality (van Breemen, 1995). Pore water pressure measurements might be warranted as pressures below -100 mb cause hyaline cells to drain, preventing water uptake (Hayward and Clymo, 1982; Price et al., 2003). In natural peatlands, pore water pressure generally is not an issue but in harvested bogs this can be a limiting factor (Price et al., 2003). Hydraulic conductivity and lateral seepage are reduced by drainage-induced consolidation and rewetting-induced increases in CH_4 production (Price et al., 2003). This can result in a layer of new *Sphagnum*, increasing soil moisture and pore water pressure (Price et al., 2003). It may still take decades to restore hydrological and ecological functions, though expedited by water table intervention and active revegetation (Price et al., 2003).

Aquatic Sphagnum

Sphagnum coverage was greatest at the 1948 field, but the landscape here was the most homogenous (Table 3). Only three complete hummocks were observed, and instead the majority of this field was composed of lawn and wet depressions and *Sphagnum* here was largely dispersed on floating rafts. This high abundance of *Sphagnum* suggests that restoration here has progressed the furthest. For example, a study by Gunnarsson et al. (2002) investigating vegetation changes over 40 years in a natural Swedish bog revealed that *Sphagnum* continued to expand, even in this natural system. However, water table position is highest and flooding most extreme here (Figure 4). Though *Sphagnum* were not able to be identified to species in this study, it appears possible that aquatic *Sphagnum* are prevalent here and there is potential that an alternative stable state has been reached. This observation is not necessarily negative but can serve as a starting point to facilitate colonization by terrestrial *Sphagnum* species if source populations are in close proximity.

Couwenberg et al. (2011) observed that very wet hollows were characterized by abundant aquatic *Sphagnum*, especially *Sphagnum cuspidatum*. Additionally, research by Gonzalez et al. (2014a) in abandoned block-cut ombrotrophic peatlands in Eastern Canada post rewetting found increased coverage ranging from 17 to 54 times more aquatic *Sphagnum* (e.g. *S. fallax* and *S. cuspidatum*) after rewetting compared to little or no presence in unharvested reference sites. Further, they found non-aquatic S*phagnum* spp. coverage did not significantly increase. For example, they found that *S. fuscum, S. magellanicum,* and *S. rubellum* were considerably less abundant than at reference sites. Renou-Wilson et al. (2019) used the observation of *S. papillosum* and *S. magellanicum* as indication that revegetation is on the desired trajectory towards raised bog species assemblage. Even though identification to species was not possible, it appears possible that *S. cuspidatum* and *S. fallax* may predominate the 1948 field and recolonization by *S. fuscum, S. magellanicum,* and *S. rubellum* has not yet occurred.

Reference conditions for pools in natural bogs are highly varied; pools can be colonized by abundant *Sphagnum*, show *Sphagnum* dieback, or not be colonized in the first place (Money and Wheeler, 1999). Generally, though, they are inhabited by *S. tenellum*, *S. pulchrum*, and *S. cuspidatum,* which contribute less to peat accumulation than do terrestrial species like *S. fuscum* (Andrus, 1986; Lindsay et al., 2014) as these aquatic species decay more quickly (Johnson and Damman, 1991). More than occasional flooding encourages *S. cuspidatum*, *S. angustifolium*, and *S. fallax* and inhibits *S. fuscum* and *S. capillifolium* (Price et al., 2003).

Shallow flooding can result in expedient colonization of a floating *Sphagnum* raft, especially where weakly decomposed peat remains (Money and Wheeler, 1999; Smolders et al., 2003). Floating raft formation is strongly influenced by depth of inundation due to rewetting and the degree of humification of remnant peat (Tomassen et al., 2004) and these conditions were best met at the 1948 field (Appendix A, Figure A-2; Christen et al., 2016). Upon flooding from ditch blocking, remnant peat either expands up to the new water table or floats and becomes a matrix for *Sphagnum* growth (Smolders et al., 2003). More terrestrial species including *S. magellanicum* and *S. papillosum* as well as bog rosemary and sundews (Money and Wheeler, 1999) and *S. rubellum* can then colonize this raft (Wheeler and Shaw, 1995). These species are later colonizers and contribute more to peat accumulation (Andrus, 1986; Lindsay et al., 2014). Both bulk density of the peat remnant and CH_4 production in the catotelm influence this pathway, with CH_4 contributing to buoyancy (Smolders et al., 2003). Additionally, submerged *Sphagnum* can result in a floating mass if there is sufficient CO_2 in the submerged zone and light penetration through the water (Smolders et al., 2003). The fate of a floating *Sphagnum* mass is not entirely predictable, but it can eventually ground and become the acrotelm post peat harvest (Money and Wheeler, 1999).

Interestingly, *Drosera* spp. at the 1948 field were largely restricted to bare peat as opposed to *Sphagnum* rafts.

Regeneration of a functioning acrotelm is crucial to bog restoration (Smolders et al., 2003). *S. magellanicum, S. papillosum and S. rubellum* are critical to this process, but these are slow to establish hummock species compared to the aquatic *S. cuspidatum* and *S. fallax* (Smolders et al., 2003). *S. cuspidatum* can quickly and completely colonize the water layer (Smolders et al., 2003). *S. fallax* also grows quickly and can tolerate persistent dry conditions (Andrus, 1986). After the establishment of an ideal hydrological regime, the non-aquatic *Sphagnum* spp. can still take decades to colonize (Smolders et al., 2003). Introduction of these species to hollow *Sphagnum* carpets has been successful where colonization has proved too slow (Smolders et al., 2003) and should be considered at the 1948 field. Introducing propagules of these species enables their capacity to engineer their habitat by establishing lawns and hummocks (Smolders et al., 2003).

Propagules (i.e. plant fragments as small as 0.5 cm) can be manually introduced, most commonly conducted with hummock species including *S. rubellum, S. fuscum*, and nurse species *Polytrichum strictum* (Pouliot et al., 2011). Within 20-30 years post-introduction, hummocks comparable to pristine peatlands have been observed (Pouliot et al., 2011). Hummocks and hollows are considered resilient to a changing environment, but likely not impervious to dramatic climate change in the long term (Pouliot et al., 2011). Regardless, a varied surface topography is inherently more resilient to climate change due to species diversity and heterogenous water regime (Taminskas et al., 2016). Intervention may be required to increase species diversity in the 1948 field, for example by transfer of the desired *Sphagnum* spp. (Renou-Wilson et al., 2019).

Improvements to water storage in other areas of Burns Bog should be monitored to ensure a suitable hydrological regime for peat formation. *S. magellanicum*, *S. papillosum,* and *S. rubellum* produce more biomass and are more decay resistant than the aquatic species and thus contribute more to mire breathing (Smolders et al., 2003), acrotelm function, and peat accumulation (Lindsay et al., 2014). Extensive pools are not inherently negative, in that they contribute to stabilization of the rising water table and increase water storage capacity, encouraging site revegetation (Daigle and Gatreau-

Daigle, 2001). Intervention might now be required in the 1948 field to increase diversity, surface heterogeneity, and subsequently peat accumulation. Importantly, these aquatic *Sphagnum* spp. tend to reproduce horizontally, leading to floating carpet formation but contributing little to peat accumulation (Malmer et al., 1994). Also in these cases, a nurse species such as *Eriophorum* spp. can provide a scaffold for non-aquatic and target *Sphagnum* spp. (Smolders et al., 2003). It is possible that the recolonization by hummock *Sphagnum* spp. will occur with time, but in the face of anthropogenic climate change acting on a northern hemisphere raised bog at the southern extent of its climatic zone, intervention may be required to increase water storage before the water table depresses and *Sphagnum* is inhibited. As resources are limited and Burns Bog is very large, pilot studies serve the function of trialing specific approaches to determine those most appropriate in this climate. There seems to be great potential for hummock creation by the introduction of hummock forming species and nurse species at the 1948 field. This trial could then serve as a model for restoration in other areas of Burns Bog with a very shallow water table. As Burns Bog is a "Ramsar Wetland of International Importance," and thus involved in international collaboration on informed conservation (Daigle and Gatreau-Daigle, 2001; Metro Vancouver, 2012), pilot studies here can add to the growing knowledge base of raised bog restoration.

Nurse Species

Nurse species can aid in the recolonization and spread of *Sphagnum*. Living or plastic plants can increase water content of peat, maintain a more humid microclimate, and moderate extreme temperatures, sometimes enabling passive restoration in place of manual rewetting (Money and Wheeler, 1999). *Sphagnum* can be protected from radiation and wind by adjacent shrubs, reducing evapotranspiration and increasing survival and productivity (Potvin et al., 2015). Remnant baulks, especially with overhanging Ericaceae, can also provide this microclimate (Price et al., 2003). Interestingly, young *Sphagnum* cushions isolated from the water table can proliferate as long as shade is provided (Price et al., 2003).

