INTERACTING EFFECTS OF POST-WILDFIRE HYDROPHOBICITY AND VEGETATION RECOVERY IN A POOR FEN PEATLAND

INTERACTING EFFECTS OF POST-WILDFIRE HYDROPHOBICITY AND VEGETATION RECOVERY IN A POOR FEN PEATLAND

By Brandon MacKinnon, B. Sc.

A Thesis Submitted to the School of Graduate Studies in Partial Fulfillment of the Requirements for the Degree Master of Science

McMaster University

MASTER OF SCIENCE (2016) McMaster University (School of Geography and Earth Sciences)

TITLE: Interacting Effects of Post-Wildfire Hydrophobicity and Vegetation Recovery in a Poor Fen Peatland

AUTHOR: Brandon MacKinnon, B. Sc. (McMaster University)

SUPERVISOR: Professor J.M. Waddington

NUMBER OF PAGES: xi, 113

ABSTRACT

To investigate the prevalence and magnitude of hydrophobicity in near-surface peat, a poor fen was characterized into four main post-fire microforms: i) severely burned hollows (SB-H), ii) severely burned *Sphagnum fuscum* hummocks (SB-Sf), iii) lightly burned *S. fuscum* hummocks (LB-Sf) and, iv) lightly burned feathermoss lawns (LB-F). The SB-H possessed the most hydrophobicity at the surface (85 ± 20 s) and increased at the 2 cm depth (183 ± 35 s). In comparison, the LB-F experienced an increase in hydrophobicity from the surface (44 ± 10 s) to 5 cm (323 ± 32 s) and remained high to the 10 cm depth (211 ± 31 s). Results on *Sphagnum* recovery show that only LB-Sf are recovering and the SB-H show marginal recovery of pioneer species such as *Ceratodon purpureus* and *Polytrichum strictum.* Moreover, *S. fuscum* had a mean surface cover of 56 ± 5.9% in the LB-Sf and both pioneer species together possessed a total cover of 15 ± 4.4% in the SB-H. While the vascular cover was correlated with increased transplant productivity which in conjunction with moisture availability (preference for hydrophilic substrate), transplant size (15cm diameter preferred over smaller colonies), and transplant location (SB-H preferred over LB-F) should all lead to decreased mortality in treatments.

However, each species possesses slightly different characteristics that may be more desirable under reclamation conditions. Species that typically form hummock microform types like *Sphagnum fuscum*, *Sphagnum magellanicum,* and to some extent *Sphagnum angustifolium* can retain moisture under dry conditions (Clymo and Hayward, 1982; Andrus, 1986) and may be optimal for areas experiencing droughts or water limitations. Areas that are commonly inundated with water may benefit from a species that grows

through lateral expansion such as *Sphagnum angustifolium, Sphagnum riparium, or Sphagnum squarrosum* (Andrus, 1986). With *S. angustifolium* possibly being the best generalist due to its ability to remain photosynthetically active throughout a large range of moisture contents, tolerate desiccation, and grow rapidly (Silvola and Aaltonen, 1984; Andrus, 1986).

ACKNOWLEDGEMENTS

I would like to thank my supervisor Dr. Mike Waddington for his unwavering support and guidance. Our talks always kept me focussed and set on the right path whenever I was confused or needed reassurance. During our discussions he inspired me to think about the story that was being told. His insightful questioning left me with a stronger sense of what I was trying to accomplish and gave me confidence to persevere.

I would also like to thank Dr. Gustaf Granath whom offered great help with experiment structure and excellent R assistance. The minefield I call R would not have gone as smoothly without our enormous email strings discussing issues I had with the hemispheric photos. My third chapter, experiment on *Sphagnum* application, would not have been possible without his guidance on the methodology.

I would like to thank Corina Zuber for her assistance in the field throughout my second field season. I would not have been able to finish the work necessary for my thesis without her and I will be forever grateful. Her always positive demeanour made even the hottest days in a burned peatland whilst being eaten by mosquitos seem not so bad after all.

I will always cherish my late nights talking "*Sphagnum*" with Maxwell Lukenbach. His peatland talks and MATLAB assistance were invaluable to my thesis and he was always there for me as a friend and co-worker when I needed him. I would also like to thank Dr. Paul Moore for his assistance on LAI and other MATLAB activities I would have spent hours struggling without his guidance.

I would like to thank the rest of the MAC Ecohydrology Lab Group including Cam Irvine, Kelly Hokanson, Sarah Irvine, Cameron McCann, and Adriana Williams for their assistance in the lab and the field.

I would like to thank Syncrude and NSERC for without their funding this thesis would not have been possible. Lastly I would like to thank my mom and dad for providing me with those great left overs in the fridge every late night I came home hungry. Although every "starving student" has their own challenges having an excellent household to focus and work in was invaluable and not paying rent was a significant bonus.

TABLE OF CONTENTS

LIST OF FIGURES

Figure 3.1: Diagram of transplanted colony and fragment plots. The 20 cm buffer area with same treatment as subplots can be seen around the 3 x 3 grid. The colonies can be seen in the corners of the 3 x 3 subplot grid with the fragments located between them. The central subplot is the control area to determine success of fragments over natural regeneration.

Figure 3.2: Microhabitat colony plot with the preferred arrangement of colonies in two parallel linear lines of increasing size. Species type along the size increase was random and this method only chosen if surface slope was negligible.

Figure 3.3: Microhabitat transplant plot arrangement if the colonies could not be transplanted in two linear parallel lines, while still abiding by the 20 cm limit between species type and size (shown with the black arrows).

Figure 3.4: Transplanted colony survival rating average from the 1-4 scale at the end of summer, 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

Figure 3.5: Colony surface area increase using the installation diameter of 5 cm and final diameters measured in August of 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

Figure 3.6: Colony average chlorophyll fluorescence during summer of 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

Figure 3.7: Fragment survival rating average from the 1-4 scale at the end of summer, 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

Figure 3.8: The relationship between canopy closure and chlorophyll fluorescence of all active colonies on August 19th, 2014 in lightly burned feathermoss lawns. The trend is statistically significant with $p<0.001$.

Figure 3.9: The relationship between canopy closure and chlorophyll fluorescence of all active colonies by type in lightly burned feathermoss lawns on August 19th, 2014. *Polytrichum strictum* is statistically significant ($p<0.001$) however the other three species are not. Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

Figure 3.10: Survival rating of the transplanted microhabitat colonies by species and size from the 1-4 scale at the end of summer, 2014. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the colony diameter at installation.

Figure 3.11: The percent increase from the original size until end of summer 2014. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation.

CHAPTER 1: INTRODUCTION

Wildfire is the dominant disturbance in Alberta Boreal Plains peatlands, which under the right environmental conditions and location can result in the release of large amounts of carbon to the atmosphere via combustion (Turetsky et al., 2002; Hokanson, 2014). The loss of this long-term sink can take many decades to centuries to re-sequester into a resilient state (Kasischke and Turetsky, 2006; Benscoter and Wieder, 2003; Lukenbach, 2015). Moreover, because climate change is expected to increase the frequency of large and severe fires (Flannigan and Wotton, 2001; Flannigan et. al., 2005) there is concern that an enhanced release of CO_2 from peatlands via combustion may result in a positive feedback to climate change (Kasischke and Turetsky, 2006). This is especially important because peatlands pose a significant threat to wildfire fighting personnel through high intensity burning of black spruce stands and high ignition probability of late successional stage areas (Cumming, 2001; Rupp et. al., 2002; Krawchuk et. al., 2006). Consequently, there is an important need to restore the carbon sequestering potential of peatlands as rapidly as possible following wildfire, potentially through climate change adaptation/restoration strategies. Current estimates suggest that it takes ~20 years post-fire for burned peatland ecosystems to once again become net carbon sinks (Wieder et al., 2009). Moreover, Benscoter and Vitt (2008) indicate that it takes ~20-80 years since fire for *Sphagnum* to once again become the dominating species, making this avenue of investigation important to current and future environmental issues. This is especially important because peatlands also offer a variety of other ecosystem services including water production and acting as natural refugia to keystone species (Rochefort, 2000;

Reed et al., 2014). Numerous studies have investigated ways of restoring the ecological community of peatlands following peat mining including the use of companion species and the application of *Sphagnum* fragments (diaspores) (Campeau and Rochefort, 1996; Rochefort et al., 2003; Cobbaert et al., 2004; Graf and Rochefort, 2008). While this has been a successful approach to reverse ecosystem regime shifts and re-sequester lost carbon stocks several key factors influence peatland restoration success, most importantly restoration of the soils moisture regime (Campeau and Rochefort, 1996; Price and Whitehead, 2001; Schouwenaars and Gosen, 2007). This is usually accomplished through the blockage of drainage ditches and reducing surface roughness to have consistent moisture conditions throughout the peatland (Gonzalez et. al., 2013). In addition, it is necessary to reduce the surface evaporative demand such that the applied diaspores are not subjected to stressful conditions (Graf and Rochefort, 2010). This is accomplished by promoting the growth of companion species like small shrubs and other vascular plants (Graf and Rochefort, 2010) or through the application of straw mulch that protects the newly germinated spores (Price et al., 1998; Cobbaert et. al., 2004). Although the application of straw mulch can be labour intensive. Moreover, these techniques may be successful for anthropogenic disturbances but changes to the surface as a result of wildfire would make these techniques obsolete under certain conditions.

1.1 Hydrophobicity

Recent research suggests that a hydrophobic (water repellent) layer present in peatlands post-fire could be detrimental to vegetation recovery through a reduction of water fluxes to the surface (Kettridge et al., 2014). Although the presence of a hydrophobic layer post-fire could be beneficial to moss growth in some areas as reduced surface evaporation could lead to sustained moisture regimes in areas suitable for moss regeneration that would otherwise be inhospitable (Kettridge et al., 2014).

During fire heat can volatilize organic compounds typically originating from plant material, which condense on the near-surface substrate to create a hydrophobic (water repellent) layer (Doerr et al., 2000). The substrate profile is usually characterized by a hydrophilic or less hydrophobic ash layer above a layer of hydrophobic peat that was heated yet did not combust during the fire. It is this heated layer, typically around 2-5 cm deep (Kettridge et al., 2014), which possesses the greatest hydrophobicity and although not directly at the surface subsequent water demands cannot be met due to its presence. The Alberta boreal region undergoes decadal wet and dry periods (Devito et al., 2012) that could possibly affect the presence and persistence of hydrophobicity depending on where the fire occurs in the cycle of fluctuating water levels, thus influencing the available moisture within the peatland. Hydrophobic substrates possess a moisture threshold that must be surpassed in order to inhibit any hydrophobic characteristics within the peat (Berglund and Persson, 1996; DeJong et al., 1999; Dekker and Ritsema, 2000; MacDonald and Huffman, 2004). If recovery is occurring in a wet period this might be easily attainable, yet under dry conditions the areas affected within the peatland could be large.

Hydrophobic media retains its hydrophobic properties until the moisture content, specific to an individual sample, is surpassed. Once the moisture content threshold is surpassed in the hydrophobic peat (Berglund and Persson, 1996; DeJong et al., 1999; Dekker and Ritsema, 2000; MacDonald and Huffman, 2004) the moss should be able to retain moisture to supply the surface with enough water. The presence of water at the surface has shown to be of critical importance to the regeneration of some moss species (Price et al., 2002; Schouwenaars and Gosen, 2007; Lukenbach et al., 2015).

1.2 Companion Species

It is widely accepted that vascular plant regeneration is beneficial to moss recovery (Campeau and Rochefort, 1996; Rochefort et al., 2003; Cobbaert et al., 2004; Graf and Rochefort, 2010) and since moss growth is a primary vector for carbon sequestration (Benscoter and Vitt, 2008) it is important to understand the limiting factors of carbon sequestration and vascular vegetation recovery dynamics.

It is hypothesized that near-surface hydrophobicity following fire does not have a strong influence on vegetative shoot production in common peatland vascular vegetation since the pre-fire structures already have connection to the water table and are not inhibited by moisture limitations at the surface (Karlin and Bliss; 1983; Mallik and Gimingham, 1985; Agren, 1989; Hebert and Thiffault, 2011). The shoots should be able to penetrate the hydrophobic layer to alter the near-surface environment. It is possible that the alteration of the substrate by the vascular vegetation may facilitate weakening or elimination of the hydrophobic layer allowing the moss cover to regenerate more rapidly. Moreover, the increased moisture as a result of vascular vegetation cover could reduce the magnitude of hydrophobic persistence in the peat. This is especially important since near-surface hydrophobicity has been shown to decrease vegetation germination through decreased moisture (Doerr et al., 2000) and would likely create a sub optimal environment for the *Sphagnum* revocery.

Peatland vascular vegetation such as *Rubus chamaemorus* and *Rhododendron groenlandicum* are common in Alberta boreal peatlands and provide shade to the underlying moss surface as mentioned previously. The vegetation would likely enhance the recovery of the underlying moss by creating a microclimate more suitable for growth (Campeau and Rochfort, 1996). The vascular vegetation cover increases near-surface relative humidity and decreases diurnal temperature variations creating a more habitable environment (Graf and Rochefort, 2010).

R. groenlandicum can sprout from root heads and rhizomes that are found up to 50 cm below the peat surface (Hebert and Thiffault, 2011). The rapid post-fire recovery of *R. groenlandicum* is attributed to this characteristic as burning rarely eliminates all reproductive organs of the plant below ground (Hebert and Thiffault, 2011). Conversely, plants such as *R. groenlandicum* are unable to recover, germinate, or root in areas inundated with water or where the water table is close to the surface (Karlin and Bliss, 1983). As such areas with high burn severity close or below the water table would not be colonized by vascular vegetation; although since moisture is in excess at these locations they should be of low priority given what has been previously stated about surface moisture and *Sphagnum* recovery.

Each species alters the near-surface environment to varying degrees based on, but not limited to, the physical structure, height, and density of the vegetation (Graf and Rochefort, 2010). Research on companion species has primarily been focused on reclaiming harvested peatlands and restoring carbon sequestration potential. Results indicate that hummock forming species such as *Sphagnum capillifolium* reach photosynthetically active radiation (PAR) saturation at values of 400-600 µmol m^{-2} s^{-1}

(Chong et al., 2012) which given growing season averages of approximately 1400-1600 µmol m^{-2} s^{-1} (Chen et al., 1997) shows that optimal growth should occur under partial or shaded conditions (Hayward and Clymo, 1983) that would also serve to increase surface moisture. The study by Chong et al. (2012) indicates that increasing vascular biomass has a negligible effect on moisture at 10 cm depth within the peat column; therefore there is low risk of reducing water availability for the surfaces mosses through the presence of vascular species. Furthermore, in a bog, similar to the poor-fen investigated in this study, evapotranspiration is dominated by water loss from the moss layer and it is suggested that increasing biomass serves to limit water loss through decreased wind speed at the surface (Heijmans et al., , 2001). On the other hand, too much vascular vegetation can limit PAR availability at the surface and reduce CO_2 exchange sufficiently enough to cause moss mortality (Chong et al., 2012). However, this is only likely to happen under high nutrient conditions similar to fertilizer application studies (Berendse et al., 2001; Chong et al., 2012).

1.3 Diaspore and Colony Transplants

Research has shown the presence of a persistent spore bank within the peat (Sundberg and Rydin, 2000), which potentially acts to benefit post disturbance moss recovery. This is important in a post-fire recovery scenario as the hydrophobic layer would likely severely reduce any success that stored spores, within the peat profile, would have at germination. The application of diaspore fragments on the surface would show whether the surface is largely inhospitable to spores, making the presence of a spore bank obsolete in a post-fire recovery scenario. This process could be important in a reclamation scenario, as studies in harvested peatlands have shown that degradation of surface peat layers often form a dry biocrust that inhibits recovery (Salonen, 1987; Groeneveld and Rochefort, 2002; Rochefort et al., 2003; Rochefort et al., 2013). Desiccation of the surface peat could increase the persistence of hydrophobicity and inhibit spore germination. There is also evidence that nutrient release from the litter of vascular plants is needed for successful establishment of *Sphagnum* spores (Sundberg and Rydin, 2002). If the applied diaspores were to survive it would show that the surface environment could also be suitable for spore bank recruitment, however if survival were low then the substrate would likely not be suitable.

A study by Price and Whitehead (2001) sought to establish hydrologic thresholds for *Sphagnum* growth in cutover peatlands. Their study showed that areas with recovery were characterized by water tables near the surface (-24.9 +/- 14.3 cm), soil volumetric moistures greater than 50%, and pore water pressure greater than -100 mb (Price and Whitehead, 2001). In burned peatlands these values may only be possible in severely burned hollows that have lost enough peat elevation to put them in close proximity with the water table (Lukenbach, 2015).

Alternatively, the use of transplanted *Sphagnum* colonies, which are inserted into the substrate, could potentially be able to cope if the hydrophobic layer was breached. However, if the colonies are not inserted deep enough the hydrophobic layer would likely limit upward water movement (Doerr et al., 2000) to the *Sphagnum* capitula and subsequently lead to mortality. Colonies such as this may be subject to Allee effects that limit growth or production if a certain mass is not obtained within a short period of time, since the environment that they were collected from would not be moisture limited, they could lack high enough capitula density to resist desiccation (Elumeeva et al., 2011).

Allee effects are evident when a population needs to reach a certain size in order to possess a high chance of survival. Until a certain size is achieved the population remains highly vulnerable to disturbance and has a low chance of survival. As colony size increases so should the resilience and coinciding moisture within it (Hope et al., 2005). Through increased moisture on the periphery of the colony there should be a reduction in hydrophobic persistence, allowing the moss to recover laterally across the peat. It is therefore important to investigate the effects that vascular vegetation has on moss recovery and the breakdown of near-surface water repellency, as well as the efficiency of active restoration through the application of moss fragments or colonies.

A study conducted in cut-over bogs investigated sensitivity of recovering moss and concluded that the moss is most vulnerable when the *Sphagnum* mat thickness is 5-15 cm as it still relies on the connection to the higher density peat beneath it and cannot sustain its own moisture (Schouwenaars and Gosen, 2007). This may be pertinent for recovery of hollows post-fire as the peat there is generally of high density but also high moisture content. This would indicate that not only would the surface need to be dominated by *Sphagnum* mosses to return to a resilient state but also possess substantial growth (>15 cm deep) of new moss, thus emphasizing the need to increase resiliency within these systems pre and post-fire as efficiently as possible (Schouwenaars and Gosen, 2007). With hydrophobic peat limiting water loss from the peatland (Kettridge et al., 2014) hollows should stay moist and inundated with water.

1.4 Area Available for Recovery Post-Fire

It has been suggested that post-fire vegetation recovery is largely influenced by the pre-fire microform distribution, fire intensity, and depth of burn within the peatland

(Kovaleva and Ivanova, 2013). The peat moisture retention properties are dependent on microform type and could influence the presence of hydrophobicity, thus affecting bryophyte recovery. The fire intensity would be directly linked to depth of burn and in conjunction with the other variables, determine post-fire surface characteristics (Wein and Maclean, 1983; Kovaleva and Ivanova, 2013). Severity of the fire affects the proportion of the community that is present after fire, as burning can change the proportion of living biomass above and below the original surface (Wein and Maclean, 1983; Benscoter and Wieder, 2003; Benscoter, Wieder, and Vitt, 2005; Benscoter, Vitt, and Wieder, 2005; Benscoter and Vitt; 2008; Kovaleva and Ivanova, 2013). Since severely burned hummocks, lightly burned hummocks, severely burned hollows, and lightly burned feathermoss lawns all possess extremely different post fire characteristics (Lukenbach, 2015), fire severity is extremely important as a greater or lesser depth of burn can influence the fraction of these microforms on the landscape. This implies that if a lightly burned area was instead severely burnt that the location could be changed to a severely burned hollow through the loss of elevation upon combustion, which would depend on the pre-fire moisture contents and layering within the substrate. Ultimately the pre fire microform distribution would determine the area of the peatland that is able to actively recover post-fire due to moisture regime differences among microform types (Lukenbach et al., 2015).

1.5 Thesis Objectives and Arrangement

The study seeks to contrast the success of diaspore application through *Spaghnum* fragments against the transplanting of small colonies. It is hypothesized that when the colonies are installed to a depth of 5 cm or greater (Kettridge et al., 2014), in most cases, the moss should have a sufficient supply of moisture to survive. Furthermore, identifying the spatial coverage and magnitude of hydrophobicity within peatland microforms post-fire, specifically how hydrophobicity influenced vegetation recovery will be of importance to peatland recovery and restoration of ecosystem services. As mentioned previously vascular vegetation should be relatively unaffected by the presence of hydrophobicity and allow byrophytes to recover in the suboptimal environments present post-fire. This was fulfilled using a broad scale point measurement of hydrophobicity within microform types, addressed in chapter 2, and a temporal experiment investigating the success of various bryophyte applications in the presence of hydrophobic peat and vascular vegetation (see Figure 1.1 for arrangement).

The following questions are investigated in this study:

1) Is bryophyte recovery inhibited by near-surface fire induced hydrophobicity?
2) Is vascular vegetation recovery unaffected by hydrophobicity and if so does it help restore suitable hydrological regimes for bryophyte recovery?
3) Does hydrophobicity limit the success of restoration efforts post-fire?

CHAPTER 2: HYDROPHOBICITY CONTROLS POST-FIRE PEATLAND VEGETATION RECOVERY

2.1 Introduction

The peatlands of western continental Canada encompass 2.1% of the world's terrestrial carbon yet occupy only 0.25% of the land area which is equal to 365 157 km^2 (Vitt et al., 2000). It is estimated that ~48 Pg of carbon is stored in these Canadian continental peatlands, yet it is predicted that increases in forest fire and drought (Vitt et al., 2000; Flannigan et al., 2000; Flannigan et al., 2005) due to future climate change will put added stress on the contemporary carbon storage potential of these peatlands. The contemporary carbon storage rate for peatlands in this region is estimated at 19.4 g C m^{-2} y^{-1} (Vitt et al, 2000) but fires have the potential to release a large amount of the long-term carbon stored in these ecosystems (Hokanson, 2014) and reduce the carbon accumulation rate for years to decades (Turetsky et al., 2002). Moreover, recent research has indicated that an increase in large fire and total burned area coupled with increasing temperatures (Kasischke and Turetsky, 2006; Turetsky et al., 2011a; Turetsky et al., 2011b) could force some peatlands to experience regime shifts (Kettridge et al., 2015).

While soil hydrophobicity (water repellency) has been well studied with over a thousand studies completed on the subject (Dekker et al., 2003) and even more recently in peatland ecosystems as well (O'Donnell et al., 2009; Kettridge et al., 2014), the means by which hydrophobicity is created is largely debated as many sources have been identified (Doerr et al., 2000). It is believed that organics near the surface are volatilized by the heat of the fire and migrate down into the sub surface peat. The compounds then condense under the surface to the substrate and depending on the soil characteristics, moisture content, and fire intensity the intensity at which these compounds condense could differ (Doerr et al., 2000). In the case of peatlands, there is potential for this process to be substantial and persistent (Kettridge et al., 2014).

While research has shown that near-surface hydrophobicity can be created or exacerbated post-fire based on the peat type and depth (O'Donnell et al., 2009; Kettridge et al., 2014) the spatial extent or temporal persistence is relatively unknown. The study undertaken by O'Donnell et al. (2009) showed minimal persistence of hydrophobicity within *Sphagnum* and feathermoss species. The study was conducted within 24 months post-fire on several peatlands and it was concluded that hydrophobicity persistence was relatively non-existent. In contrast, the study undertaken in northern Alberta 15 months post-fire showed significant and persistent near-surface hydrophobicity in a large percentage of the burned samples of feathermoss and *Sphagnum* species (Kettridge et al., 2014). The feathermoss exhibited hydrophobic tendencies in the field in burned and unburned settings; however, the hydrophobicity was greater in the burned feathermosses (Kettridge et al., 2014). The *Sphagnum* exhibited slight hydrophobicity in burned samples and essentially none in those that were unburned (Kettridge et al., 2014). It has also been suggested that desiccation of peat can exacerbate any hydrophobic tendencies that may be present (Valat et al., 1991).

Considering current predictions it is important to understand the spatial variation and temporal longevity of hydrophobicity within peatlands post-fire. It has been shown to have a lasting effect on the recovery of the peatland (Kettridge et al., 2014) and influence the trajectory of recovery based on the surface peat characteristics (Lukenbach et al., 2015). The influence of the surface peat characteristics has been outlined in current research in terms of recovery trajectory (Lukenbach et al., 2015), however the fundamental processes need more investigation to determine the spatial and temporal heterogeneity and longevity.

The aim of this research is to examine whether long-lasting near-surface peat hydrophobicity inhibits the moss recovery following wildfire. While the near-surface hydrophobic layer can be beneficial to limiting water loss in the peatland due to net water retention (Kettridge et al., 2014), the reduction in evaporation would also leave much of the peatland uninhabitable to germinating moss spores as they require high moisture contents and humidity at the surface to be successful (Sundberg and Rydin, 2002; Smolders et al., 2003; Koyama and Tsuyuzaki, 2010). Kettridge et al. (2014) showed that the botanical origin of the peat influenced post-fire hydrophobic tendencies. However, vascular vegetation should be unaffected by the presence of hydrophobicity if they possess the ability to regenerate vegetatively through pre-fire structures. Species such as *R. groenlandicum* should be able to regenerate through hydrophobic peat as it already possesses connection to the water table with below ground structures. Vascular cover on the surface of the peat can serve to reduce evaporative demand through a reduction in wind speed and solar radiation (Heijmanset al., 2001; Farrick and Price, 2009). The presence of moderate density vascular cover could provide *Sphagnum* and other moss species with more suitable recovery area. However, if the vascular cover is too dense it can result in growth inhibition due to light availability as seen in a study involving the effects of nitrogen fertilization and vegetation dynamics on moss growth (Chong et al., 2012).

The following questions are investigated in this study:

1) Is bryophyte recovery inhibited by near-surface fire induced hydrophobicity?
2) Is vascular vegetation recovery unaffected by hydrophobicity and if so does it help restore suitable hydrological regimes for bryophyte recovery?

2.2 Study Site

The study was conducted in the Utikuma Region Study Area (URSA) which is located ~350 km northwest of Edmonton, Alberta (Devito et al., 2012). The landscape is characterized by a large proportion of pond-peatland complexes which are under long term water deficits as PET≥P (Devito et al., 2012; Ferone and Devito, 2004). As the region is under a long term water deficit, peatlands in this region have adapted to limit water loss and as a result are able to persist in a sub-humid climate (Devito et al., 2012). The region is characterized by decadal wet and dry cycles (Devito et al., 2012) that could influence the recovery patterns of a site considering moisture supply is a main constituent in *Sphagnum* growth and has the ability to shape where certain species can recover (Gonzalez et al., 2013). The range of annual precipitation is 300 to 700 mm and an annual potential evapotranspiration with a much more narrow range of 490 to 550 mm. The site (adjacent to pond 17) (see Ferone and Devito, 2004) is within a 144.2 ha catchment and 41% of this area is covered by peatland ecosystems (Ferone and Devito, 2004). In May of 2011 the site was burned by the Utikuma Complex forest fire, a ~90 000 ha wildfire (SWF-060) which removed standing biomass and killed the majority of surface cover (vascular and bryophyte). There was no vascular vegetation and little bryophyte cover left at the site based on visual analysis of photos taken within several months post-fire. The site experienced heterogeneous depths of burn due to variation in microtopography and moisture influenced by species composition (Thompson and Waddington, 2013; Lukenbach et al., 2015). The average depth of burn throughout the site remains low at <0.05 m based on measurements collected and analyzed by Hokanson (2014). Prior to burning the dominant vascular species were *Picea mariana* and *R. groenlandicum*. The moss cover was heterogeneous but dominated by *Sphagnum fuscum* hummocks, *Pleurozium schreberi* lawns, and most likely *Sphagnum angustifolium* hollows. This was inferred through identifiable charred remains and singed moss surface. For analysis and comparison the microforms were defined similar to Lukenbach et al. (2015) comprising of 4 main microform types to describe the peatland. The burned *Sphagnum* was classified based on the presence or absence of capitula indicating light and severe burning respectively and referred to as lightly burned *Sphagnum fuscum* hummocks and severely burned *Sphagnum fuscum* hummocks. Feathermoss was classified as any of the following species; *Hylocomium splendens*, *Pleurozium schreberi*, *Ptilium crista-castrensis*, and *Thuidium abietinum* because it would be difficult to determine with certainty the species from charred remains. However, most areas were identified as *Pleurozium schreberi* and as such referred to as lightly burned feathermoss lawns. The final microform type, severely burned hollows, was not associated with a species as the botanical origin pre-fire could not be identified due to the degree of burning at these locations.

2.3 Methods

The study involved selecting a total of 160 sample locations distributed evenly within severely burned hummocks, severely burned hollows, lightly burned feathermoss lawns and lightly burned hummocks. Measurements of Volumetric Moisture Content (VMC), water table depth, field and lab hydrophobicity, depth to peat of *Sphagnum* botanical origin, slope, aspect, canopy closure, vegetative percent cover, destructive LAI, and chlorophyll fluorescence measurements were collected. The sample size for each type (n = 40-42, some extras were included if accidentally sampled) was chosen to give a spatial representation within the peatland while still accommodating the time requirements needed to minimize variation over time, as hydrophobicity can be influenced by recent precipitation and relative humidity (Doerr et al., 2002; Doerr and Moody, 2004; Nyman et al., 2014). The peat cores were analyzed and collected over a three day period with several days absent rain prior to commencement. This was done to ensure low variances between each day of the study. An equal number of each microform type was sampled on each day to ensure that differences between days were consistent between each microform type. Therefore, if there was an environmental influence on the WDPT between one day and another during collection, it should affect all of the microform types to the same degree.

2.3.1 Wetness, Water Table and *S. fuscum* Depth

An HH2 portable hand-held readout unit coupled with a WET sensor (Delta-T Devices) was used to determine the VMC in the field at the surface and depths of 2, 5, and 10 cm within the near-surface peat. Calibration curves were calculated through progressive drying from saturation of several different peat types. These included; lightly burned feathermoss lawn surface, deeper burned feathermoss, severely burned *S. fuscum* hummocks, lightly singed *S. fuscum* hummocks, S. angustifolium hollows, *P. strictum* hollows, severely burned hollows consisting of *Sphagnum* peat, and severely burned hollows consisting of needles and feathermoss type peat. The calibration samples were measured twice daily with the wet sensor and weighed to determine the water content. Samples were allowed to air dry for three weeks in the lab and a final reading with the wet sensor and scale was recorded. In the field surface type was identified so that the proper calibration could be applied. Moisture was measured just under the surface, so that all of the prongs were completely covered at depths of 2, 5, and 10 cm.

The water table within the peatland is flat, as shown in previous research by Lukenbach et al. (2014) and confirmed through the measurement of two groundwater wells on the periphery of the sampling area. Therefore, the water table measurements were representative of elevation differences and relative distances to the water table between plots. A perforated poly vinyl chloride (PVC) pipe was inserted into the ground next to the plot and other measurements were recorded as the water table stabilized within the tube. Given the usually high hydraulic conductivity of the surface peat this process took approximately 10-15 minutes.