Eriophorum spp. tolerate minor flooding and can serve as a scaffold, enabling growth of non-aquatic *Sphagnum* spp. (Smolders et al., 2003). These microenvironments can also maintain humidity, allowing *Sphagnum* to withstand dry conditions (Smolders et al., 2003). As water table position fluctuates more in harvested sites, this genus would likely have a positive influence here and should be considered for introduction in the 1948 field due to low abundance and common flooding here (Figures 4 and 6). *Eriophorum* spp. tussocks can also play an undesirable role by providing nurse microhabitat for birch (*Betula* spp.) encroachment (Couwenberg et al., 2011) and should be monitored.

Polytrichum is another important genus of nurse plants, especially early in the rewetting process as they are more drought- and disturbance-tolerant than *Sphagnum* (Potvin et al., 2015). This genus was observed in very minor abundance across this study and was completely absent from study plots in both the 1966 and unharvested fields (Figure 4). Gonzalez et al. (2014a) also observed an increase in *Polytrichum strictum* at rewet sites that surpassed the abundance seen at their reference sites (Gonzalez et al., 2014a). *Polytrichum* spp. and *Eriophorum* spp. are both key players in the successional trajectory of rewet bogs (Grosvernier et al., 1995). *Polytrichum* spp. are very productive in open, sunny sites, even in the face of drought stress and thus provide safe sites for *Sphagnum* recolonization (Potvin et al., 2015).

Peat Humification

A *Sphagnum* plant’s core is susceptible to decomposition by acidobacteria, but hyaline cells are less susceptible due to sphagnan and phenolics in these cell pores, which are resistant to mechanical and chemical breakdown (Bu et al., 2019). A von Post value of H3 to H5 equates to partial decomposition of leaf hyaline cells, H6 to H8 equates to indistinguishable leaf structures, and H9 and H10 represent essentially completely decomposed material (Bu et al., 2019). Alternatively, soil classification guides consider H1 to H4 to equate to fibric peat, H5 to H6 to hemic peat, and H7 to H10 to sapric peat (Government of Canada, 2013). Interestingly, when deep peats are extracted and exposed to the same conditions as surface peat, the respiration rate of deep peat is lower (Hogg, 1993). Even when comparing peat at 0.025 m and 0.125 m depths, this trend is prevalent (Hogg, 1993). Therefore, the rate of decay of exposed peat will depend on its depth and exposure to decay prior to harvest.

Anaerobic conditions, decay resistant *Sphagnum*, and low soil nutrient content considerably slow decomposition and enable persistent accumulation of peat over time (Lindsay et al., 2014). The physical and chemical organization of peat changes as it decomposes; microbes first consume the simplest fragments, leaving less bioavailable components behind (Grover and Baldock, 2013). Thus, the rate of peat accumulation increases over time as a larger accumulated peat mass incorporates new peat more quickly than a thin peat layer, meaning older peat is more efficient at peat accumulation (Clymo, 1984). Further, duration spent in the acrotelm is the key indicator in total peat accumulation, while temperature fluctuations, climatic moisture conditions, and vegetation also play a role (Clymo, 1984).

Plant composition and water table position alone cannot assert whether a bog is healthy and functioning, as abundant and productive vegetation is possible without peat accumulation (Clymo, 1984). Up to 90% of a peat mass can be composed of *Sphagnum,* which influence the hydrological regime and engineer conditions for their survival and persistence (Andrus, 1986). Additionally, the *Sphagnum* spp. present influence the degree of humification (Hogg, 1993). A functioning acrotelm is crucial to bog restoration and thus serves as a marker of restoration success (Wheeler and Shaw, 1995). This layer is distinguished from the catotelm not by depth, but by chemistry, physical characteristics, and degree of decomposition (Grover and Baldock, 2013). High hydraulic conductivity and water storage capacity in this layer, due to the abundance of large pores, enables the self-regulating and elastic capacity of the bog surface (Ingram, 1978). Passive *Sphagnum* regeneration varies widely under rewetting conditions and when below 25%, acrotelm establishment is impaired and its self-regulation and peat accumulation functions with it (Gonzalez et al., 2014a). Interestingly, the only field with *Sphagnum* coverage under 25% is the unharvested field (Figure 6).

A healthy acrotelm contains spongy fibric peat that ranges from H1-H4 on the von Post Degree of Humification Scale (Andriesse, 1988; Smolders et al., 2003; Government of Canada, 2013). This should overlay the anaerobic and strongly humified catotelm with comparatively low hydraulic conductivity (Ingram, 1978; Smolders et al., 2003). This upper layer can be highly varied in degree of humification and hydraulic conductivity, while the catotelm should be much more homogenous (Price et al., 2003). Decomposition, though slow in the catotelm, does still occur at a rate of approximately 1% that of the acrotelm (Clymo, 1984) (Appendix A, Table A-2). In an Australian peatland, Grover and Baldock (2013) observed that physical peat properties explain about half of the variation in hydraulic conductivity and peat chemistry likely explains the remainder.

Observations in Burns Bog indicate a trend of increasing degree of humification and decreasing hydraulic conductivity with depth (Howie and Hebda, 2018, unpublished data). Further, edge sites in Burns Bog tend to have shallower peat and a smaller amplitude of mire breathing (Howie and Hebda, 2018). Bu et al. (2019) found degree of humification increased roughly linearly with depth in the top 0.15 m of a Chinese peat bog, but no trend with depths of 0.15 to 0.3 m.

It was expected that degree of humification would increase with depth (Barrett and Matmough, 2015; Bu et al., 2019); however, this general trend was only observed in the 1948 and 1966 fields. Mean decomposition in the 1957 field increased with depth until 0.3 m and then decreased, while mean decomposition in the unharvested field fluctuated with depth (Figures 11 and 12). These observations possibly indicate disturbance and deeper exposure to oxygen in these two fields. Additionally, a large fire more than 50 years ago in the unharvested area may further explain this observation (Howie, pers. comm.).

In this study, surface cracking of bare peat in the dry season was most extensive at the unharvested site, likely due to the deeper and less oscillating water table (Figure 4). Surface cracking in this area of Burns Bog was also noted by Christen et al. (2016). Thickness of the acrotelm is difficult to discern from these findings as wide fluctuations in degree of humification are present at all depths, except at the 1948 field (Figure 12). It appears that the acrotelm is 0.1-0.2 m thick at this site as the degree of humification ranges from H2 to H4 at these depths and then varies more widely below (Andriesse, 1988; Smolders et al., 2003; Government of Canada, 2013). At the other three fields, peat coring revealed a larger range. At the 1957 and unharvested fields, the top 0.1 m varies in humification from H3 to H8, indicating a less than healthy acrotelm. The top 0.1 m of the 1966 field varies only between H3 and H6, indicating restoration here may have been more successful. It is possible that a larger sample size would elucidate a clear demarcation between acrotelm and catotelm. In terms of peat characteristics, the 1948 field appears to be the most restored and even the unharvested field is suitable for restoration. Interestingly, regression analysis determined that *Sphagnum* cover was not largely influenced by degree of humification in the majority of this study (Figure 16).

Howie and Hebda (2018) hypothesize that a 0.1-0.2 m thick new, highly compressible peat layer has formed in some white beak-rush fields, like those in this study. The compressibility of this new peat is important to water storage and elasticity, potentially indicating that this regenerated layer is serving as a functional acrotelm (Howie and Hebda, 2018). The impacts of long-term exposure to drainage can result in decreased peat elasticity and water storage capacity due to oxidation and subsequent peat loss (Howie and Hebda, 2018). Upon rewetting, if peat elasticity was maintained through disturbance, peat volume can change with the water table; however, oxidation damage is irreversible while peat consolidation is somewhat reversible (Price and Schlotzhauer, 1999).

The acrotelm’s self-regulation function seems to protect peat from drought stress as it swells and holds additional water before dry conditions begin (Howie and Hedba, 2018). According to the findings of Howie and Hebda (2018), it is possible, at least at Burns Bog, that a raised water table not only shortens the pathway for water transport, but also increases the water storage capacity of the acrotelm where it has re-established. Wheeler and Shaw (1995) and Money and Wheeler (1999) found elsewhere that when the acrotelm was not completely removed, this layer likely maintained its self-regulation and water storage capacity, thus maintaining contact between the water table and *Sphagnum* capitula. This highlights the importance of a regenerating acrotelm for raised bog restoration. Importantly, though, only 29% of the acrotelm was preserved in all of Burns Bog during harvest activities (Hebda et al., 2000). Where removed, the underlying but now exposed catotelm has higher bulk density and reduced water storage compared to the acrotelm, and is thus incapable of self-regulation function (Money and Wheeler, 1999).