In addition to the wet sensor measurements, samples were also taken back to the lab for further analysis. These samples measured approximately 10 cm X 10 cm X 10 cm and were not oven dried as they were to be used for WDPT tests to establish hydrophobic tendencies when moisture is controlled for field variance between samples, thus allowing any trends due to differences in peat composition and burn severity to be revealed.

The depth to *S. fuscum* botanical origin peat was measured in all sites. Comparisons were only performed on the lightly burned feathermoss lawns and the severely burned hollows as the other two microform types by selection always had *S. fuscum* material present at the surface. This was of importance due to the high moisture retention and capillary transmittance capabilities characteristic of this species. If it were closer to the surface the *S. fuscum* origin peat could supply more moisture to mosses regenerating at the surface than peat from other species.

2.3.2 Water Drop Penetration Test

Water Drop Penetration Test (WDPT) was performed in the field and the lab using the same protocol as Dekker et al (2000). Distilled water was dispensed using a pipette held just above the peat surface and seven equally sized drops applied. The time was measured upon contact until the complete infiltration of the drop on the surface of the peat. For logistical reasons the droplets were monitored until they were absorbed or until a maximum time of 600 seconds. This was done to optimize the amount of samples collected, as in the classification scheme used by Dekker et al. (2000), the next time interval is 600-3600 s. However this should be taken into consideration during the following analysis as it implies that some samples would have higher means than reported. The first class is hydrophilic characterised by penetration times <5 s, followed by slightly hydrophobic at 5-60 s, strongly hydrophobic at 60-600 s, and severely hydrophobic at 600-3600 s. Samples that were tested in the field were brought back to the lab and allowed to air dry for 2 months. The procedures used in the field were then repeated to determine the hydrophobic persistence of each burn type under standardized moisture content. The samples were not oven dried as this would have influenced lab results as shown in previous analysis indicating that even low oven temperatures can induce hydrophobic persistence (Crockford et al., 1991; Dekker et al., 1998; Doerr, 1998).

2.3.3 Vegetation Surveys

Vegetation surveys were conducted two weeks prior to the hydrophobicity tests using a 25 cm by 25 cm quadrat plot. The values were recorded as a percentage of the plot surface area and as such the vascular vegetation could be greater than 100% while the ground surface cover must equal 100%. This allows for a representative percentage for each vascular species in relation to the plot area.

Leaf Area Index (LAI) and gap light values were obtained utilizing two techniques, Digital Hemispheric Photography (DHP) and destructive sampling. DHP was conducted using a Sunex 185° SuperFisheye 5.6mm F/5.6 lens on a Canon 60D DSLR camera. The camera was pointed with the top of the frame facing north and held at the surface parallel with the slope to capture as much of the vascular leaf area as possible. The surface aspect and slope were recorded for each plot and included in the gap light analysis using the cimesr software package in R Studio (Gonsamo et al., 2011; Granath et al., 2014). The code conducts similar processes to other gaplight analyzer software except it incorporates slope and aspect as a radiation function and allows batch processing of files. The software package takes the file and extracts a circular image using specific calculated points based on the camera and lens used. It then extracts the blue channel pixels from the image and sets a threshold value to convert to a 1-bit black-and-white image. The blue channel is extracted as plants do not reflect the blue wavelengths well and the sky does (Zhang et al., 2005). The program takes the proportion of black and white pixels to ascertain the leaf area index or the amount of closed canopy above the plot location. Immediately prior to the hydrophobicity measurements all above ground biomass was collected and stored in large re-sealable plastic bags. The biomass samples were then returned to the lab where all of the leaves were manually removed from the stems of each plant and then sorted by species. Pictures were taken of each species per plot on a white board measuring 7848 cm^2. A Canon 60D camera was supported by a tripod above the white board and the corners of the photograph lined up with the whiteboard extent, exposure settings were done in manual mode to reduce any variation between pictures and an external flash used to provide even lighting. The pictures were captured in RAW format so that exposure, contrast, and other levels could be adjusted for conversion to 1-bit black-and-white images. The area ratios were then assessed using MATLAB to determine the leaf area in relation to the original plot area of 625 cm^2.

2.3.4 Microform Distribution Transect

A 50 m transect was placed east-west through the hydrophobicity and colony study areal extent. Surface cover was identified and assigned to one of the four microform types one meter to the left, middle, and one meter to the right of the transect every 0.25 m. The number of hits for each microform type were tallied and given a percentage out of 603 measurements. The area was identified to be one of the four categories: lightly burned *S. fuscum* hummock, severely burned *S. fuscum* hummock, lightly burned feathermoss lawn, or severely burned hollow. This was completed to get a microform distribution of the area being studied.

2.3.5 Statistical Methods

The data acquired was often skewed with high kurtosis values. This was due to the large number of zeros obtained during the measurements in terms of the hydrophobicity, leaf area index, and other measured variables. The values represent actual zeros and as such would not benefit from being transformed using simple statistical transformations.

However, due to usually large sample sizes a one-way ANOVA was completed followed by Holm-Sidak comparison of means. The Holm-Sidak method was chosen to compare the many means of different groups as it has more power than the Tukey and Bonferroni methods (Seaman et al., 1991). This was also beneficial since it can be used on any set of P-values and does not need to be performed post ANOVA analysis (Holm, 1979; Aickin and Gensler, 1996). However, one downfall is that this test cannot calculate confidence intervals although this is not as important given the large amount of samples and that they were specifically chosen based on surface composition and not at random between types. The analysis was completed using SigmaPlot version 11.0 and MATLAB 2012b.

2.4 Results

2.4.1 Field Hydrophobicity

The severely burned hollows have the lowest elevations and are closest to *S. fuscum* botanical origin peat. However, they exhibit more ($p<0.001$ for light hummocks and severe hummocks and $p<0.05$ for burned feathermoss comparisons) hydrophobic persistence at the surface (85 ± 20 s) than all other microforms tested. The burned feathermoss lawn (44 ± 10 s) is significantly greater ($p<0.05$) than the lightly burned *S. fuscum* hummocks (4 ± 2 s) but not the severely burned *S. fuscum* hummocks (20 ± 5 s) (Figure 2.1). The severely burned *S. fuscum* hummocks and lightly burned *S. fuscum* hummocks are not significantly different at the surface.

Hydrophobicity at the 2 cm depth increased substantially in the burned feathermoss lawns (224 ± 29 s) and the severely burned hollows (183 ± 35 s) and were significantly greater than the other two microform types ($p<0.001$) but not each other. The severely burned *S. fuscum* hummocks (33 ± 11 s) and the lightly burned *S. fuscum* hummocks (0 s) were not significantly different in their means however hydrophobicity is still more prevalent in the severely burned *S. fuscum* hummocks at this depth.

The burned feathermoss lawns were the only microform to experience an increase at the 5 cm depth (323 ± 32 s) and the only microform to be significantly greater than the others (p<0.001). The other three types were not significantly different from each other while the lightly burned hummocks possessed the lowest value of 0 s the severely burned hollows (23 ± 14 s) and the severely burned hummocks (30 ± 12 s) were not significantly greater.

The burned feathermoss lawn still possessed strong hydrophobicity at the 10 cm depth (211 ± 31 s) and significantly greater (p<0.001) than all other microforms, however all microforms have decreased at this depth. The other microforms are not significantly different than each other. The severely burned hollows and lightly burned hummocks now possess similar values (0 s) while the severely burned hummocks still seem to exhibit some hydrophobicity (6 ± 3 s).

2.4.2 Lab Hydrophobicity

All of the lab hydrophobicity microform means were greater than in the field, however the same general trends seem to exist but with stronger differences between types. The highest hydrophobicity is present in the severely burned hollows (409 ± 33 s) and is significantly greater (p<0.001) than all other microform types at the surface (Figure 2.2). The burned feathermoss lawns (197 ± 24 s) and the severely burned *S. fuscum* hummocks (261 ± 24 s) are not different from each other but are significantly different (p<0.001) from the other two microform types. As with the field analysis the lowest WDPT times were in the lightly burned *S. fuscum* hummocks (80 ± 22 s).

All of the means for the 2 cm depth were significantly different from each other. The severely burned hollows possessed the highest hydrophobicity (461 ± 28 s) and is significantly greater than all other types ($p<0.001$ and $p<0.05$ for the burned feathermoss lawns). The burned feathermoss lawns increased from the surface measurement almost two fold (380 ± 27 s) while the severely burned hummocks increased marginally to 287 ± 28 s. The lightly burned hummocks (65 ± 13 s) remained relatively constant and are significantly less than all other groups ($p<0.001$).

The burned feathermoss lawns continued to become more hydrophobic with depth (496 ± 21 s) and are significantly greater at 5 cm ($p<0.001$) than all other microform types. The severely burned hummocks increased slightly at 5 cm (312 ± 33 s) and the severely burned hollows decreased (347 ± 32 s) such that they were no longer significantly different from each other yet still strongly hydrophobic. The lightly burned hummocks stayed consistent in their hydrophobicity measurements (71 ± 15 s) and were significantly less ($p<0.001$) than any other microform type.

The trends for the 10 cm depth are similar to 5 cm, however the burned feathermoss lawns (505 ± 26 s) experience an increase in hydrophobicity and are significantly greater than all other microform types ($p<0.001$). The severely burned hummocks are almost identical at 10 cm (313 ± 28 s) as they were at the 5 cm depth and are not significantly different than the severely burned hollows (257 ± 29 s) which experienced a decrease of approximately 100 seconds in average water drop penetration time. The lightly burned hummocks (57 ± 12 s) remain significantly less ($p<0.001$) hydrophobic at all depths and experience a slight decrease from the previous values up to the surface. A frequency of all individual measurements and where they are located within the hydrophobicity ratings can be seen in Figure 2.3.

2.4.3 Microform Distribution Transect

The percentages calculated from with the transect measurements were: 12% for lightly burned *S. fuscum* hummocks, 26% for severely burned *S. fuscum* hummocks, 27% for lightly burned feathermoss lawns, and 35% for severely burned hollows.

2.4.4 Bryophyte Recovery

Only the lightly burned *S. fuscum* hummocks possessed significantly greater surface cover (56 ± 5.9 %) of *S. fuscum* ($p<0.001$) than all other types (Figures 2.4 and 2.5). *S. fuscum* was the only *Sphagnum* species to have a measureable recovery (consistently >5% in the plots) and is the species of primary concern. The burned feathermoss lawns had the lowest *S. fuscum* recovery, with zero recovery of any bryophyte species exhibited in all of the plots. The severely burned hollows experienced slightly higher recovery followed by the severely burned hummocks with 1 ± 1 % and 2 ± 1.6 % respectively, although none of these means were significantly different.

A summation of all other moss species excluding *S. fuscum* showed that all other bryophytes are only significantly recovering ($p<0.001$) in the severely burned hollows (15 ± 4.4 %) (Figure 2.6). All other groups possessed a mean of 0 and were not significantly different.

Firemoss (*Ceratodon purpureus*) and *P. strictum* (Figures 2.7 and 2.8 respectively) mosses possessed significantly higher values in the severely burned hollows ($p<0.001$) compared to the other three microform types. P. strichtum possessed the highest mean (9 ± 3.3 %) with firemoss occupying the remainder of moss cover in the hollows (7 ± 2.4 %).

2.4.5 Leaf Area Index

The highest total LAI was observed in the severely burned *S. fuscum* hummocks (1.2 ± 0.1) which was significantly higher than the burned feathermoss lawns and severely burned hollows ($p<0.001$). The severely burned *S. fuscum* hummocks were not significantly greater than the lightly burned *S. fuscum* hummocks (1.1 ±0.1). The lightly burned *S. fuscum* hummocks were significantly greater than the burned feathermoss lawns and severely burned hollows ($p<0.05$ and $p<0.001$ respectively). The severely burned hollows possessed the lowest LAI (0.26 ± 0.04) and was significantly less ($p<0.001$) than the burned feather moss lawns (0.8 ± 0.1) (Figure 2.9).

The severely burned hollows were the only microform to exhibit significantly less ($p<0.001$) *R. groenlandicum* leaf area (0.14 ± 0.04) than the other three types. The burned feathermoss lawns (0.75 ± 0.1), severely burned *S. fuscum* hummocks (0.75 ± 0.1), and the lightly burned *S. fuscum* hummocks (0.71 ± 0.06) did not exhibit any significant differences (Figure 2.10).

Rubus recovery was grouped into two statistically significant microform classes with the severely burned *S. fuscum* hummocks (0.22 ± 0.04) and the lightly burned *S. fuscum* hummocks (0.15 ± 0.03) not significantly different from each other, but are from the other two microform types ($p<0.001$ and $p<0.05$ for the lightly burned *S. fuscum* hummock compared to the severely burned hollow). The burned feathermoss lawns and severely burned hollows exhibit much lower LAI than the other two microforms (0.02 ± 0.01 and 0.06 ± 0.02 respectively) (Figure 2.11). These two microforms were not significantly different from each other.

The trends for *V. oxycoccos* and *V. vitis-idaea* were similar to *R. chamaemorus* however the differences between the two groups were even more pronounced with these two species being almost non-existent in the burned feathermoss lawns (0.02 ± 0.005) and severely burned hollows (0.01 ± 0.004). The lightly burned *S. fuscum* hummocks (0.22 ± 0.03) and the severely burned *S. fuscum* hummocks (0.20 ± 0.03) were significantly greater ($p<0.001$) than the other two microforms previously mentioned (Figure 2.12).

2.4.6 Depth to *S. fuscum*

The depth to *S. fuscum* was significantly different for all group types except the light and severely burned *S. fuscum* hummocks as they had *S. fuscum* present directly at the surface. The burned feathermoss lawns had the greatest depth to *S. fuscum* (0.228 ± 0.013 m (mean ± SEM)) with the severely burned hollows having an intermediate value (0.126 ± 0.008 m). All groups were significantly different from each other ($p<0.001$) except for the lightly burned hummocks and severely burned hummocks (Figure 2.13).

2.4.7 Depth to Water Table

All of the microform types were significantly different with respect to water table depth ($p<0.001$). The severely burned hummocks presented the greatest depth to water table (0.715 ± 0.018 m) followed closely by the lightly burned hummocks (0.657 ± 0.016 m).

The burned feathermoss lawns had an intermediate value (0.500 ± 0.011 m) while the severely burned hollows were represented by the lowest depth to water table (0.335 ± 0.017 m) (Figure 2.14).

2.5 Discussion

2.5.1 Hydrophobicity

The lab experiments were used to control for moisture differences, as under natural settings the microform types exhibited different depths to the water table, vegetation coverage, and other moisture influencing variables (Figure 2.15) which could have reduced differences between them and made identification of fire induced changes more difficult. Furthermore, hydrophobic persistence increased for all samples due to drying in the lab but the differences between them became more pronounced, displaying the influence that burn severity and botanical origin possess on hydrophobic persistence (Figures 2.1 to 2.2).

Under wet conditions the hollows do not usually exhibit hydrophobic tendencies (Lukenbach et al., 2015) likely due to high moisture contents that are great enough to exceed the threshold of hydrophobicity development in the substrate; however at the time of collection hydrophobicity was experienced in the hollows, primarily at the surface. Furthermore, after drying in the lab they remained the most hydrophobic and increased substantially compared to field measurements, indicating that under drier conditions, 35% of the peatland could potentially develop hydrophobic properties, possibly halting recovery in these areas. At the surface the severely burned hollows were the most hydrophobic microform class with an average double that for the lightly burned feathermoss lawns (Figure 2.1), but this quickly disappeared with depths greater than 2 cm for the hollows while increasing with depth in the feathermoss. The lightly burned *S. fuscum* hummocks exhibit slight hydrophobic tendencies and not surprisingly, continue to do so in the lab analysis (Figure 2.2), likely due to low amounts of heat and volitilization of organic compounds into their surface.

The hydrophobic tendencies of all of the microform classes except the lightly burned *S. fuscum* hummocks increased from the surface to the 2 cm depth, similar to what was observed by Kettridge et al. (2014). Just below the surface is typically the interface between heated and cool substrate and the location where volatilized organic compounds could condense (Debano, 2000; Certini, 2005). The severely burned hollows and feathermoss lawns both experience a large increase in WDPT and display large frequencies of severely hydrophobic areas (Figure 2.3).

While the severely burned *S. fuscum* hummocks experience a slight increase in hydrophobicity, it is important to note that in almost all cases the moisture increased to the 2 cm depth, likely due to the increased frequency of hydrophilic peat. The lightly burned *S. fuscum* hummocks decrease to a WDPT of 0 ± 0 s at the 2 cm depth and all subsequent depths. While the lab results for the severely burned and lightly burned *S. fuscum* seem to show stable frequencies with depth, with the severely burned *S. fuscum* being more hydrophobic, indicating that moisture supply and exposure to heat from fire influence the persistence of hydrophobicity in the field.

At the 5 cm and 10 cm depths the moisture threshold in all of the microform classes, except the lightly burned feathermoss lawns, is surpassed and as such there is generally low hydrophobic persistence. Conversely at the 5 cm depth hydrophobic values continue to increase in the feathermoss lawns, a feature that could be due to their low bulk density and naturally hydrophobic surface (Kettridge et al., 2014), promotes the creation of deep hydrophobic persistence. It is not until the 10 cm depth that the feathermoss lawns experience a reduction in hydrophobic persistence, likely due to increasing moisture.

Feathermoss does not possess the same moisture holding properties as *Sphagnum* mosses and as such would not have high surface moisture (Bisbee et al., 2001). Without moisture to cool down the feathermoss lawns, it could allow them to be subjected to a more gradual heat gradient that induces hydrophobic persistence and disperses the effects more vertically throughout the peat column. Furthermore, heat from the fire could have changed the structure so that it is more prone to desiccation and thus it could be the lack of moisture in conjunction with the compounds that leads to the severe hydrophobic persistence observed. It is suggested that future work investigate in depth the natural persistence of hydrophobicity within feathermosses to support this work and that done by Kettridge et al. (2014). Both the field and lab results display that burning and botanical origin of the peat have a large influence on the depth and severity of hydrophobic persistence within the peatland.

2.5.2 Microform Based Moisture Regime Differences

Interestingly there was a significant difference in the elevation of the severely burned *S. fuscum* hummocks and the lightly burned *S. fuscum* hummocks. Typically *Sphagnum* moisture stress develops at a pore water pressure threshold less than -100 mb (Price and Whitehead, 2001). Up to this point hydrological connection with the water table is usually maintained. The moisture retention ability is what helps *Sphagnum* mosses to resist burning. The severely burned hummocks seem to be at increased risk to burning due to their increased elevation over the lightly burned hummocks. As such the post fire environment is more suitable to growth in the lightly burned hummocks as they have not sustained as much modification due to fire and should have more moisture from closer proximity to the water table. There was only 5.8 cm difference (Figure 2.14) in average elevation between the two types however the lightly burned hummocks still had most of their capitula intact, while the severely burned hummocks did not, which could be attributable to the moisture differences.

The depth to *S. fuscum* could only be compared (and is only of importance) in the burned feathermoss lawns and the severely burned hollows. The burned feathermoss lawns possess the greatest depth to *S. fuscum*, however the severely burned hollows possess a greater frequency of near surface severe hydrophobicity. Although a large proportion of the hollows was still hydrophilic most likely lending to its increased bryophyte recovery over the feathermoss lawns (Figure 2.5). This is of critical importance to *Sphagnum* and other bryophyte recovery as a consistent supply of moisture is beneficial (Price and Whitehead, 2001). This is more likely if *S. fuscum* botanical origin peat is closer to the surface as it has a good moisture retention capacity and should promote growth. Thus, it is unlikely for any natural bryophyte regeneration to occur on the burned feathermoss lawns given their current hydrophobic persistence, depth to the water table, and moisture retaining *S. fuscum* (Figures 2.3, 2.14, and 2.13). With the current analysis on feathermoss hydrophobic persistence occurring 38 months or just over three years post-fire it is uncertain how long this hydrophobic peat may be resistant to recovery.

On the other hand, hydrophobic persistence was of little influence to the recovery of *R. groenlandicum,* which was the dominant shrub in leaf area measurements for all microform types except the hollows (Figure 2.10 compared to 2.9). The other species analysed preferentially grew in the hummock microform types. As shown in Chapter 3 the presence of *R. groenlandicum* in the lightly burned feathermoss lawns led to increased photosynthetic activity likely due to less harsh microclimates created by increasing LAI. The presence of *R. groenlandicum* has not previously shown to influence the recovery of moss species under natural regeneration, however Chapter 3 demonstrates that it can be utilized to increase the success of restoration efforts. The presence of the other species within the other microform types would inevitably assist in recovery as long as densities did not lead to shade-induced reductions in growth. The presence of *R. groenlandicum* in hydrophilic and severely hydrophobic substrate at the same densities supports the assumption that it can recover post-fire uninhibited by near surface persistent hydrophobicity. An explanation for its absence in the severely burned hollows could be that the surface roots were eliminated in deep burning and the proximity to the water table prevented the establishment pre-fire as saturated conditions have been shown to prevent root formation in this species (Karlin and Bliss, 1983).

The only microform type exhibiting *S. fuscum* recovery three years post-fire were the lightly burned hummocks (Figure 2.5). This is not surprising as it is also the only microform type that experienced very high hydrophilic tendencies and low alteration due to fire. The severely burned hummocks are hypothesized to be the next to return *Sphagnum* dominance however at this time that has not occurred. The quick recovery in the lightly burned hummocks is likely attributable to the high capitula preservation post-fire and thus the ability to regenerate vegetatively. Conversely the severely burned hummocks have been stripped of their capitula and display hydrophobic tendencies near the surface.

Even though the hollows do not have protection from a moisture altering vascular plant they are exhibiting recovery of *P. strictum* and *C. purpureus* (Figure 2.6). These mosses are pioneer species that normally initiate succession on post-fire peat as they are specialized to grow in harsh conditions (Johnson, 1981; Morneau and Payette, 1989; Maltby et al., 1990; Corradini and Clement, 1999). It is hypothesized that these mosses moderate the surface environment making it suitable for *Sphagnum* species to recover; specifically *P. strictum* has been shown as an effective nurse plant (Groeneveld et al., 2007). Even though *Sphagnum* species are not currently established in the hollows the presence of *P. strictum* and other pioneer mosses are beneficial to the establishment of *Sphagnum* under natural conditions and for remediation efforts to “fast-track” succession (Groeneveld et al., 2007).

2.5.3 Implications to Recovery, Remediation, and Resiliency

The microform distribution transect outlines the composition of the peatland and while it is a simplified representation of the microforms found within, it still represents the major microform types. The composition of the peatland would depend on the successional stage (Bisbee et al., 2001; Benscoter and Vitt; 2008), disturbance regime (Rydgren et al., 2004; Kettridge et al., 2015), and among other variables the hydrogeological position in the watershed (Hokanson, 2014). Consequently, the only areas recovering bryophytes are the severely burned hollows and the lightly burned *S. fuscum* hummocks which represent 38% of the surface area within the studied peatland. Given that recovery is unlikely due to high frequencies of hydrophobic peat present in the other microforms, the area available for recovery is quite small.

Hydrophobicity is still present in many areas three years post-fire with some microform types exhibiting tendencies of severe hydrophobicity. Therefore, it could be suggested that the hydrophobic peat may need to be broken down mechanically or through biological decomposition in order to reduce hydrophobic persistence. Considering *Spahgnum* has difficulties recovering on bare peat surfaces (Benscoter, 2006) that are not hydrophobic, the probability of regeneration on surfaces exhibiting moisture limitations from hydrophobicity might be very low. The post-fire vegetation recovery would be beneficial to the establishment of *Sphagnum* post-fire and potentially, remediation efforts as investigated in Chapter 3.

Hydrophobicity would be detrimental to recovery of *Sphagnum* mosses in those areas, although the decrease in evaporation may invoke some resilience to the system as the moisture limitations in these areas would help keep other areas moist. As *Spahgnum* has a difficult time establishing a stronghold on dry, hot, inhospitable peat it may be beneficial to the system to prevent other species from encroaching on the habitat that, through time, will be suitable once again to *Sphagnum* moss recovery. The decreased evaporation in the system as a whole may be beneficial for water conservation and promote sustainable runoff development (Kettridge et al., 2014) in the sub-humid environment found in the boreal plains of Alberta (Devito et al., 2012) during periods of drought.

It is unclear how long the hydrophobic and inhospitable peat will take to break down and become conducive to *Sphagnum* growth. Future research is needed to determine the best way to restore a system to exhibit *Sphagnum* dominated surface cover or develop cost effective management practices that would mitigate the presence of such surfaces post-fire. Forest management practices could be implemented to regulate peatland microform distribution as a wildfire prevention and severity reduction method. This could be possible if the surface cover of the peatland was maintained at a middle succession state that is dominated by *Sphagnum* mosses such as *S. fuscum*. Through natural post fire succession *Picea mariana* (black spruce) densities increase until much of the forest floor is shaded. Moisture holding hummock species such as *S. fuscum* are not well adapted to the light limiting environment and are out competed by feather mosses such as *Pleurozium schrebberi* which grow over the *Sphagnum* species in shaded areas (Bisbee, 2001; Benscoter and Vitt, 2008). The *P. schrebberi* does not hold moisture as well as *Sphagnum* and thus is more prone to combustion in a wildfire scenario and consequently takes longer to recover post-fire, as observed in this study.

The current conceptual model on peatland microform trajectory in the Western Boreal Plains is outlined by Lukenbach et al. (2015) and describes the typical recovery of these peatland complexes. The lightly burned feathermoss lawns have shown to have a very slow recovery with essentially no moss formation within three years post fire. On the other hand, the severely burned hollows and severely burned hummocks have shown a moderate recovery of pioneer species, which should facilitate the growth of *Sphagnum*. Conversely the lightly burned *S. fuscum* hummocks have the most rapid recovery time and depending on the wildfire, can largely resist damage to the capitula on the surface due to high moisture retention characteristics. Thus, microform distribution and botanical origin has a distinct influence on the resilience of the system, wildfire burning characteristics, post-fire recovery regime, and hydrophobic persistence.

CHAPTER 3: ASSESSING POST-WILDFIRE *SPHAGNUM* MOSS ESTABLISHMENT METHODS IN A BOREAL PLAINS PEATLAND

3.1 Introduction

Wildfire is the dominant disturbance in the boreal plains (Kasischke and Turetsky, 2006). On average, over 12 million ha are burned each year in the boreal biome through naturally and human ignited fires (Flannigan et al., 2009; Turetsky et al., 2010). The recovery of peatland mosses, particularly the *Sphagnum* genus species are often retarded due to inhospitable surface conditions (Benscoter and Vitt, 2008). Since moss spore availability has shown to be a non-limiting reagent in recovery (Campbell et al., 2003; Sundberg, 2013; Chapter 2) it is essential to modify the surface characteristics or facilitate recovery through bypass of the spore germination stage if the surface environment is unsuitable to germination. It is during the initial stages of recovery that the moss is most vulnerable to changes in its environment and as new organic matter starts to accumulate to a depth greater than 15 cm, the resistance and resilience of the community increases (Schouwenaars and Gosen, 2007).

Much of the historical research done on moss recovery in peatlands was situated in eastern Canada where pressure from harvesting has eliminated ecosystem services in many areas (see Waddington et al., 2002). Recovery in these peatlands is slow or nonexistent post-harvesting due to typical removal of the upper peat layers and life sustaining moisture conditions through trench construction. The limiting reagent of recovery is consistent moisture availability, which in harvested peatlands is commonly re-established through the blockage of drainage ditches and application of moss diaspores (Cobbaert et al., 2004; Graf and Rochefort, 2008; Graf and Rochefort, 2010; Gonzalez et al., 2013).

However, under natural conditions these do not efficiently recover and often do as poorly as the area left abandoned (Price et al., 1998; Price et al., 2002). The use of nurse plants or companion species has shown to be effective at increasing survivorship of the moss diaspores (Hope et al., 2005; Graf and Rochefort, 2010; Chong et al., 2012). The same effect is often accomplished through mulch that reduces evaporative demand and heat stress at the underlying peat surface (Rochefort et al., 2003).

Mulch and the presence of vascular plants native to peatlands such as *Eriophorum vaginatum* have been shown to considerably increase the survivorship of immigrating spores and seeds (Smolders et al., 2003; Farrick, 2008; Graf and Rochefort, 2010; Kayama and Tsuyuzaki; 2010). The positive impacts are experienced when vascular densities are great enough to reduce stresses from heat and moisture loss while still allowing enough PAR to sustain photosynthesis (Hayward and Clymo, 1983; Farrick, 2008). A study done in Australia focused on the use of transplanted *Sphagnum* colonies post-fire and exposed samples to different shading and fertilizer additions (Hope et al., 2005). However, the size of their colonies were much larger than those proposed to be used in this experiment ranging from fist-sized to approximately 20 x 20 x 30 cm (Hope et al., 2005). It is essential to determine the optimal colony size for transplanting to efficiently restore *Sphagnum* dominance and optimize for survivorship on an annual basis. Capitalizing on natural vegetation to increase the survival of transplanted colonies would reduce labour, however it is important to understand as much as possible about a multifaceted system like peatlands. Incorporating knowledge of *Sphagnum* stress thresholds and hydrophobic persistence post-fire would allow cost effective management practices to be implemented.

As shown in Chapter 2 and in other research hydrophobicity within burned peat is resilient to breakdown through saturation (Lukenbach et al., unpublished data) and persistent in a natural setting (Kettridge et al., 2014). The hydrophobicity is characteristic to specific burn types and depths and as such must be taken into consideration when deciding where to transplant the *Sphagnum* colonies (Chapter 2). The effectiveness of natural peatland vascular plants *Rhododendron groenlandicum*, *Rubus chamaemorus,* and *Vaccinium spp*. on recovery in conjunction with near surface hydrophobicity has not been investigated prior to this study. It is crucial to identify suitable species and colony size to determine best practices depending on the burn severity and/or area being reclaimed.

Peatland vascular species *R. groenlandicum* will be investigated to determine the benefits of transplanting colonies in moderately high leaf area index microforms. As shown in the second chapter, the recovery of this species is only considerably reduced in hollows. Furthermore, under wet climate conditions the presence of vascular plants in these areas would be less crucial considering the hollows would have surplus moisture that would be beneficial to moss recovery. Considering this and the implications that hydrophobicity has on moss recovery it would be beneficial to transplant colonies in areas of higher vascular cover if they are beneficial to recovery and as such know the optimal species and colony size for the procedure.

It is proposed that transplant success will be limited in areas of high hydrophobicity specific to microforms mentioned in the second chapter and that larger transplants will be more successful due to moisture retention capabilities allowing them to resist desiccation. Since hydrophobicity was shown to be depth dependent if the colonies are transplanted such that they are able to penetrate this layer they should possess a higher success rate when transplanted in areas with low near surface moisture availability.

The following questions are investigated in this study:

1) Is bryophyte recovery inhibited by near-surface fire induced hydrophobicity?
2) Is vascular vegetation recovery unaffected by hydrophobicity and if so does it help restore suitable hydrological regimes for bryophyte recovery?
3) Does hydrophobicity limit the success of restoration efforts post-fire?