Strong humification can result from the reduced *Sphagnum* cover due to harvest activities (Barrett and Watmough, 2015) and is very common in harvested bog remnants (Smolders et al., 2003). This is because over time, the extended *Sphagnum* deficiency can establish a positive feedback loop in which wide water table fluctuations enable further decomposition, and further peat losses in the absence of peat inputs, exacerbated by high water tension (Smolders et al., 2003; Barrett and Watmough, 2015). The most *Sphagnum* covered peatlands have the least humified peat, likely due at least in part to the resistance to decay that characterizes this genus (Barrett and Watmough, 2015). This corroborates the observation that lag time between harvest and rewetting might be a crucial factor in restoration success (Price, 1997; Howie and Hebda, 2018).

Bog vegetation is only marginally productive, as nutrients are provided exclusively by precipitation (Lindsay et al., 2014). Annual peat accumulation is usually 0.5-1 mm (Lindsay et al., 2014), though faster rates have been observed at Burns Bog, 1.2 mm/year average in four wet sites and 6.3 mm/year average in six hummock sites (Biggs, 1976). There is considerable variation in reported accumulation rates in peatlands, potentially due to a lack of uniformity in the calculation process (i.e. a failure to differentiate litter accumulation in the acrotelm from peat input to the catotelm) (Taminskas et al., 2016). Therefore, peat accumulation rates should be interpreted with caution. Research by Taminskas et al. (2016) supports the growing notion that peatland morphology is more complicated than the diplotelmic model or a single depth threshold (e.g. Morris et al., 2011). Instead, these authors found that microtopographic variation also entailed separate hydrological regimes constituting subbasins as opposed to homogenous moisture conditions. Therefore, surface variation has strong implications for hydrological conditions in specific areas of a peatland (Taminskas et al., 2016).

Microtopography

RDA indicated that the three species most influenced by environmental variables were white beak-rush, *Eriophorum* spp., and bog blueberry (Figure 13). Bog blueberry was most common on hummocks, white beak-rush most common on hummock-hollow combinations and *Eriophorum* spp. most common with highly decomposed peat at 0.5 m depth. This highlights that the 1948 field had the highest proportion of hollows and also white beak-rush, a finding corroborated by Moore et al. (2002) who found that sedge productivity was positively correlated with water table positions above 0.3 m. Also, Malmer et al. (1994) noted that this species tends to predominate flat areas, which are most abundant at the 1948 field (Table 3). It is interesting that the distribution of *Eriophorum* spp. was positively correlated with well decomposed peat at the 0.5 m depth (Figure 13). Multiple regression, with field, topography type, and degree of humification as predictors, corroborated this finding and also elucidated that the distribution of this genus followed the same trend at all depths (Table 4). This genus is also associated with flat areas (Hogg, 1993) but, surprisingly, was considerably more prevalent at the 1957 and 1966 fields (Figure 6). It is unsurprising that bog blueberry was commonly found on hummocks as its vascular system enables efficient water uptake, it thrives after drainage, and hummocks were most prevalent on the unharvested site (Lindsay et al. 2014).

Linear regression illustrated that *Sphagnum* distribution was not strongly influenced by topography type (Figures 14 and 15). When constrained by field, R^2 values were larger, indicating that site characteristics are influencing this distribution, as expected. Across all fields, *Sphagnum* percent cover was greater off hummocks (Figure 15). When constrained by site there was little variation between fields and only at the 1966 field was there a statistically significant correlation ($R^2 = 0.5$, $p = 0.0095$) (Figure 14). Here S*phagnum* coverage was highest on hummocks.

Drainage increases bulk density and further decreases the capacity for water transport (i.e. hydraulic conductivity) (Price, 2003) and can result in peat subsidence, altering microtopography (Price and Schlotzhauer, 1999; Lindsay et al., 2014). Theoretically, the process to restore disturbed raised bogs is straightforward; firstly, adequate precipitation and water storage is needed, followed by a source population of bog vegetation propagules (Money and Wheeler, 1999). Importantly, though, topography alterations may be required to facilitate hydrological and vegetative regeneration to the historical, self-regulating capacity (Money and Wheeler, 1999). Natural bogs encompass a mosaic of microhabitats due to the assortment of growth habits of *Sphagnum* spp., resulting in an undulating landscape (Lindsay et al., 2014). These microhabitats create niche environments for vegetation, birds, and invertebrates and are long-lasting due to stability of the water table (Wallen et al., 1992; Lindsay et al., 2014). Additionally, productivity tends to be higher for hollow species compared to hummock species (Andrus, 1986) for example because there are generally more vascular plants on hummocks than hollows (Malmer et al., 1994).

Hummocks were most prevalent at the 1957 and unharvested fields, and these were also the sites with the greatest depth to water table (Table 3; Figure 4). The 1948 field had the most depressions and most shallow water table. The homogenous topography at this field coupled with the prevalence of aquatic *Sphagnum* here may indicate stagnated restoration (Bonsel and Sonneck, 2011). Even though regression analysis did not indicate a strong correlation between topographic type and *Sphagnum* cover in this study*,* this relationship has been seen in other studies (Price et al., 2003) and recontouring to increase topographic variation can be performed in conjunction with increasing water retention (Wheeler and Shaw, 1995; Money and Wheeler, 1999; Smolders et al., 2003). However, as recontouring is resource-intensive and can introduce invasive species, I first recommend identifying *Sphagnum* spp. in the 1948 field to determine if terrestrial *Sphagnum* is present. If certain species are absent, I suggest conducting a pilot study in this field with the introduction of hummock-forming *Sphagnum* spp. (e.g. *S. rubellum* and *S. fuscum*) as water table here is very shallow and these species are capable of engineering hummocks in these conditions (Smolders et al., 2003; Pouliot et al., 2011). Nurse species, including *Polytrichum strictum* should also be further introduced for their role in creating microclimates suitable for young *Sphagnum* propagules (Pouliot et al., 2011).

The distribution of hummocks and hollows is based on the varying decomposition rates of different S*phagnum* spp., the propensity of productive vascular species to colonize raised ground to avoid anaerobic root environments (Wallen et al., 1992), and the vertical growth pattern of hummock *Sphagnum* spp. (van Breemen, 1995). Up to 90% of hummock productivity can be attributed to these vascular species, which determine surface layer conditions through shading, providing structural support and transporting deeper nutrients to the surface (Wallen et al., 1992). As microtopography determines water availability, it also constrains vegetation distribution and productivity, nutrient availability, and rates of peat decomposition and accumulation (Briggs, 1976; Potvin et al., 2015; Shi et al., 2015). For example, Biggs (1976) found peat accumulated in Burns Bog at a faster rate in hummock sites versus wet depressions. Elevated hummocks experience more runoff, resulting in drier microsites, while hollows and pools are concave and hold stagnant water, resulting in anaerobic conditions (Lindsay et al., 2014; Christen et al., 2016). Price et al. (2000) observed that drained and undrained sites differed most in terms of microtopography, resulting in dissimilarity between revegetation trajectories. Unharvested baulks retained in manual peat harvest activities serve as source populations for recolonization because, upon establishment, *Sphagnum* can engineer its local environment for colonization (Price et al., 2003). Therefore, an increase in hummocks in the 1948 field would likely increase peat accumulation here, contributing to acrotelm re-establishment. Further, with their increased efficiency in water transport, an increase in hummocks on harvested sites would confer greater resilience to anthropogenic climate change and increased duration of summer drought.

Potvin et al., (2015) noted that Ericaceae rely on mycorrhizal associations to obtain nutrients due to shallow roots. Intolerance of flooding conditions results in their distribution on lawns and hummocks, resulting in expansion in long-term drought. In the case of considerable water table depression, Ericaceae will likely be too shallowly rooted to obtain water. Instead, aerenchymous sedges will be favoured due to deep rooting and the capacity of their roots to transport oxygen. This likely contributes to the low abundance of Ericaceae at the 1948 field (Figure 7).

Rewet sites had similar microinvertebrate assemblages compared to pristine reference sites but considerably reduced species diversity and abundance of macroinvertebrates, and time since rewetting did not strongly increase this number (van Duinen et al., 2003b). A lack of habitat diversity may be responsible as macroinvertebrates require a number of habitat types to complete their lifecycle (van Duinen et al., 2003b). Restoration activities should safeguard and enhance habitat diversity to aid in recolonization by and protection of the macrofauna characteristic of raised bogs (van Duinen et al., 2003a).