3.2 Study Site

The study was conducted in the Utikuma Region Study Area (URSA) which is located ~350 km northwest of Edmonton, Alberta (Devito et al., 2012). The landscape is characterized by a large proportion of pond-peatland complexes which are under long term water deficits as PET≥P (Devito et al., 2012; Ferone and Devito, 2004). As the region is under a long term water deficit, peatlands in this region have adapted to limit water loss and as a result are able to persist in a sub-humid climate (Devito et al., 2012). The region is characterized by decadal wet and dry cycles (Devito et al., 2012) that could influence the recovery patterns of a site considering moisture supply is a main constituent in *Sphagnum* growth and has the ability to shape where certain species can recover (Gonzalez et al., 2013). The range of annual precipitation is 300 to 700 mm and an annual potential evapotranspiration with a much more narrow range of 490 to 550 mm. The site (adjacent to pond 17) (see Ferone and Devito, 2004) is within a 144.2 ha catchment and 41% of this area is covered by peatland ecosystems (Ferone and Devito, 2004). In May of 2011 the site was burned by the Utikuma Complex forest fire, a ~90 000 ha wildfire (SWF-060) which removed standing biomass and killed the majority of surface cover (vascular and bryophyte). There was no vascular vegetation and little bryophyte cover left at the site based on visual analysis of photos taken within several months post-fire. The site experienced heterogeneous depths of burn due to variation in microtopography and moisture influenced by species composition (Thompson and Waddington, 2013; Lukenbach et al., 2015). The average depth of burn throughout the site remains low at <0.05 m based on measurements collected and analyzed by Hokanson (2014). Prior to burning the dominant vascular species were *Picea mariana* and *R. groenlandicum*. The moss cover was heterogeneous but dominated by *Sphagnum fuscum* hummocks, *Pleurozium schreberi* lawns, and most likely *Sphagnum angustifolium* hollows. This was inferred through identifiable charred remains and singed moss surface. For analysis and comparison the microforms were defined similar to Lukenbach et al. (2015) comprising of four main microform types to describe the peatland. The burned *Sphagnum* was classified based on the presence or absence of capitula indicating light and severe burning respectively and referred to as lightly burned *Sphagnum fuscum* hummocks and severely burned *Sphagnum fuscum* hummocks. Feathermoss was classified as any of the following species; *Hylocomium splendens*, *Pleurozium schreberi*, *Ptilium crista-castrensis*, and *Thuidium abietinum* because it would be difficult to determine with certainty the species from charred remains. However, most areas were identified as *Pleurozium schreberi* and as such referred to as lightly burned feathermoss lawns. The final microform type, severely burned hollows, was not associated with a species as the botanical origin pre-fire could not be identified due to the degree of burning at these locations.

3.3 Methods

To investigate the hypotheses relating to the interactions that bryophyte and vascular vegetation cover have on the success of transplanted *Sphagnum* colonies and fragments, a two-year experiment was setup. The colonies were transplanted at the beginning of the first growing season and were monitored till the end of the second. The experiment was split into two sections; one that focused on the microhabitat of each colony and as such the influence of colony diameter on survival and another that focused on burn severity and influences of both bryophyte and vascular vegetation cover. The transplanted colonies consisted of three moss species: *Sphagnum fuscum*, *Sphagnum angustifolium*, *Polytrichum strictum*, and a mix of all three. The microhabitat experiment investigated colony size influence of two species, *S. fuscum* and *S. angustifolium,* in hydrophobic feathermoss lawns. This was done in order to determine the necessary colony size in order to survive such harsh conditions found within a large area of the peatland.

3.3.1 Transplanted Colony Treatments

The treatments were replicated with six blocks (Figure 1.1) that were characteristically similar in terms of microform composition, vascular vegetation cover, and burn severity. Half of the plots within each block were located in severely burned hollows and the other half in lightly burned feathermoss lawns. The eight plots within each block were then divided into four sub groups that were subjected to different treatments thought to influence the initial establishment of the fragments and colonies. These treatments were unaltered, all vascular cover removed, all bryophyte cover removed, and all vascular and bryophyte cover removed. Each 1m x 1m plot was then divided into nine separate cells in which the fragments and colonies were randomly distributed and inserted (Figure 3.1). The center cell in the 3 by 3 grid was kept as a control and no fragments or colonies were applied in this section. Each cell was 20cm X 20cm with a 20 cm buffer around the periphery. For the treatments where vascular vegetation was eliminated, the plot was cleared an extra 50 cm beyond the 1 m X 1 m plot to fully eliminate any edge effects the cover may exhibit. The four corners were randomly selected for colony transplants with the intermediate cells used for fragment application. The colony size was approximately 5 cm in diameter and 5 cm deep in order to maximize connection to the water table yet minimize disturbance to the donor plot and installation time. The fragments were counted so that each cell contained 50 moss capitula of each species attached to a 2.5 cm length stem. In the case of mixed fragments 17 of two species and 16 of one were randomly chosen and distributed evenly throughout the plot. For the colonies diameter remained the same and each species occupied roughly 33% of the colony surface area when viewed from above and were inserted to the same depth.

3.3.2 Transplant Microhabitat Colonies

This part of the experiment focused on the effect of colony size and species selection to determine the best practice for remediation. The colonies were half *S. fuscum* and half *S. angustifolium*. To reduce influencing factors characteristic of different microforms, as mentioned in Chapter 2, all colonies were inserted into lightly burned feathermoss lawns.

The experiment consisted of eight blocks each setup to follow specific guidelines to reduce variation. Each block contained a *S. angustifolium* and a *S. fuscum* colony with a diameter of 1 cm (3 capitula), 3 cm, 7 cm, 10 cm, and 15 cm. To ensure that the larger diameter colonies had sufficient contact within the peat and to keep consistency, the smaller colonies, were all installed to a depth of 10 cm instead of 5 cm like the fragment/transplant colonies. Colonies of the same size were placed no further than 20 cm apart to keep moisture variation as similar as possible and each colony of a different diameter was no further than 20 cm from another pair. When possible colonies were placed in linear parallel lines (Figure 3.2) however the species selection was random for each line. This was done to streamline the sampling method as it is more efficient to gather the data. If parallel lines were not possible the colonies were still placed within the distance limitations addressed above (Figure 3.3).

3.3.3 Moisture

Moisture inside and adjacent to the colony was measured on a weekly basis throughout the duration of the second growing season with an HH2 portable hand-held readout unit coupled with a WET sensor (Delta-T Devices). The WET sensor uses an electrical pulse and the delay between the two probes gives the moisture of the sample when calibrated with proper moisture retention curves, which were created using the adjacent peat and colony parent peat. Representative samples were collected near the plots and healthy moss samples from the donor peatland for the transplants. The samples were saturated and then periodically measured with the WET sensor and weighed to derive volumetric moisture content for each peat type for a given output from the sensor. However the calibration outside of the colony was used for the 1 cm diameter microhabitat colonies as they were too small for the prongs on the WET sensor to sample without including the adjacent peat.

3.3.4 Chlorophyll Fluorescence

Chlorophyll fluorescence was measured using a handheld ADC BioScientific Ltd. OS-30p+ Rapid Plant Stress Screening Device. The device comes with a set of dark adaptation clips that are typically used for measuring chlorophyll fluorescence on leaves. However, this possessed a logistical problem in measuring the fluorescence of the transplanted *Sphagnum* capitula. It would not have been possible to dark adapt the *Sphagnum* without disturbing the colonies due to their size and importance. Instead custom dark adaptation mats were constructed that would not damage the colonies and still yield accurate results. The mats were 25 cm in diameter and one fitted with a 3/8 inch tube the same size as the factory adaptation clips. This was only done to one of the covers as the others were placed on the peat surface for the 15 minute dark adaptation time and removed when measurements were to be taken. The mats covering the colonies were slid out from the one attached to the instrument so that a reading could be taken. The probe was then pressed up against an appropriate location on the colony so that a reading could be taken. If no location was found to respond to the probe then a reading of 0 was recorded for that colony as, at that time, it was not fluorescing enough for the instrument to measure at the given intensity. The instrument was calibrated through repeated measurements of the sample stress range and species and was set to give accurate readings for the largest range possible without having to change the settings throughout the measurement period. The settings used on the device were 80%, 20%, and 1 second for the modulation src power, saturation flash power, and width respectively.

The device is equipped with a solid state 660 nm LED with a saturating intensity of 525-6000 µ/m2s and a PIN photodiode with a 700-750 nm bandpass filter (BioScientific Ltd., 2013). It displays and records the Fo, Fm, Fv/Fm, Fv/Fo. For the purposes of the study only Fv/Fo was analyzed as it is more sensitive than Fv/Fm because it is normalized over the minimum fluorescence of the sample compared to the maximum (Baker and Rosenqvist, 2004). It is important to note that even readings of different sampling dates should be compared with caution as the pre sampling conditions can affect the reading values. In this case environmental factors like precipitation, solar insolation, and heat stress should be similar prior each sampling event.

3.3.5 Surface Temperature

Surface temperatures were measured using a Flir i5 Thermal Imaging Camera. However, the field of view on the camera is too narrow to take a thermal image of a single plot therefore the device was used to determine single location temperatures. The Flir camera was held above the peat surface until a constant temperature was displayed on the screen and then manually recorded. This was done for the surface of the colony and the surface adjacent to the colony or a suitable surface with similar slope and aspect.

3.3.6 Slope and Aspect

The slope and aspect were recorded using a compass and a protractor. To measure slope a metal pin was inserted vertically, using a leveling device, into the peat surface and a protractor held up against it to determine the angle from the vertical pin.

3.3.7 Capitula Density

When the colonies were transplanted their size was recorded and diameters measured perpendicular to each other. The measurements were oriented such that the colony represented an ellipse or circle as closely as possible. The same measurements were completed again at the end of the second growing season to determine the lateral expansion of each colony. Capitula density was determined using the area measurements and manually counting the number present in each colony at the end of the second growing season.

3.3.8 Survival Rating

A survival rating was created using an ordinal scale of 1-4 to represent the percent survival of each colony. The percent living for each class was >75%, 75-5%, 5-1%, and dead for 1, 2, 3, and 4 classes respectively. For the fragments the survival rating was slightly different in that the percent ground covered was >10%, 10-1%, few living shoots, and no living shoots for 1, 2, 3, and 4 respectively.

3.3.9 Gap Light Analysis

Gap light and radiation values were obtained utilizing Digital Hemispheric Photography. This was conducted using a Sunex 185° SuperFisheye 5.6mm F/5.6 lens on a Canon 60D DSLR camera. The camera was pointed with the top of the frame facing north and held at the surface parallel with the slope to capture as much of the vascular leaf area as possible. The surface aspect and slope were recorded for each plot and included in the gap light analysis using a software package ran through R Studio called cimesr (Gonsamo et al., 2011; Granath et al., 2014), described in detail in Chapter 2. The images were captured in RAW format so that exposure and levels could be adjusted to exaggerate contrast in the image. The image was then processed in R Studio in which the following actions were performed. The image was cropped based on calculated pixels to isolate the necessary data. The blue band of the image was then isolated because it has the best transmittance from the sky and the lowest reflectance from the vascular vegetation (Zhang et al., 2005). Lastly, this was combined with the slope and aspect measurements to give the gap light and radiation values at each colony.

3.3.10 Statistical Methods

The data was skewed and possessed high kurtosis readings and would not benefit from simple statistical transformations. However, due to usually large sample sizes a one-way ANOVA was completed followed by Holm-Sidak comparison of means. The Holm-Sidak method was chosen to compare the means of different groups as it has more power than the Tukey and Bonferroni methods (Seaman et al., 1991). However, one downfall is that this test cannot calculate confidence intervals although this is not as important given the large amount of samples and that they were specifically chosen based on surface composition and not at random between types. Moreover, the test was beneficial as it can be used on any set of P-values and does not need to be performed post ANOVA analysis (Holm, 1979; Aickin and Gensler, 1996). Linear regressions were used to determine if vascular vegetation had a beneficial impact on colony survival, however only values that had a measurable chlorophyll florescence were included in the analysis. The analysis was completed using SigmaPlot version 11.0 and MATLAB 2012b.

3.4 Results

3.4.1 Colony Rating

The colonies were rated on a scale from 1 to 4 as described previously. The results showed no significant trends for the four treatments; all vegetation left, all removed, only moss removed, and only vascular cover removed. The results seemed to be influenced primarily by differences between light and severe depth of burn (Figure 3.4). The lightly burned feathermoss lawns were similar to those chosen for the experiment in Chapter 2. All of the lightly burned samples were significantly worse than the colonies in severe burns with p<0.001 for all of the means except the light burn mixed colony when compared to the severe burn *S. angustifolium* and *P. strictum* which were p=0.004 and 0.003 respectively.

3.4.2 Colony Area Increase

Since all of the colonies were installed to have the same surface area their productivity was also analyzed based on the percent increase from transplanting. This was calculated using the installation area and the final area by the end of the second growing season. The colony that experienced the largest increase was the severe *S. angustifolium* (1351 ± 239 %) (Figure 3.5). As a result it experienced a significantly greater (p<0.001) growth than all of the colonies in lightly burned feathermoss. The severe mixed colonies also experienced a large increase in surface area (1130 ± 228 %) which was likely due to the inclusion of *S. angustifolium* within them as the *S. fuscum* and *P. strictum* experienced the least amount of growth. The severe mixed colonies were also significantly greater than all colonies in the lightly burned feathermoss (p<0.001) while the other colonies in severely burned hollows were not significantly different from those in lightly burned feathermoss.

3.4.3 Colony Chlorophyll Fluorescence

The colony chlorophyll fluorescence was analyzed based on the average value for the second growing season (Figure 3.6). All of the values were averaged since not all of the colonies may have been active based on the environmental conditions prior to sampling dates. The severe *P. strictum* (1.234 ± 0.212) had the highest chlorophyll fluorescence values and was significantly greater ($p<0.001$) than all of the colony types in the light burn. The severe mixed colonies (1.138 ± 0.155) had the second greatest chlorophyll fluorescence and was also significantly ($p<0.001$) greater than all of the colony types in the light burn. The severe *S. fuscum* (0.953 ± 0.138) did not do as well as the mixed and *P. strictum* however it was significantly greater ($p<0.001$) than all of the colony types in the light burn except the *S. fuscum* (0.402 ± 0.123). The severe *S. angustifolium* (0.775 ± 0.159) was not significantly greater than any of the light burn and not significantly less than any of the severe burn colony types.

3.4.4 Colony Elevation

To determine if elevation differences of the lightly burned and severely burned plots could be affecting moisture availability due to water table proximity a GPS survey was completed. Each plot center point was measured using a Trimble DGPS and then differences from this point for each colony was made using a ruler and water filled tube. The severe plots had an elevation of 656.535 ± 0.013 masl and were significantly lower ($p<0.001$) than the light burn plots at 656.614 ± 0.013 masl that were on average 0.08 m higher in elevation.

3.4.5 Fragment Ratings

Due to high spatial variation of the fragments, generally low success rate, and precision requirements for the chlorophyll fluorescence machine to conduct accurate field measurements a rating system similar to the colonies was used as described in the methods. However, the fragments did considerably worse than the colony transplants with averages in the high 3's for the light burn applications and high 2's to mid 3's for the severe burn applications (Figure 3.7). The severe mixed fragments (2.6 ± 0.3) and the severe burn *S. fuscum* applications (3.6 ± 0.2) did considerably better ($p<0.001$) than all of the lightly burned fragment applications.