Climate Change

As Burns Bog is the Fraser Lowland’s largest bog, it offers important habitat for waterfowl and other wildlife (Hebda et al., 2000). It marks the southernmost extent of Canadian *Sphagnum*-predominated raised bogs and is one of western North America’s most southern raised bogs (Hebda et al., 2000). Precipitation here, measured at the Environment and Climate Change Burns Bog weather station, demonstrates extension of the dry season in recent years and an increase in annual maximum precipitation in a single day in the fall and winter seasons (Figure 5).

Due to a forecasted 20% decrease in summer rain (i.e. from 21 consecutive rain free days currently to 26 projected by the 2050s and 29 by the 2080s) (Metro Vancouver, 2016) and the location of Burns Bog at the southern extent of its climatic tolerance (Howie and Hebda, 2018), this raised ombrotrophic bog is highly threatened by climate change. The projected extension of the growing season into September could cause *Sphagnum* mortality if water retention and storage is not increased (Verry, 1988; Metro Vancouver, 2016). The response in Burns Bog to the installation of drainage ditches provides insight to predict the bog’s response to climate change. The major difference, however, is the rate of change as climate change is progressing more slowly than the local response to a new drainage ditch (Frolking et al., 2011).

The forecasted increase in fall precipitation (Metro Vancouver, 2016) suggests that improved water retention may afford resiliency to summer drought (e.g. Price et al., 2003), though heavy rain events may result instead in considerable runoff (Frolking et al., 2011). Resilience of a peatland is rooted in its capacity to endure disturbance and drought (Ingram, 1982; Clymo, 1984). Drying of peat can result in irreversible alteration to structure and elasticity (Ingram, 1982) and this loss of function has been observed in both harvested and unharvested fields of Burns Bog (Howie and Hebda, 2018). This suggests that re-establishing a functional acrotelm via hydrological management and restoration of a suitable vegetation community is crucial to climate change resiliency. Frolking et al. (2011) suggest that annual runoff might better indicate water availability than precipitation. Moisture availability, specifically the combination of both precipitation inputs and outputs (i.e. runoff), determines the volume of stored water, with average and seasonal changes providing the best indicator (Frolking et al., 2011). Projected increases in precipitation during the wet season, especially increasing rainfall during extreme precipitation events, (Metro Vancouver, 2016) suggest that runoff will become a larger concern and should be monitored (Figure 5).

Water retention can be improved, for example, by creating additional microtopographic mosaics across the bog (Price et al., 2003). Constructing bunds along contour lines to create open water basins that enable passable lateral water transport, the addition of nurse plants to create shade and moist microhabitats, or even adding artificial plants to generate shade can be used to increase fall and winter precipitation retention (Price et al., 2003; Lindsay et al., 2014) (See Recommendations for Restoration and Future Research).

If water table position is maintained above 0.4 m below the surface (Verry, 1988), *Sphagnum* can avoid mortality due to desiccation, thus accumulating and storing carbon in the long term (Schipperges and Rydin, 1998). Also through water table management, CH_4 emissions can be managed, maintaining the role of peatlands as carbon stores rather than sources (Evans et al., 2014). Through vegetation monitoring and management and minimizing nutrient input (especially nitrogen), encroachment by vascular plants can be minimized and carbon sink function maintained (Evans et al., 2014). Under a warming climate, CH_4 emissions from peat will increase, especially in the top 0.2-0.3 m (Wilson et al., 2016).

Robroek et al. (2007a) conducted manipulative experiments exposing four species of *Sphagnum* to varying water table positions and temperatures to elucidate responses to anthropogenic climate change, supporting their hypothesis that *Sphagnum* spp. composition will likely transform in a changing climate. With changing species composition, ecosystem function, too, will be altered due to climate change (Robroek et al., 2007a). A depressed water table position saw decreases in growth and productivity of lawn species *S. magellanicum* and a reduced capitula water content (Robroek et al., 2007a). Therefore, this species is more vulnerable to climate change and could see a decrease in coverage and productivity in the future if further hydrological management is not successful in maintaining a raised water table. The decrease in capitula water content was less pronounced in the hummock species, pointing to one way in which these hummock specialists are less threatened by climate change (Robroek et al., 2007a).

Sphagnum spp. lack vascular tissue, a waterproof cuticle, and rely on external transport of water via capillarity (Lindsay et al., 2014). They also differ by species in their resilience to drought stress (Potvin et al., 2015) and rate of drying (Schipperges and Rydin, 1998). *Sphagnum* capitula drying passed 10% moisture content must be avoided to maintain photosynthetic capacity (Schipperges and Rydin, 1998). Extension of the dry season (Metro Vancouver, 2016) will thus limit productivity and affect species distribution. For example, the 1948 field currently appears more restored as it has the greatest *Sphagnum* coverage, shallowest water table, and a potential acrotelm layer 0.1-0.3 m thick; however, this field might be least resilient to extended drought due to a potential overabundance of aquatic *Sphagnum* spp. as this group has inferior water transport compared to hummock species (e.g. Robroek et al., 2007b). Aquatic *Sphagnum* spp. are more drought-tolerant and more decomposition-resistant, implying a reduced rate of peat accumulation over time in this area of Burns Bog in the absence of colonization by non-aquatic *Sphagnum* spp. (Lindsay et al., 2014). This highlights the importance of identifying *Sphagnum* to species here, investigating mire-breathing activity, and introducing non-aquatic *Sphagnum* propagules and nurse species that serve as a scaffold for these late colonizers (Andrus, 1986; Smolders et al., 2003). This diversity is crucial to a functioning acrotelm, a key factor in resilience to anthropogenic climate change (Lindsay et al., 2014; Howie and Hebda, 2018). Additionally, a topographically varied bog surface is more resilient as this implies more diverse *Sphagnum* functional groups are present and thus greater adaptive capacity (Lindsay et al., 2014).

Study Limitations

It is difficult to tease apart the influence of these different variables due to their highly interrelated nature. For example, RDA demonstrates that increased humification is negatively correlated with *Sphagnum* coverage (Figure 13). However, *Sphagnum* coverage is greatest at the 1948 field and here the water table is very shallow (Figure 4). A greater proportion of the peat mass is thus submerged, and decomposition subsequently inhibited (Figure 12). As identification of *Sphangum* to species is very difficult and requires training and resources that were not available, it was not possible in this study. This information would vastly benefit the assessment of restoration status.

Further, it is impossible to quantify the influence of the various ditches, but their proximity to the study transects provides some idea of varying drainage impacts in this study. For example, from figure 2 it appears that the 1966 field may be most impacted by drainage ditches, but their depth and time since blocking is not available.

Study design was limited by the location of piezometers for which long-term depth to water table measurements were available and figure 2 demonstrates the potential benefit afforded by eliminating this restriction, for example by measuring soil moisture instead. Additionally, because depth to water table was manually measured on a monthly basis, fine-scale temporal variability is lost. Price (2003) found temporal heterogeneity in peat moisture measurements at multiple depths, so important information may be lacking.

Reference Site Limitations

A comprehensive inventory and review of Burns Bog published in 2000 determined that the unharvested area surrounding and including the reference area of this study is crucial to bog survival (Hebda et al., 2000). As a result, Metro Vancouver have monitored this area for approximately 14 years, including annual vegetation surveys (Thomas Munson, Metro Vancouver contract field technician, pers. comm.).

Interestingly, the unharvested reference site was considered undisturbed and unaffected by drainage or fire (Howie and Hebda, 2018; Howie, pers. comm.). This study, however, demonstrates that water table position has decreased to concerning levels in recent years (Figure 4), drought-tolerant species thrive (Figure 6), and surface peat humification is present (Figure 11). Additionally, recently received information indicates the potential occurrence of a large fire here over 50 years ago (further discussed below) and the presence of a small drainage ditch very near the unharvested transect (Figure 2).

In their research in Burns Bog, Christen et al. (2016) described the unharvested area as “relatively undisturbed” as it was not cleared of vegetation. They characterized these areas by stunted pines and almost total ground cover by *Sphagnum* spp., Ericaceae spp., and *Cladina* spp.. Though they did not discuss observations of bracken fern or a lack of sundew, they did note potential disturbance due to previous drainage.

The presence of bracken fern in the reference area indicates that conditions are dry, and disturbance likely (Money and Wheeler, 1999; Hebda et al., 2000 from Madrone consultants Ltd., 2000). Dry conditions can facilitate encroachment by birch, bracken fern, and *Eriophorum* spp. and can facilitate a trajectory towards wet heath (Money and Wheeler, 1999). Further, the lack of sundew in the reference plots indicates dry conditions (Renou-Wilson et al., 2019) and, potentially, greater nutrient availability (Daigle and Gatreau-Daigle, 2001) due to close proximity to the landfill (Figure 1). Interestingly, despite a similarly depressed water table at the 1957 field, sundew cover here was similar to that at the 1948 and 1966 fields (Figure 6). Dense Ericaceae, abundant non-Sphagnum mosses (Lindsay et al., 2014), and hollows predominated by lichens also suggest insufficient moisture conditions (Wallen et al., 1992).

Additionally, a large fire more than 50 years ago may explain some of these findings, including the presence of humified surface peat and abundant lichens (Howie, pers. comm.). Upon a large fire event with loss of *Sphagnum*, 50 or more years are needed before *Sphagnum* returns (Lindsay et al., 2014). Restoration of function, therefore, takes additional decades (Lindsay et al., 2014). Even 80 years post fire, vegetation and microtopography are likely still altered and acrotelm impaired, which in turn affects the catotelm and carbon sequestration capacity (Lindsay et al., 2014).

A bog centre tends to have a shallower water table position, with distance from surface to water table reduced further by peat removal and subsequent ditch blocking (Howie et al., 2009). Harvest activities in Burns Bog were extensive and restricted to the non-edge portions of the bog, leaving no unharvested reference sites in the same area. Regardless, if unharvested strips remained, they would still be impacted by drainage. Therefore, reference conditions were difficult to ascertain. This reference area was also chosen as the vegetation community at harvested sites (i.e. White Beak-rush – *Sphagnum*) is expected to transition through succession towards that seen at the unharvested reference field (i.e. Pine-*Sphagnum*-Low Shrub) (Howie, 2004). It therefore appears that the unharvested reference site of this study is disturbed and thus has limited function as a model for reference conditions, though it does provide insight into unharvested conditions at Burns Bog. I recommend restoration activities also be focused in these areas to prevent further degradation and loss of function.

Recommendations for Restoration and Future Research

Bog restoration is site-specific and thus standardized best management practices do not exist (Price et al., 2003). Burns Bog presents unique challenges and opportunities for hydrological and vegetative restoration that can be performed concurrently. By monitoring the impacts of disturbance and restoration, timely intervention is possible to establish a fully functioning raised ombrotrophic bog (Smolders et al., 2003). This suggests the importance of resuming depth to water table monitoring in the harvested fields.

Restoring drained peatlands requires time and resources. Maximizing cost efficiency and minimizing damage are both important in monitoring. For example, multisensory unmanned aerial vehicles (MUAV) can measure plant height and, with five spectral indices, can create 3D models of peatland surface on a regional scale (Beyer et al., 2019). This technology therefore has the potential to reduce the footprint of monitoring in these sensitive ecosystems. The use of multiple sensor types increases the amount of differentiation possible between similarly coloured species (Beyer et al., 2019), and can be used to establish an inventory of *Sphagnum* spp. that would aid in predicting the vegetation response to climate change.

Ditch blocking is relatively cost effective and minimally invasive (Gonzalez et al., 2014a), but further interventions in targeted areas of Burns Bog can serve as pilot studies to better understand the site-specific response to restoration. More invasive interventions including plant propagule introductions and surface restructuring to increase water retention (e.g. shallow basins, berms, and terracing) can increase the rate of revegetation and result in greater species diversity (e.g. more than just aquatic *Sphagnum* spp.). These interventions are much more costly though and should be targeted because rewetting alone can regain function in many cases (Gonzalez et al., 2014a).

Further water table management could benefit the 1957 and 1966 fields, but is also recommended in the unharvested field. Both the unharvested and 1957 fields are approaching the critical depth to water table threshold for bog vegetation growth of 0.4 m and should be prioritized to prevent further loss of mire breathing function in the acrotelm (Verry, 1988). Water retention, too, should be improved to withstand the increasing duration of summer drought (Metro Vancouver, 2016). Surface recontouring, additions of peat embankments (Money and Wheeler, 1999), creation of water reservoirs in targeted areas (Smolders et al., 2003), and bunding patchy regions (Wheeler and Shaw, 1995) can be useful in increasing the storage of fall precipitation. These targeted options allow prioritization in specific areas, pilot studies, and iterative research to determine the most effective site-specific methods, as increasing saturation over a landscape as large as Burns Bog is difficult (Wheeler and Shaw, 1995; Smolders et al., 2003). For example, success of bunding is contingent at least on the presence of highly decomposed and impermeable peat beneath (Money and Wheeler, 1999).

As study design was limited by the location of piezometers for which long-term depth to water table measurements were available, soil moisture at each plot should also be measured. Depth to water table measurements, where available, should be compared to soil moisture to elucidate trends in pore water drainage and exposure to oxidation (Price, 2003). Additionally, probes enabling continuous measurements, as opposed to the monthly manual depth to water table measurements by the City of Delta, would reveal fine-scale temporal variability. Price (2003) found soil moisture to be temporally variable at multiple peat depths, suggesting the utility of a pilot study within Burns Bog to investigate this parameter. If wide fluctuations occur over time, continuous measurements might be warranted to corroborate monthly depth to water table measurements and reveal patterns in water movement. If temporal variability is not extensive, future investigation should measure soil moisture at each plot to integrate this parameter into multivariate analysis. Further, by reducing reliance on available piezometers, study design could better represent the spatial extent of harvested fields. Figure 2 demonstrates the need to run transects on field edges to ensure proximity to piezometers.

Raised bog restoration research should also look at patterns in mire breathing to establish baseline trends in surface oscillation and better understand local elasticity and resilience in the face of climate change (Potvin et al., 2015; Howie and Hebda, 2018). Measuring bog surface changes in conjunction with depth to water table elucidates temporal fluxes in water storage capacity and transport, and also how bog function is influenced in the long term (Price and Schlotzhauer, 1999; Howie and Hebda, 2018). For example, extended drought causes pore spaces in peat to collapse, increasing bulk density and decreasing peat elasticity (Price and Schlotzhauer, 1999; Potvin et al., 2015). This lowered hydraulic conductivity inhibits water transport and restricts capillary flow in plant roots (Price and Schlotzhauer, 1999). Interestingly, though, increased bulk density can aid in retention of precipitation in heavy rain events (Price, 2003).

Investigations by Christen et al. (2014) on greenhouse gas emissions in rewetted fields of Burns Bog noted that in fields harvested via the Atkins-Durbrow Hydropeat method, baulks (i.e. elevated, unharvested strips) were left unharvested between fields and can serve as an “undisturbed” reference. Although these microsites are not truly undisturbed, due to their proximity to peat extraction, trampling, and drainage, they can serve as an additional determinant of reference conditions as the unharvested areas of Burns Bog were deemed disturbed. For example, paired analysis could be used to compare vegetation at harvested plots to nearby remnant baulks of similar microtopography. Interestingly, these remnant pieces were less common at the 1948 field.

Since the 1948 field is a major emitter of CH_4 and potentially colonized largely by aquatic *Sphagnum*, intervention here should involve introduction of nurse species including desirable *Eriorophorum* spp. and *Polytrichum* spp.. The microclimatic conditions created by these species are suitable for *Sphagnum* colonization and vertical growth (Smolders et al., 2003). Upon establishment of nurse species, the introduction of terrestrial *Sphagnum*, including *S. magellanicum, S. papillosum* and *S. rubellum*, would likely contribute to hummock formation and subsequent increases in peat accumulation (Andrus, 1986; Smolders et al., 2003). These species have been identified on site (Hebda et al., 2000) and can be collected from the donor sites identified in Howie (2004) every 2-5 years without harming these areas (Daigle and Gatreau-Daigle, 2003). A diverse *Sphagnum* community is required for acrotelm functionality, increasing water storage in the wet season to withstand more consecutive dry days (Metro Vancouver, 2016). It appears that a 0.1-0.2 or 0.3-m thick acrotelm has established at the 1948 site (Figure 12); however, if aquatic *Sphagnum* spp. predominate, peat accumulation will be minor (Andrus, 1986). Especially as productivity and thus peat contribution is greater on hummocks, of which there are very few (Table 3). Relatively stable flooding encourages aquatic species and the lack of hummocks may indicate that terrestrial sphagnum has struggled to reach this area (Smolders et al., 2003). A topographically heterogeneous and undulating bog surface is typical of natural bogs due to the diverse *Sphagnum* community and should be a goal of restoration (Money and Wheeler, 1999; Lindsay et al., 2014). If future research endeavours to determine maximum peat depth, systematic rather than random design should be considered to adequately measure depth under hummocks, hollows, and pools as Biggs (1976) found peat accumulated in Burns Bog at a faster rate in hummock sites versus wet depressions.