3.4.6 Regression Analysis for LAI and Transplant Productivity

August 19th was the only measurement day with a sufficient number of lightly burned transplants fluorescing. The August 9th regression analysis proved to be statistically insignificant since there were not enough lightly burned colonies fluorescing. The summer had unseasonably low rainfall thus making it difficult to measure on most days. The August 9th measurements were taken a few hours after a 2mm rain event at approximately 9 am while the August 19th values were taken a few hours after a 3 mm rain event however they were taken at 4 am before the sun could evaporate much of the water. A polynomial regression analysis was done from 0 order to 3rd order polynomials to see which provided the best fit for the correlation. This was done for colonies in the severe burn and the light burn peat. Both the light and severe sample was only statistically significant under a linear regression (1st order polynomial) analysis. The trend between canopy closure and productivity was slightly more significant ($F = 6.177$, $p = 0.019$) for the lightly burned transplant locations (Figure 3.8) than severe ($F = 3.961$, $p = 0.05$). The individual species were then measured to see if there was a species specific trend that showed a correlation between canopy closure and chlorophyll fluorescence. Linear regressions were done for all species on August 9th and 19th and only the *P. strictum* showed a strong linear relationship (F = 40.420, p = 0.001) on August 19th (Figure 3.9). Only samples that had a reading greater than zero were included for the analysis. The chlorophyll fluorescence machines sensitivity was set such that it would give measurements for the greatest number of samples. Due to the time sensitivity of the measurements the chlorophyll fluorometer could not be re-calibrated for higher sensitivity at specific transplant locations. This means that measurements of zero are not necessarily dormant only that they are outside the range of the measurement device at its given calibration which provides justification for exclusion.

3.4.7 Microhabitat Rating

The microhabitat transplants of *S. angustifolium* and *S. fuscum* were rated using the same scale as the other transplant colonies. The larger colonies were on average much more successful than the smaller ones however this mostly applied to the 10 and 15 cm diameter colonies (Figure 3.10). Both the 15 cm *S. fuscum* and *S. angustifolium* (1.375 ± 0.375) were significantly healthier ($p<0.001$) than the 1 cm *S. fuscum* (4 ± 0) and *S. angustifolium* (3.625 ± 0.375). Furthermore, the 15 cm *S. fuscum* and *S. angustifolium* were significantly better ($p<0.001$) than the 7 cm *S. fuscum* (3.375 ± 0.375). The 10 cm *S. angustifolium* (1.750 ± 0.412) was significantly healthier ($p<0.001$) than both of the 1 cm microhabitat colonies while the 10 cm *S. fuscum* (2.000 ± 0.378) was only significantly healthier ($p<0.001$) than the 1 cm *S. fuscum*. Lastly, the 7 cm *S. angustifolium* (1.875 ± 0.398) was also significantly healthier ($p<0.001$) than the 1 cm *S. fuscum*. Although, it should be noted that the health range possessed by a colony rated a number two is rather large whereas the others possess a narrower scope which could skew the statistical results. However, the functional difference between transplants that have a higher percentage of visually healthy capitula versus a lower percentage may be quite small as both would be under significant stress. All other relationships were not significant enough to stay below the Holm-Sidak critical level of significance.

3.4.8 Microhabitat Percent Increase

The percent increase is fairly even between all sizes and species type. However, there is a slight increasing trend with increase in microhabitat size as seen in Figure 3.11. The only type that was significantly greater than any others was the 3 cm *S. angustifolium* (1556 ± 414 %). The 3 cm *S. angustifolium* was significantly greater ($p<0.001$) than all other species and size of microhabitat colony except the 3 cm *S. fuscum* (938 ± 204 %). Differences between other sizes and species were not great enough to be statistically significant. Percent increase was calculated from the difference in size at installation till the size at the end of the second field season.

3.4.9 Microhabitat Elevation

To verify that all of the different size colonies were in similar proximity to the water table their elevation differences were tested. Differences between the means of each group returned a positive result to the study meaning they were not significantly different ($F = 0.0647$, $p = 1.000$). Therefore, the samples are within the range acceptable for random sampling design.

3.4.10 Microhabitat Chlorophyll Fluorescence

To accurately test the health of the microhabitat transplants the handheld chlorophyll fluorescence was used similar to the other fixed colony size portion of the experiment as a more verifiable method than the rating system. The values are an average of the summer and do include 0 values in the comparison as they help to reduce the average, if for portions of the summer one size was doing well when others were not. The 15 cm *S. fuscum* (0.822 ± 0.198) was the most productive and is significantly greater ($p<0.001$) than both 1cm, both 7cm, and the 3cm *S. angustifolium* (0.184 ± 0.040). Conversely, the 15 cm *S. angustifolium* only does significantly better ($p<0.001$) than the 1 cm *S. fuscum* (0 ± 0) and the 1 cm *S. angustifolium* (0.034 ± 0.034). There is a general increasing trend similar to the percent increase in size that can be seen with an increase in colony diameter (Figure 3.12).

3.4.11 Microhabitat Area Increase

Similar to the percent increase this was done to determine the spatial coverage that the microhabitat colonies could cover from install till the end of the second growing season. However, this type of analysis does not take into account size differences between the colonies and instead only the actual area increase (cm^2). The 15 cm *S. angustifolium* (705 ± 179 cm^2) increased significantly more ($p<0.001$) than all other sizes and species type sub groups. The 15 cm *S. fuscum* (596 ± 73 cm^2) was not significantly greater than the 10 cm *S. fuscum* (283 ± 28 cm^2) or the 10 cm *S. angustifolium* (351 ± 105 cm^2) but was significantly greater ($p<0.001$) than all smaller sizes and species type sub groups. The 10 cm *S. angustifolium* was not surprisingly significantly greater ($p<0.001$) than both of the 1 cm microhabitat species sub groups which had means close to 0 for their area increase (Figure 3.13).

3.5 Discussion

3.5.1 Comparing and Contrasting Fragments and Colonies

Typical peatland restoration involves the *Sphagnum* transfer technique and the use of a straw mulch to protect the transferred *Sphagnum* (Rochefort et al., 2003). This study sought to investigate the survival of the applied spores under natural conditions and determine if naturally regenerating vegetation assisted in the success. Other research has demonstrated success with transplanted colonies (Hope et al., 2005) and as such a comparison of the two was investigated to determine the best species, application area, and application type on a naturally recovering peatland. The measurements on photosynthetic productivity were gathered with a handheld instrument for the colonies; however the chlorophyll fluorescence instrument could not be used on the majority of fragment treatment plots as the density of capitula was not high enough to give an accurate reading.

Therefore, a rating system was used to describe differences in survival between the colonies and the fragments. In general the fragment applications did poorly in all burn settings with all species, though slightly better in severe burn locations, as seen in Figure 3.14. Survival seemed to be limited to areas that stayed saturated for the majority of the sampling period, based on visual observation of the plots. It is evident that the natural vegetation density and/or the surface moisture regimes are suitable to inhibit the survival of the fragment applications. The fragments and colonies, as seen from their frequencies of survival, did better in the severely burned hollows where moisture was consistently higher throughout the summer of 2014 (Figure 3.15). The findings are consistent with current literature which state that in order for diaspore applications to be successful there needs to be high enough moisture at the surface (Price and Whitehead, 2001; Price et al., 2002), low pore-water tensions (Price and Whitehead, 2001), and sufficient protection from a mulch or nurse plant (Price et al., 1998; Groeneveld and Rochefort, 2002; Groeneveld et al., 2007; Farrick and Price, 2009), which under natural regeneration these expectations are not fulfilled in the feathermoss lawns. Furthermore, it is likely that the adverse conditions experienced in the feathermoss lawns are amplified by its increased distance from the water table and moisture retaining *Sphagnum* botanical origin peat. Overall, the colonies did consistently better in all areas, especially in the severely burned hollows with higher moisture contents (Figure 3.16) making them more ideal for any reclamation purposes in which re-establishment of *Sphagnum* is inherently difficult. The frequency analysis shows that restoration efforts should be focused on severely burned hollows instead of the lightly burned feathermoss lawns and, as seen in Chapter 2, these areas are naturally recovering pioneer bryophytes that have shown to be effective nurse plants (Groeneveld et al., 2007).

3.5.2 Companion/Nurse Species

Linear regressions were performed for one day's photosynthetic activity involving the severely burned, lightly burned and individual species (Figures 3.8 and 3.9) to determine the relationship of shading, through measurements of leaf area index, on photosynthetic activity. The trends observed were much stronger in moisture-limited situations whether analyzed by species or by colony location. Previous research has not focused on naturally occurring species in the boreal plains nor natural regeneration as companion species post-fire. As shown in Chapter 2, *R. groenlandicum* is the only species that can currently assume the nurse plant position on the feathermoss lawns but has shown to be beneficial particularly to the *P. strictum* colonies transplanted within them.

P. strictum does not possess the same moisture retaining hyaline cells that *Sphagnum* does and as such relies more heavily on external moisture for productivity (Frolking, 1997). It is hypothesized that when moisture becomes a limiting resource the degree to which nurse plants contribute to productivity increases. As a result, the only species to exhibit a significant relationship with LAI is *P. strictum* as its loosely packed stems and lack of moisture retaining organs leave it vulnerable to moisture loss at the peatland surface. It is important to note that the other species had smaller sample sizes which could be a contributing factor in their lack of significance. Interestingly when all of the species were used in a correlation to determine the effect of leaf area on the productivity based on severely burned hollows or lightly burned feathermoss lawns, the correlation was significant for the lightly burned feathermoss lawns and not the hollows. This could indicate that when moisture is limited, shading and modification of surface moisture is necessary to increase productivity of the colonies. The trends were likely less pronounced in the severely burned hollows due to shifts in resource significance, once water was no longer a limiting reagent. When both water and sunlight availability approach their optimum levels productivity may be dominated by other factors such as nutrient availability and/or competition. While the *Sphagnum* species still exhibited low survival in the presence of a natural companion species the benefits to *P. strictum* could be beneficial to future *Sphagnum* recovery as discussed previously.

3.5.2 Influence of Colony Size

The influence of colony size was investigated using colonies that were transplanted into the hydrophobic feathermoss lawns as they should offer more stable moisture conditions given the frequencies of hydrophobic persistence shown in Chapter 2 within the feathermoss lawns and similar elevations between the colony groups. Differences within and between each cluster were minimized by selecting locations with similar light exposure, elevation, and installed in close proximity to each other to minimize unforeseen differences. Larger colonies were consistently more productive, covered more surface area, and remained more photosynthetically active and as such it is advised that the largest feasible colony size be used for the best recovery. Larger colonies innately have a higher volume to surface area ratio which should allow them to retain water for longer periods of time, if all else is equal. As a result, the larger colonies were more resilient to periods of drought and had a much higher frequency of survival (Figure 3.17). Alternatively if larger colonies cannot be used smaller colonies may still be able to persist since *Sphagnum* has been shown to regain photosystem activity rapidly after rewetting, making them highly opportunistic to rain events (Proctor and Smirnoff, 2000; Hajek and Beckett, 2008). The large increase to the 3 cm diameter (Figure 3.11) suggests that there is be a threshold size that must be surpassed in order to retain adequate moisture to activate the photosystems long enough to maintain survival. A general trend for increasing colony size which corresponds to an increasing area increase can be observed in Figure 3.13. Species selection seemed to be of importance depending on the size of the colony and as seen previously on the location, however the larger colonies can store more moisture (Figure 3.18), thus likely allowing for more efficient operation of photosynthetic processes that is not feasible in the smaller colonies without a more substantial or consistent supply of moisture.

3.5.3 Species Selection and Applications

In order to restore *Sphagnum* dominance species must be selected based on growth rate, drought tolerance/resistance, moisture retention, and expansion characteristics. The ideal species must be able to grow fast and spread rapidly in order to restore original ecosystem functions within the peatland. The three species used were *S. fuscum*, *S. angustifolium*, and *P. strictum.* The *P. strictum* was chosen to determine if any facilitation would take place as a result of its presence and while this was partially observed in the mixed colonies (Figure 3.14) there was likely not enough time for facilitation to occur. The *P. strictum* seemed to be more active than the other species and as such mixed and pure *P. strictum* colonies were the most photosynthetically active but experienced lower lateral expansion. Alternatively, the other two species *S. angustifolium* and *S. fuscum* are species characteristic of late succession and possess valuable traits to ecosystem restoration (Rochefort, 2000). *S. angustifolium* has a larger niche and can survive in a wide range of conditions from hollows to intermediate microforms while *S. fuscum* is primarily considered a hummock forming species (Andrus, 1986). The *S. angustifolium* grows rapidly and due to its structure grows laterally instead of vertically, which would be beneficial to covering area within the peatland efficiently. With *S. angustifolium* possibly being the best habitat generalist due to its ability to remain photosynthetically active throughout a large range of moisture contents, tolerate desiccation, and grow rapidly (Silvola and Aaltonen, 1984; Andrus, 1986). Conversely, *S. fuscum* does not spread to the same degree but it maintains its resilience through vertical growth and investing resources to dense colony formation which leads to increased moisture retention (Figure 3.18) (Elumeeva et al., 2011). Due to its physical characteristics *Sphagnum* adds resilience and stability to the ecosystem which is beneficial for resisting forcing from climate and disturbances (Rochefort, 2000).

Both species possess desirable characteristics in ecosystem restoration and would create ecosystem stability through a greater diversity. They have both exhibited the ability to expand and survive on surrounding hydrophobic peat which should help to transform the inherent moisture regime; making the surface suitable for the colonization of less robust *Sphagnum* species.

Knowledge on successional trajectories in peatlands may assist in the development of forest management practices that promote surface moisture and maintain a resilient ecosystem to both disturbance and wildfire by optimizing the environment for desirable species. It would be important to select areas based on suitable water table positions and successional stage to modify the area in a way that it can be dominated by moisture retaining *Sphagnum* species. The practice could be as simple as thinning the forest to allow more PAR to reach the peatland surface, as observed in cut lines, that allow *S. fuscum* and other species to dominate and out compete the shade tolerate feathermosses (Bisbee et al., 2001). This process could be “fast-tracked” with the use of transplanted colonies which would breach any colonization barriers and mitigate any negative effects from the disturbance. Care would need to be taken to not change the ecosystem relationships to a degree that could cause a regime shift and as such more detailed analysis must be done to determine the best course of action to prevent this from happening.

CHAPTER 4: CONCLUSIONS

The relationships concluded within this study can assist in the development of forest management practices that help to convert the peatland ecosystem from a high intensity high-risk burn environment to one that is of lower risk and burning intensity. Through practices that promote the recovery of *S. fuscum* and other *Sphagnum* species it is possible to restore ecosystem services post-fire. In late succession peatlands, due to light availability, *Sphagnum* can no longer compete with feathermosses and loses its dominance. Feathermosses do not possess the same moisture transporting and retention properties as *Sphagnum*, thus increasing the vulnerability of the ecosystem through promotion of severe burning. Severe burning, as seen in Chapter 2 of the thesis, is subject to slow recovery and from a wildfire management perspective is difficult to control. Therefore, practices that promote *Sphagnum* growth at any successional stage are beneficial.

The study investigated the prevalence and persistence of hydrophobicity in a poor-fen peatland in order to gain a greater understanding of the limitations on recovery. To compliment this, half of the study involved the use of transplanted *Sphagnum* fragments and colonies to investigate the effects of hydrophobicity and vegetation recovery, using several different methods, on success of growth. Although hydrophobicity was not directly measured in the transplant study, due to the large spatial coverage and high significance of hydrophobic values associated with microform and peat type, it was possible to infer that the colonies transplanted within the severely burned hollows and lightly burned feathermoss lawns were exposed to hydrophobic surroundings. Therefore, the influence of hydrophobicity on transplant survival can be concluded based on the success or failures of each colony and fragment plot. The success of the larger transplants and differences between species, points to future research needs to determine suitable species under varying conditions. This would be useful in the reclamation of industrial sites and allow a forestry based management system that could potentially control the surface moisture characteristics in an attempt to mitigate severe burning fires.