10 to 30 years are typically required to re-establish the desired hummock-hollow-microtopography of natural bogs (Pouliot et al., 2011), noting that after 10 years, *S. angustifolium, S. magellanicum, S. fuscum,* and *S. rubellum* had not considerably recolonized (Gonzalez et al., 2014a). I recommend *Sphagnum* in the 1948 field be identified to species and if it is determined that aquatic species including *S. cuspidatum* and *S. fallax* predominate, a pilot study be initiated in the 1948 field as it has the shallowest water table and greatest cover of *Sphagnum.* These hummock species can then engineer hummocks and create a varying bog surface.

Rewetting can result in extensive pool formation (Beadle et al., 2015) and this is seen at the 1948 field. These pools can provide habitat and food for both terrestrial and aquatic species, but the difference between these artificial pools and pristine peatlands is understudied (Beadle et al., 2015). The inclusion of peatland restoration generally, and rewetting specifically, in both Ramsar and Kyoto agreements stresses the importance and international significance of this work (Beadle et al., 2015). For example, in pockets of non-target species, pools of varying size can be dug, increasing water retention and habitat heterogeneity (Beadle et al., 2015). Pilot pools dug in the 1957 and 1966 pools, along with the resumption of water table monitoring, can be conducted to investigate this restoration option within Burns Bog. In the case of bog pools, bigger is not necessarily better so only small pools should be trialed (Beadle et al., 2015). Importantly, though, multiple years are likely required before avian and amphibian wildlife species richness would approach natural conditions (Mazerolle et al., 2005).

Future studies should identify *Sphagnum* to species to understand trends in revegetation. *Sphagnum* spp. differ in their moisture requirements (Robroek et al., 2007b), desiccation resistance (specifically, their capacity to recover after desiccation) (Andrus, 1986) and photosynthetic response to this drying (Schipperges and Rydin, 1998). Also, a tight growth habit is more drought resistant (Schipperges and Rydin, 1998). Therefore, mapping of *Sphagnum* to species coupled with local projections of increased consecutive rain free days could be used to predict resiliency to drying and enable prioritization of sites for water retention improvements. For example, GIS mapping is promising for relating topographical data with hydrology and ecology to prioritize areas for restoration (Mackin et al., 2017b). Vegetation can also serve as a proxy for hydrological conditions and thus mapping can be used to guide management decisions (Mackin et al., 2017b).

Potvin et al. (2015) highlight the need for future research to delineate the interaction between water table position and different vegetation types, especially sedges and Ericaceae. Through their manipulation experiments, they were able to tease apart some of the interactions between plant community functions and water table position. These manipulation experiments should be conducted in other regions and with different vegetation types to understand how changes to the water table position or vegetation community present will influence peatland functionality and resilience in a changing climate.

Additional indicators of raised bog health may be appropriate in Burns Bog. For example, research in Quebec found ants to be less abundant in damaged bogs and a suitable indicator of bog type (Daigle and Gatreau-Daigle, 2001). Additionally, spiders and ground beetles were prevalent on harvested sites due to the drier environment (Daigle and Gatreau-Daigle, 2001). This or other arthropods may prove a suitable indicator of the restoration stage of a given area within Burns Bog (Daigle and Gatreau-Daigle, 2001). Amphibians such as green frogs (*Rana clamitans*) can be more abundant in natural sites and can also serve as indicators of restoration success (Daigle and Gatreau-Daigle, 2001).

This study provides insight into restoration success in Burns Bog fields harvested for peat using the Atkins-Durbrow Hydropeat method. It also elucidates the disturbance present in an area previously considered undisturbed. The variety of responses to ditch blocking efforts can be seen in the different water table position, vegetation composition, and peat humification in these fields. Each field, including the intended reference site, face different issues and are variously threatened by climate change. Increased interest in peatland restoration, evidenced for example by the purchase of Burns Bog in 2004 and its designation as a Ramsar site in 2012, indicate that these areas provide ecosystem services and inherent value worthy of protection with evidence-based restoration and scientific collaboration.

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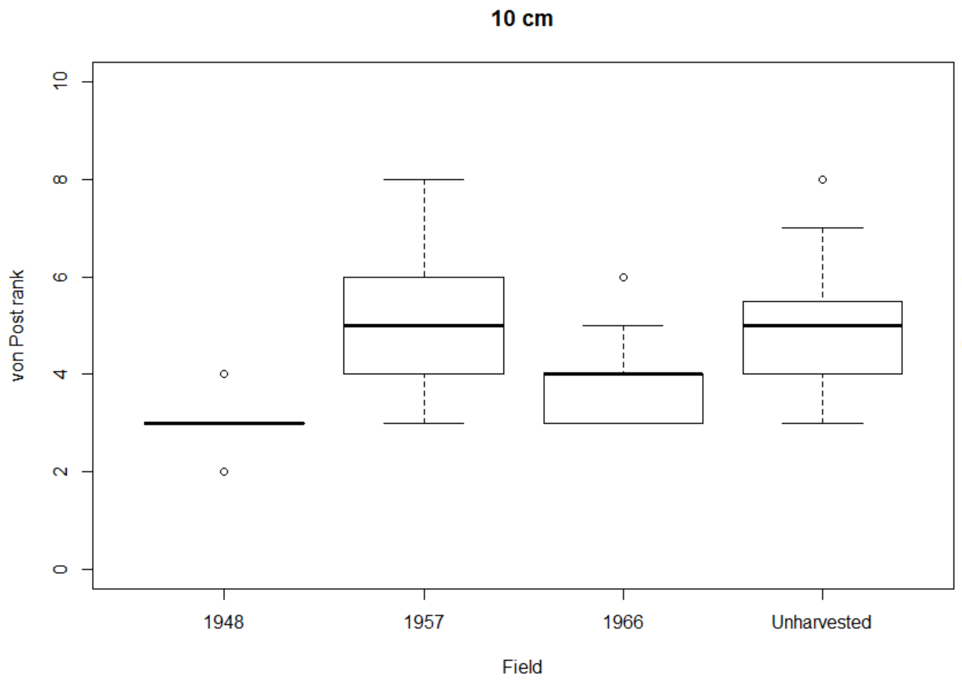
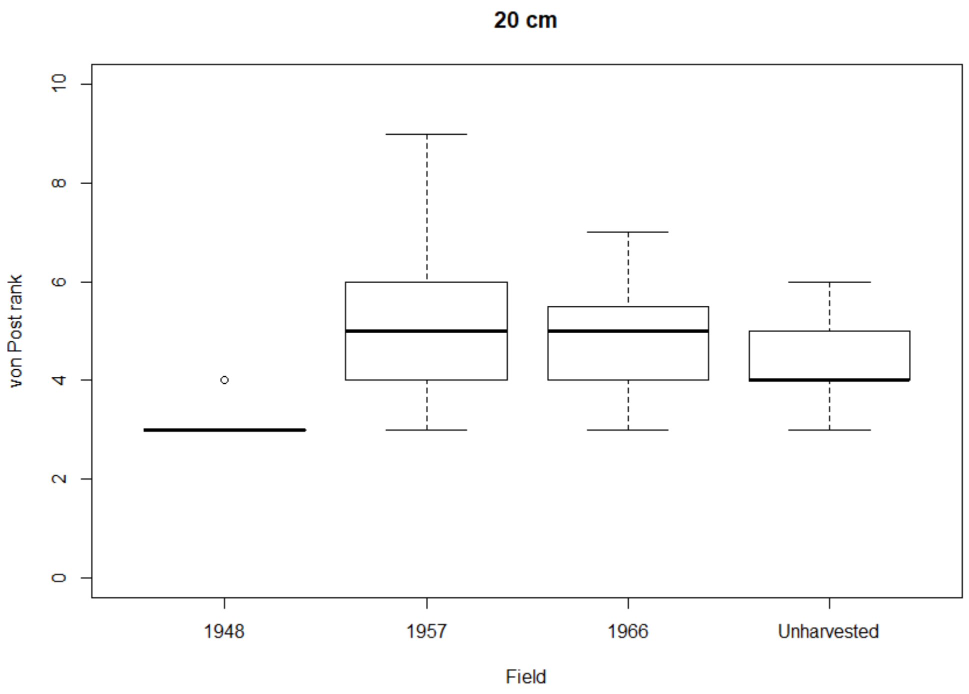
Appendix.

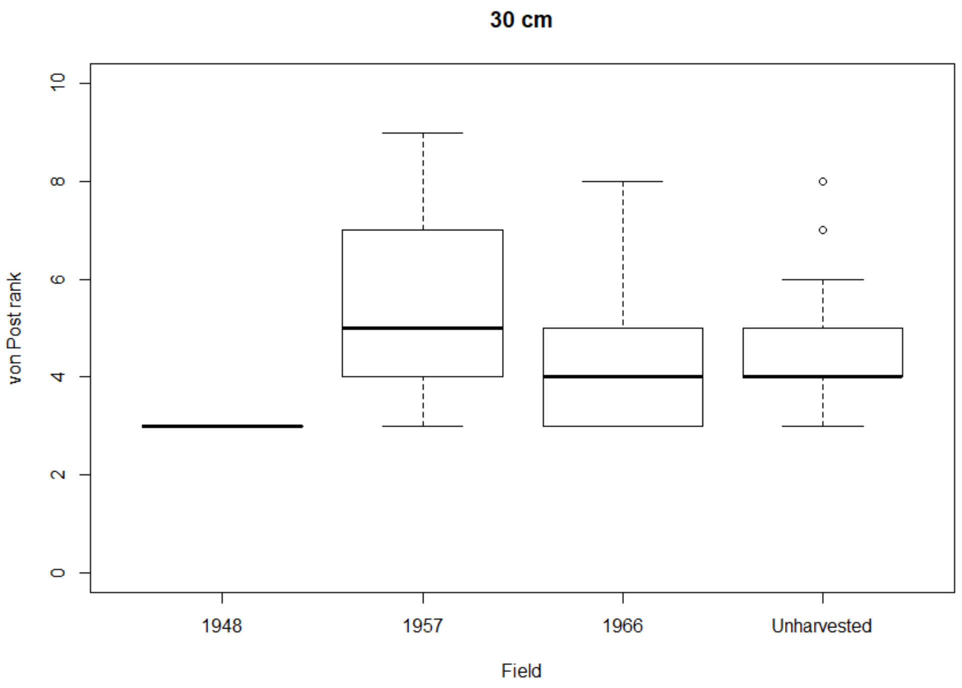
Supplemental Tables and Charts

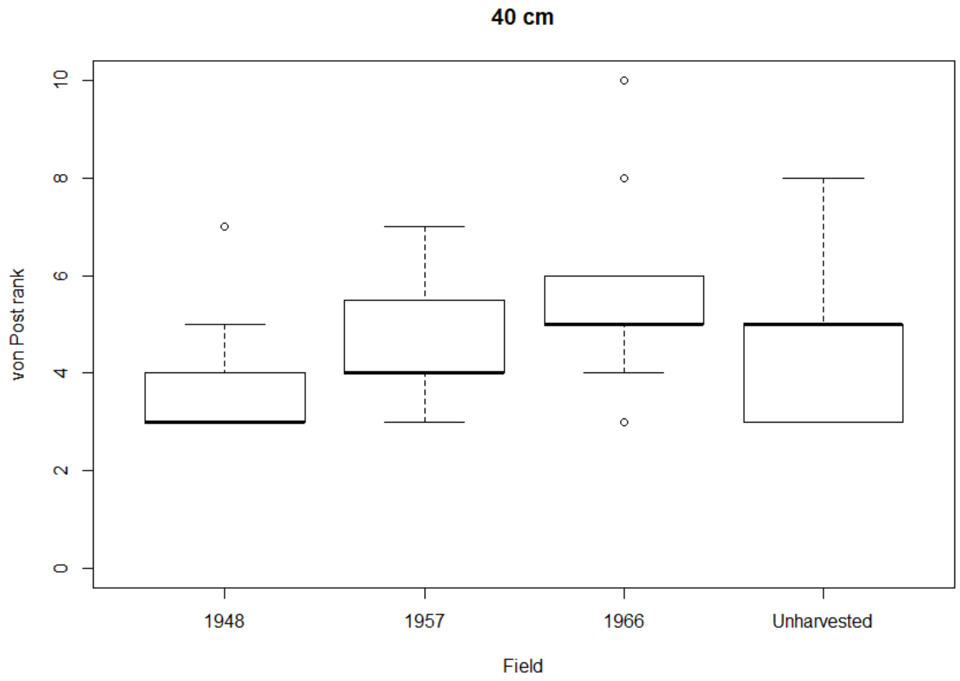
Table A-1. Range in von Post Degree of Humification Scale values in Burns Bog during June and July, 2018 (n = 20 per field and depth).

	1948	1957	1966	Unharvested
0.1 m	H2 – H4	H3 – H8	H3 - H6	H3 – H8
0.2 m	H3 – H4	H3 – H9	H3 - H7	H3 – H6
0.3 m	H3 – H6	H3 – H9	H4 - H8	H3 – H8
0.4 m	H3 – H7	H3 – H7	H3 - H10	H3 – H8
0.5 m	H3 – H8	H3 – H9	H4 - H10	H3 – H6




20 cm
von Post rank
0
2
4
6
8
10
1948
1957
1966
Unharvested
Field


30 cm
von Post rank
0
2
4
6
8
10
1948
1957
1966
Unharvested
Field


40 cm

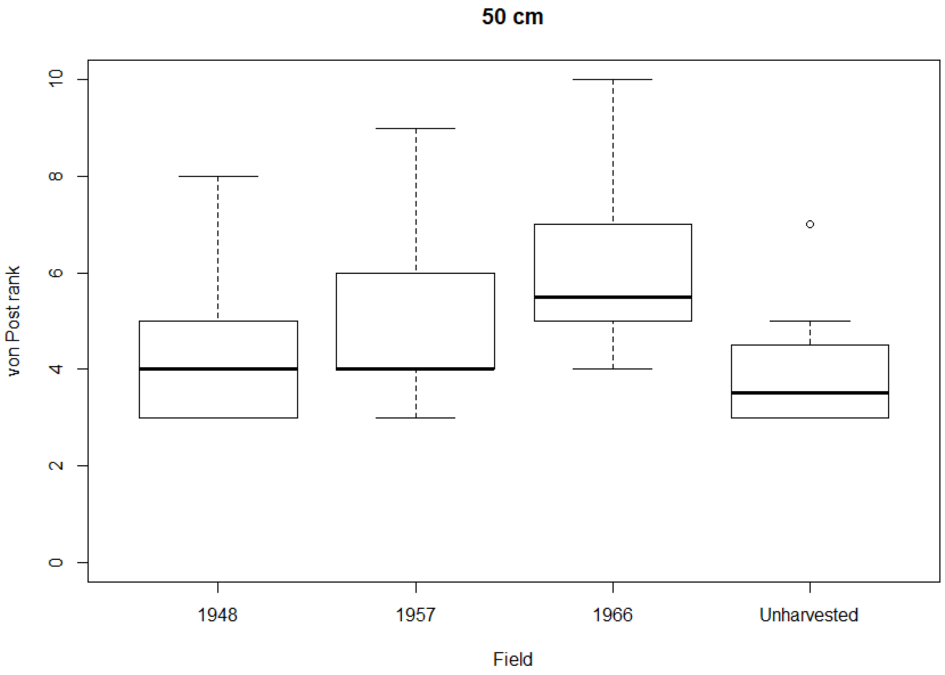

50 cm

Figure A-1. **Boxplot demonstrating degree of peat humification at specified depths of three harvested and one unharvested field using the von Post Degree of Humification field test in Burns Bog in June and July, 2018 (n = 20 per field).**

Table A-2. **Results of investigative peat coring in Burns Bog in June and July, 2018 at plot 1, 10, and 20 presenting degree of humification with summary statistics. Blank values indicate clay layer was reached.**

	0.6m	**0.7m**	**0.8m**	**0.9m**	**1m**	**2m**	**3m**	**4m**	**5m**
1948-1	2	1	2	5	10	3	3	10	
1948-10	3	3	3	3	3	4	8		
1948-20	4	5	6	8	6	3	7		
sum	9	9	11	16	19	10	18		
mean	3	3	3.666667	5.333333	6.333333	3.333333	6		
standard deviation	1	2	2.081666	2.516611	3.511885	0.57735	2.645751		
standard error	0.57735	1.154701	1.20185	1.452966	2.027588	0.333333	1.527525		
1957-1	3	2	2	6	8	2			
195710	1	1	2	3	5	10	6		
1957-20	4	4	8	7	3	3	5	2	8
sum	8	7	12	16	16	15			
mean	2.666667	2.333333	4	5.333333	5.333333	5			
standard deviation	1.527525	1.527525	3.464102	2.081666	2.516611	4.358899	0.707107		
standard error	0.881917	0.881917	2	1.20185	1.452966	2.516611	0.5		
1966-1	6	6	6	8	8	9	8	4	
1966-10	5	5	5	4	6	9	9	5	5
1966-20	3	4	2	2	3	7	5		
sum	14	15	13	14	17	25	22		
mean	4.666667	5	4.333333	4.666667	5.666667	8.333333	7.333333		
standard deviation	1.527525	1	2.081666	3.05505	2.516611	1.154701	2.081666	0.707107	
standard error	0.881917	0.57735	1.20185	1.763834	1.452966	0.666667	1.20185	0.5	