Chapter 3 of this thesis and other work, indicate that the recovery of *Sphagnum* is facilitated by the natural regrowth of vascular shrubs within peatlands (Whinam and Buxton, 1997; Cobbaert et al., 2004; Hope et al., 2005; Farrick and Price, 2009; Koyama and Tsuyuzaki, 2010; Chong et al., 2012). Vascular vegetation recovery on severely burned hummocks and lightly burned feathermoss lawns suggests that they could be utilized for reclamation to compliment the active recovery in other microforms. Selection of transplant locations should focus on areas with moderate to high vascular density where small transplanted colonies can access moisture from the underlying peat, bypassing the hydrophobic zones present within them. The water limitations due to hydrophobicity of the surrounding peat would subject the colonies to conditions similar to that of primary peat formation where precipitation and runoff during precipitation events are the only sources of moisture.

The only areas showing significant *Sphagnum* moss recovery in the peatland are the lightly burned hummocks three years post-fire, representing roughly 12% of the surface composition. The restoration efforts using colonies were more successful than the fragments in the severely burned hollows with an average area increase of 1351 ± 239% the original transplant area at the end of the study compared to the fragments which experienced minimal growth. If this process were to be applied within a burned peatland shortly post-fire it would likely be possible to have ~47% (number exclusive to the peatland studied for area of severely burned hollows and lightly burned feathermoss lawns) of the peatland actively recovering *Sphagnum* surface cover. However, the microform distribution among burned peatlands may differ based on the microform distribution pre-fire which can be related to many factors including the hydrogeological setting (Hokanson, 2014) and successional stage (Benscoter et al., 2005; Fenton and Bergeron, 2006; Benscoter and Vitt, 2008). Management work investigating how to maintain a certain desirable successional stage to increase surface moisture availability and thus resilience to deep severe burning would complement reclamation work in areas that need recovery. These relationships would help to narrow down which peatlands would benefit most from reclamation efforts involving *Sphagnum* colony transplants. Reclamation work of this kind would be useful in restoring the ecosystem services of the peatland as efficiently and rapidly as possible. Future work investigating the effectiveness of transplants within severely burned *S. fuscum* hummocks may be pertinent as hydrological connection is present just below the surface and should yield promising results.

Another action that could possibly increase *Sphagnum* regeneration, considering the severely burned hummocks possess their highest hydrophobicity just below the surface, at a depth of 2 cm, with an average WDPT of 33 ± 11 sec, would be to trim the top layer thus allowing spores within the peat to recover more efficiently from within the hummock and create more habitable surface area for other mosses. This would need to be done shortly post-fire as spore viability decreases with time since they were deposited within the peat column (Sundberg and Rydin, 2000). The peatland should be able to recover more rapidly if surface conditions are made suitable to spore germination. Since the capitula is not present removal of the fire damaged stems near surface should not have a significant impact on the evapotranspiration properties of the microform, yet would give the spores access to light and the ability to recover in more dense peat with higher moisture availability. As such peatland microform distribution would determine the best course of action regarding the location of transplant placements. The study has shown that in less than three years lightly burned hummocks and severely burned hollows can actively recover and begin progression towards *Sphagnum* dominated cover.

Increased hydrophobicity in the severely burned hummocks and lightly burned feathermoss lawns could increase resiliency within the ecosystem by limiting germination of invading species as it does with *Sphagnum*. This would allow the *Sphagnum* to establish dominance quickly in areas conducive to its own growth and once those areas are colonized the *Sphagnum* could then spread laterally over the inhospitable peat surfaces, gradually increasing moisture content above the surfaces while accumulating more peat. However, as shown in the literature, natural regeneration can take anywhere from 20-80 years (Benscoter and Vitt, 2008), therefore growth should benefit from the reclamation efforts shown in this paper.

The reclamation strategies could be implemented in a burned peatland to mitigate future fire risk by restoring *Sphagnum* moss surface cover, which has been shown to reduce the severity of burning and increase the resiliency of the system (Benscoter and Weider, 2003; Shetler et al., 2008). As *Sphagnum* cover has been shown to reduce the severity and depth of burn in peatland ecosystems, in regions with high peatland land cover it would be pertinent for wildfire managers to implement practices to increase or preserve *Sphagnum* cover in peatlands. Through management of surface moisture characteristics severe fires may be damped by the presence of peatlands dominated by high moisture retaining mosses such as species in the *Sphagnum* genus.

The processes regarding colony size, species, and location dependencies discussed in this thesis can be transferred to reclamation peatlands and for wildfire recovery where the ecosystem would benefit from the re-introduction of moss species. The surfaces of industrial reclamation sites may not be conducive to spore germination even though the near surface moisture regime and nutrient concentrations could be suitable to *Sphagnum* growth. This barrier could easily be surpassed through the use of transplanted *Sphagnum* colonies similar to those used in this study. The reclamation consultant could make an informed decision based on site characteristics such as water table depth and surface conditions to decide which species and methods to implement. It is therefore suggested that the knowledge gained in this study be used by consultants and wildfire managers to better predict *Sphagnum* growth in natural and remediated peatlands and implement reclamation procedures where necessary.

FIGURES

Figure 1.1: Project area with the extent of the hydrophobicity study outlined in white; block one of the transplant study represented by red dots; block two orange; block three light orange; block four yellow; block five light green; block six dark green; and the microhabitat plots designated by blue.

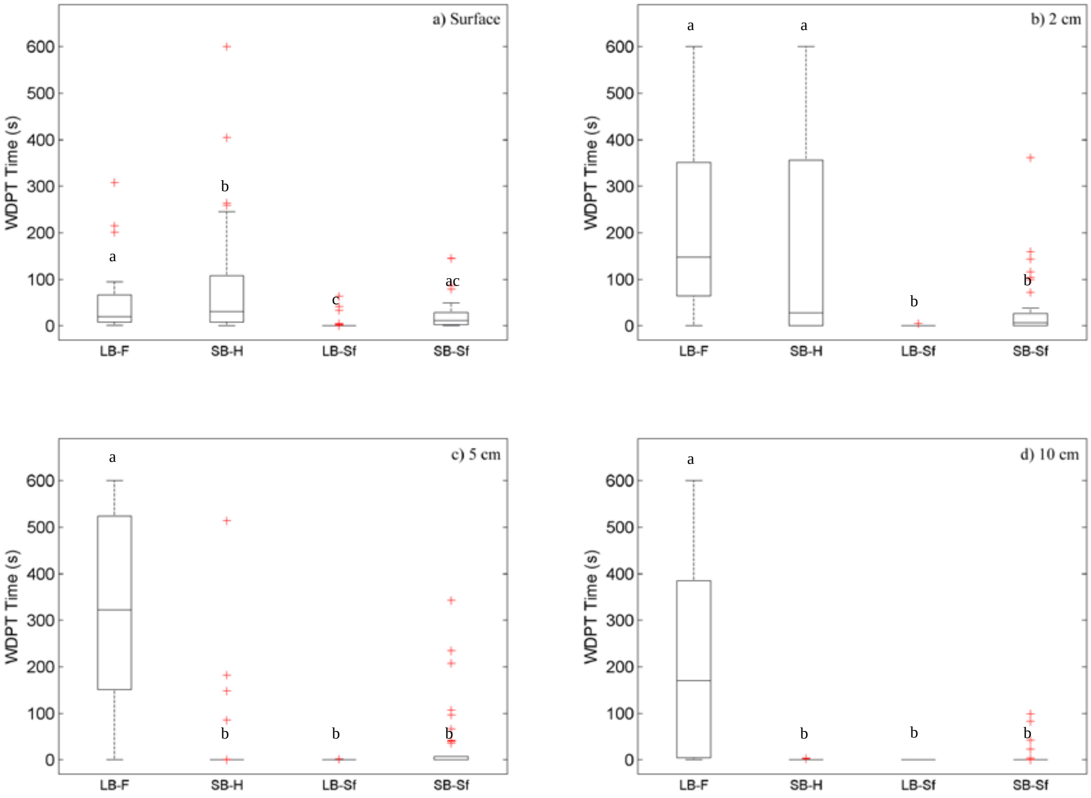

Figure 2.1: Field WDPT times for the different microform types and depths.

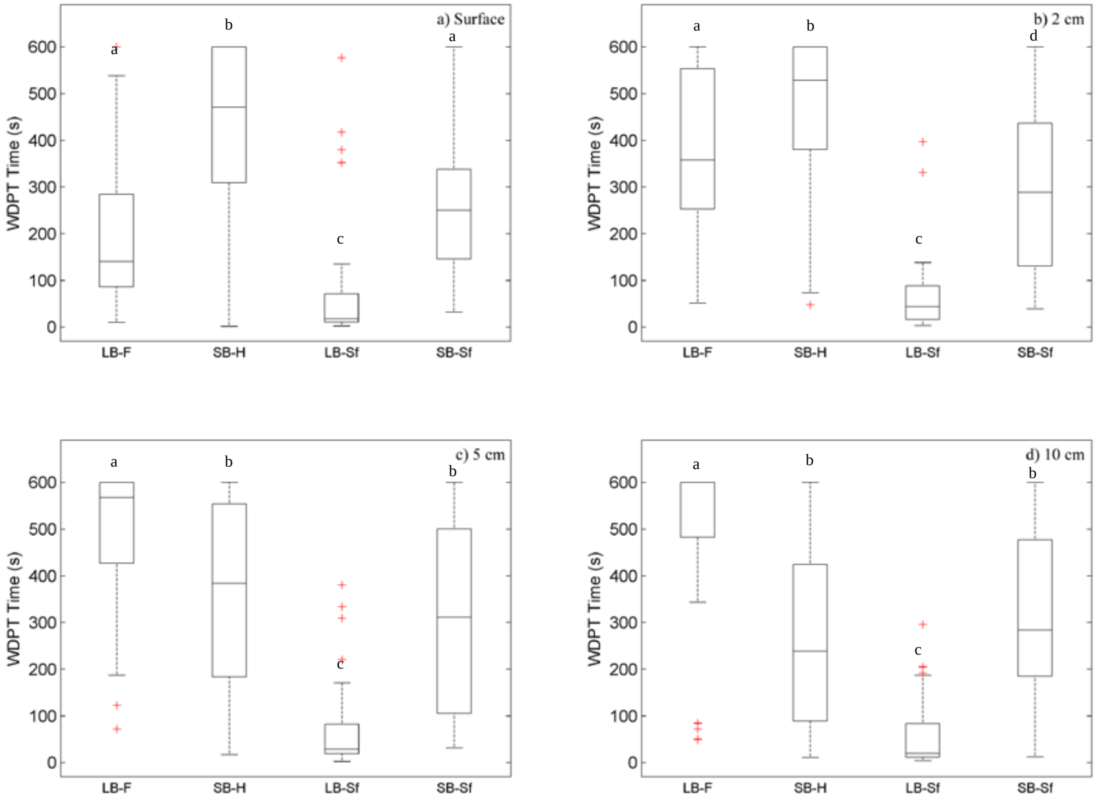

Figure 2.2: Lab WDPT times repeated on the same samples as field results. With all samples dried to control for moisture differences.

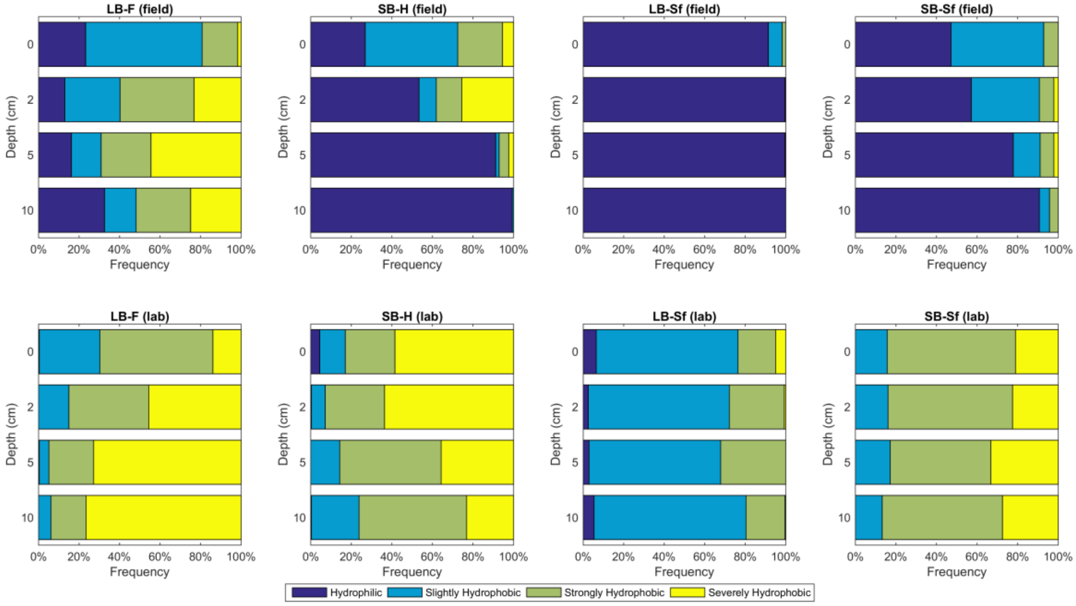

Figure 2.3: Relative frequency of water drop penetration times for all samples (n = 280 at each depth in each subplot, N total = 8960).

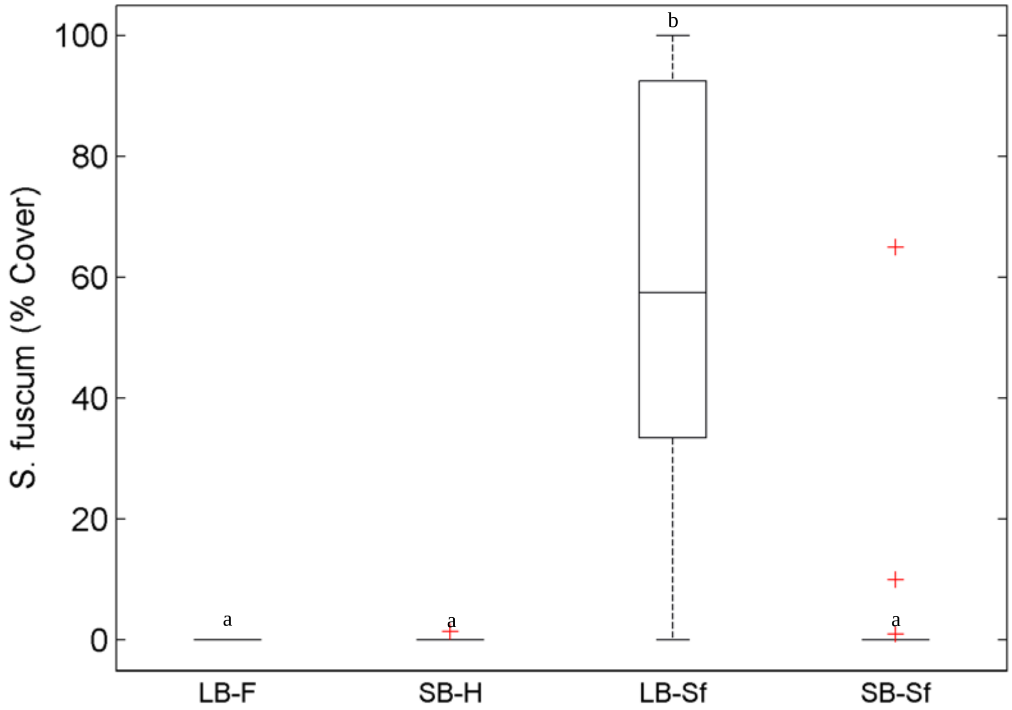

Figure 2.4: Percent surface cover of *S. fuscum* based on visual estimation by microform type.