	0.6m	**0.7m**	**0.8m**	**0.9m**	**1m**	**2m**	**3m**	**4m**	**5m**
Unharvested-1	3	4	2	2	3	6	5	4	4
Unhar.-10	3	3	3	3	3	8	4	6	10
Unhar.-20	2	2	2	2	2	4	5	5	9
sum	8	9	7	7	8	18	14	15	23
mean	2.666667	3	2.333333	2.333333	2.666667	6	4.666667	5	7.666667
standard deviation	0.57735	1	0.57735	0.57735	0.57735	2	0.57735	1	3.21455
standard error	0.333333	0.57735	0.333333	0.333333	0.333333	1.154701	0.333333	0.57735	1.855921

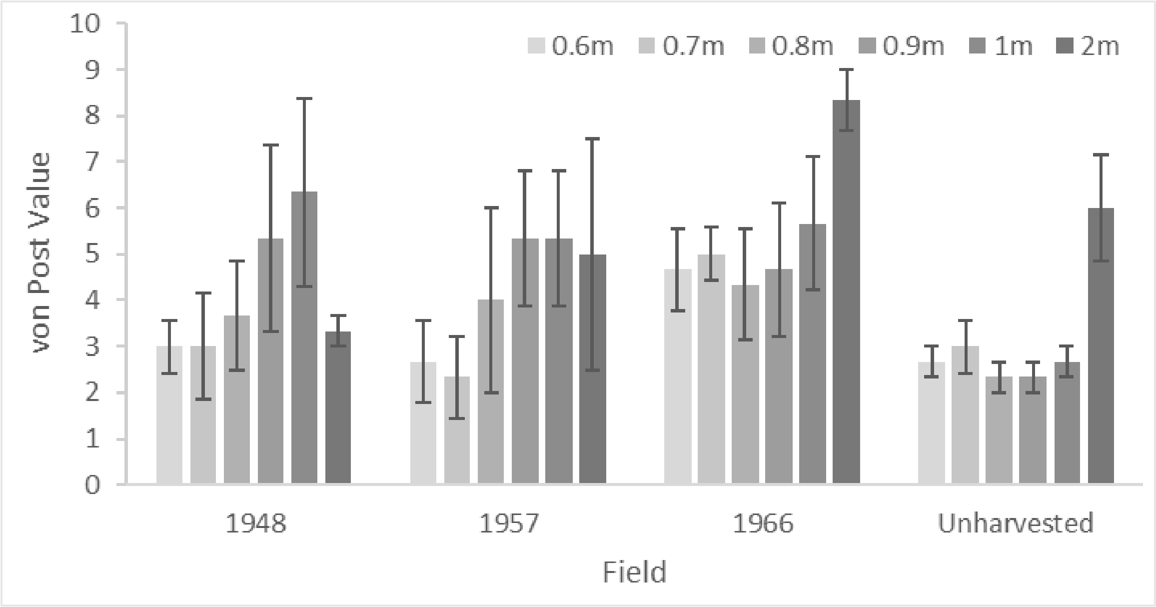


Figure A-2. **Bar graph demonstrating mean degree of peat humification in Burns Bog in June and July, 2018 (n = 3 per field). Error bars represent standard error of the mean.**

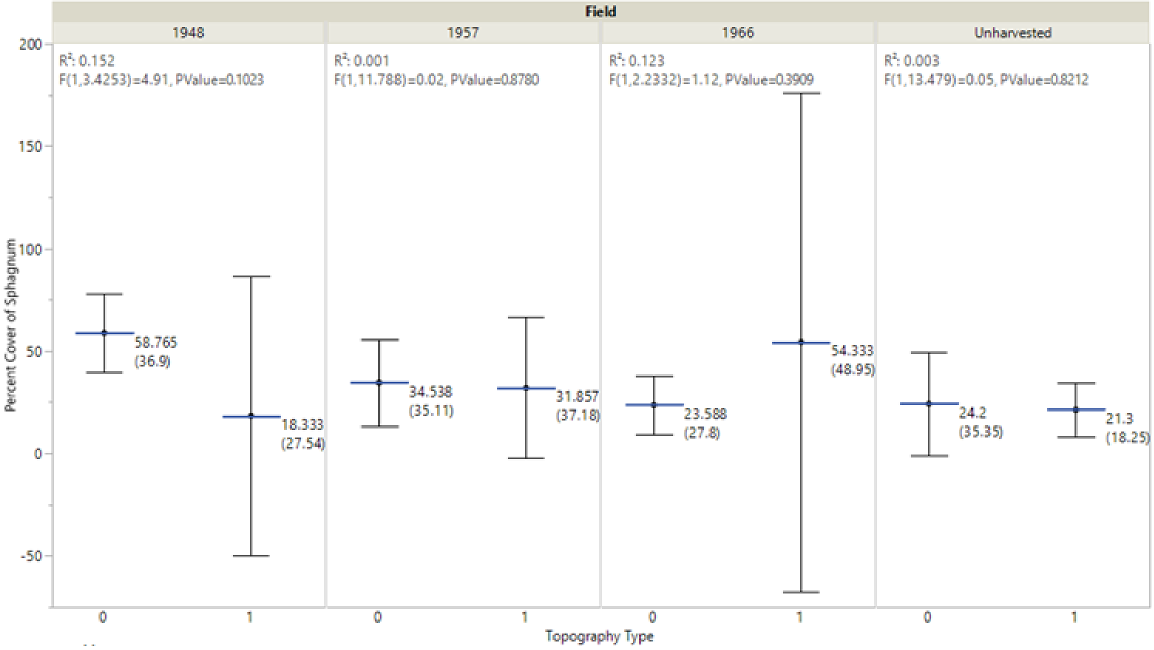


Figure A-3. **Linear regression comparing topography type to *Sphagnum* coverage, by field, in Burns Bog during June and July, 2018 (n = 20 per field, unequal variances, 95% confidence intervals). Topography type 1 refers to a complete hummock and 0 to other (i.e., lawn, hollow etc.).**

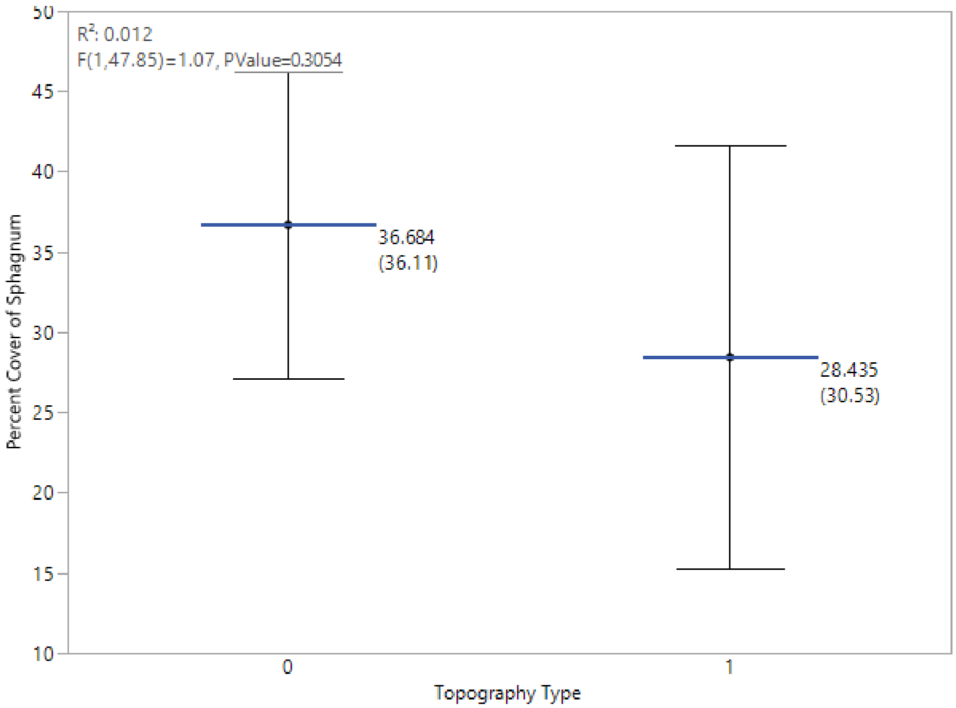


Figure A-4. Linear regression comparing topography type to *Sphagnum* coverage in Burns Bog during June and July, 2018 (n = 20 per field, unequal variances, 95% confidence intervals). Topography type 1 refers to a complete hummock and 0 to other (i.e., lawn, hollow etc.).