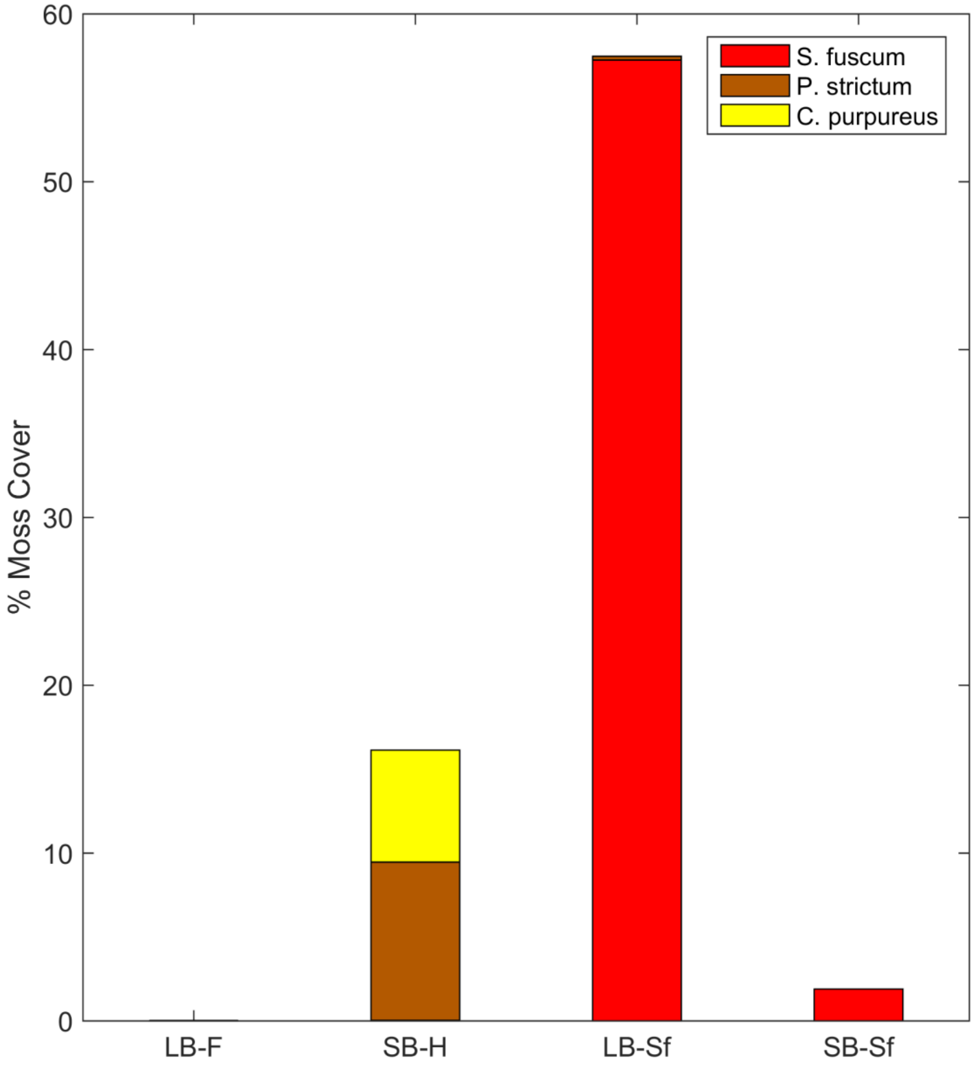

Figure 2.5: Surface cover percentage by species and microform type.

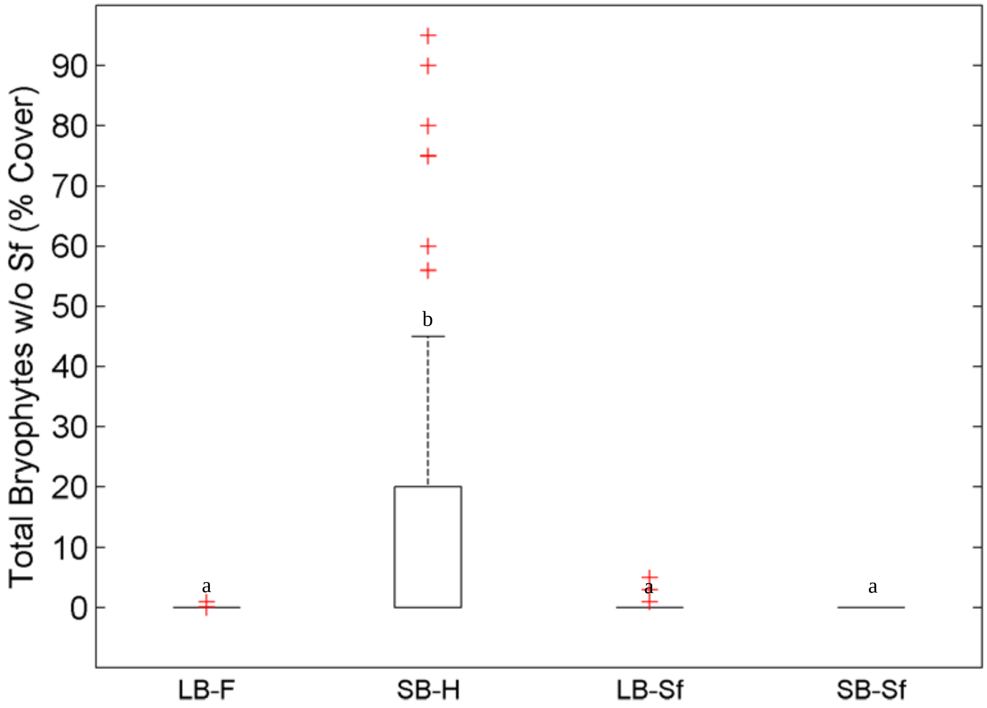

Figure 2.6: Percent surface cover of total bryophytes based on visual estimation by microform type. Total bryophytes includes all species except *S. fuscum.*

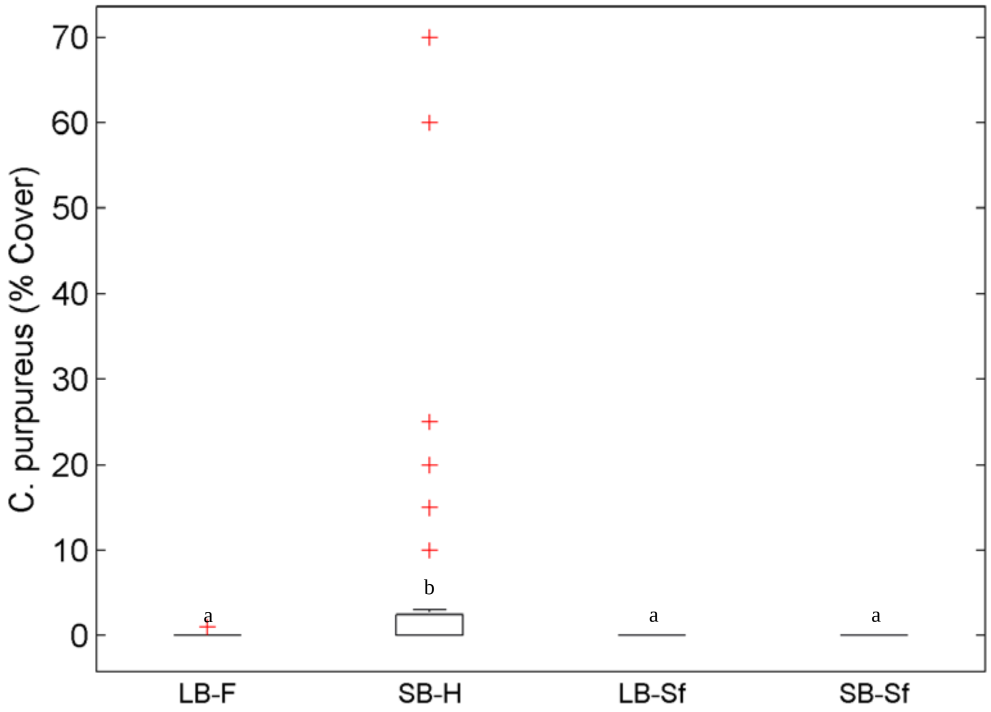

Figure 2.7: Percent surface cover of *C. purpureus* (firemoss) based on visual estimation by microform type.

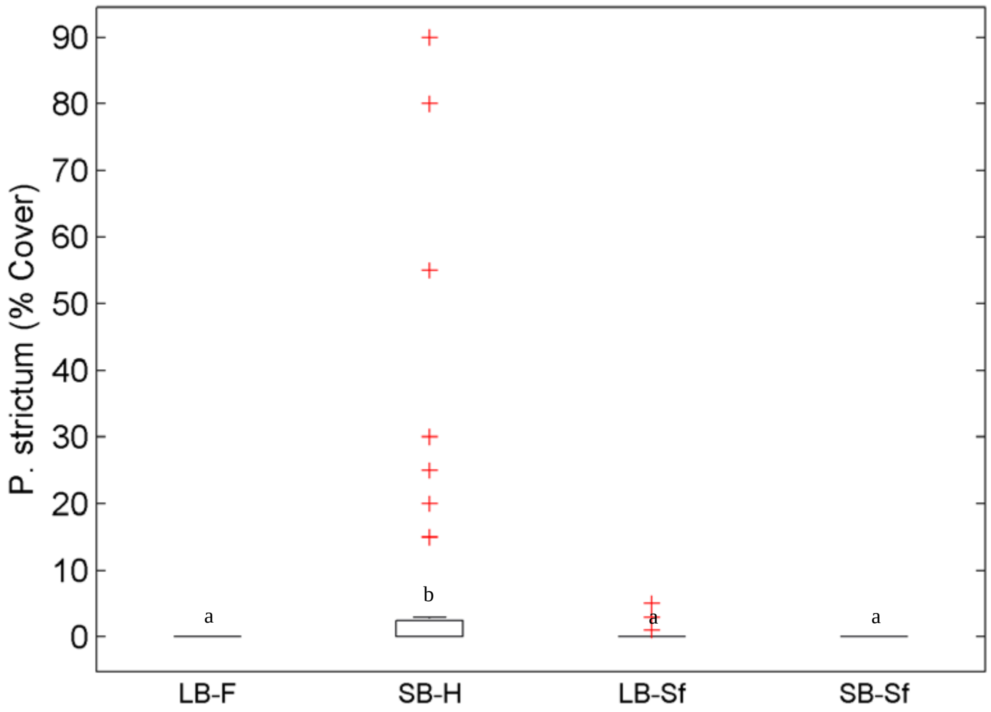

Figure 2.8: Percent surface cover of *P. strictum* based on visual estimation by microform type.

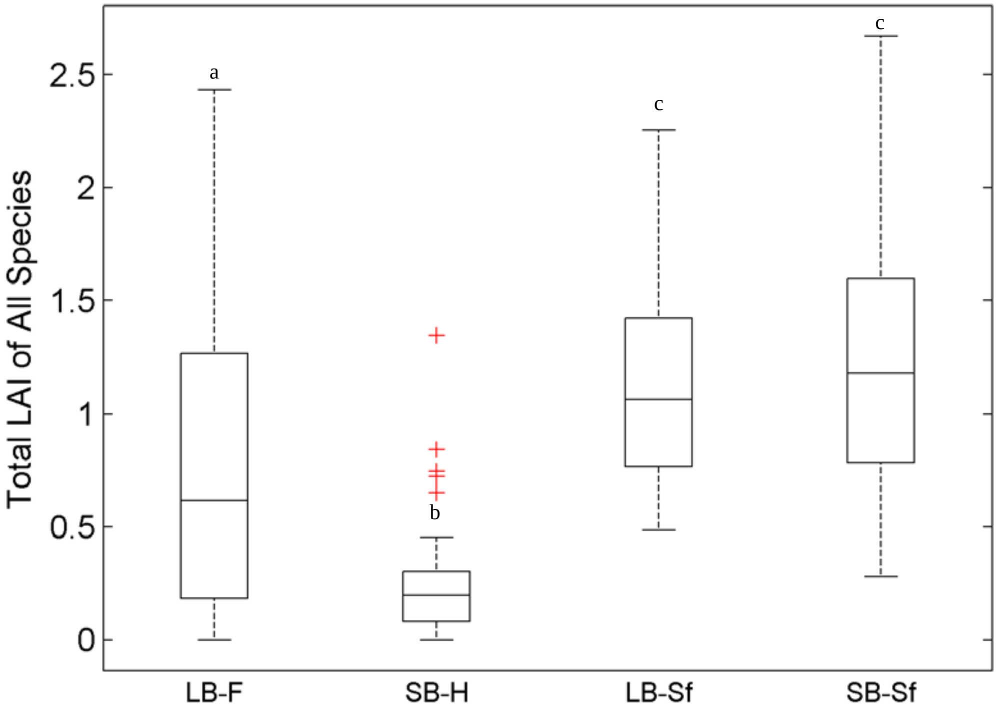

Figure 2.9: Total Leaf Area Index (LAI) based on destructive sampling of all above ground biomass within the plots by microform type.

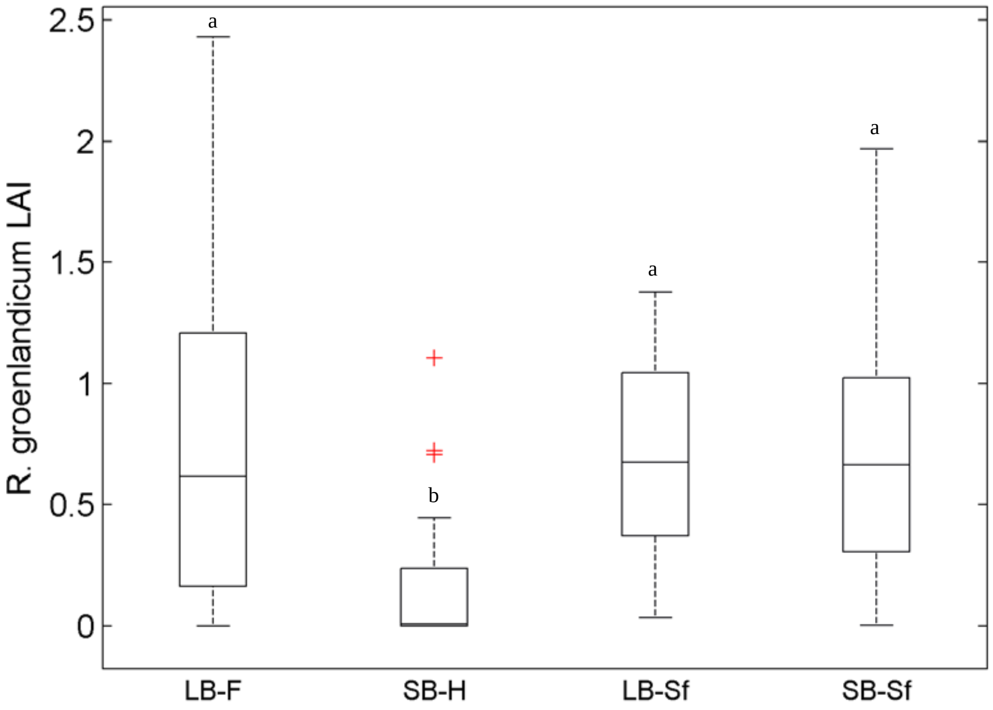

Figure 2.10: Total LAI of *R. groenlandicum* based on destructive sampling of above ground biomass within the plot by microform type.

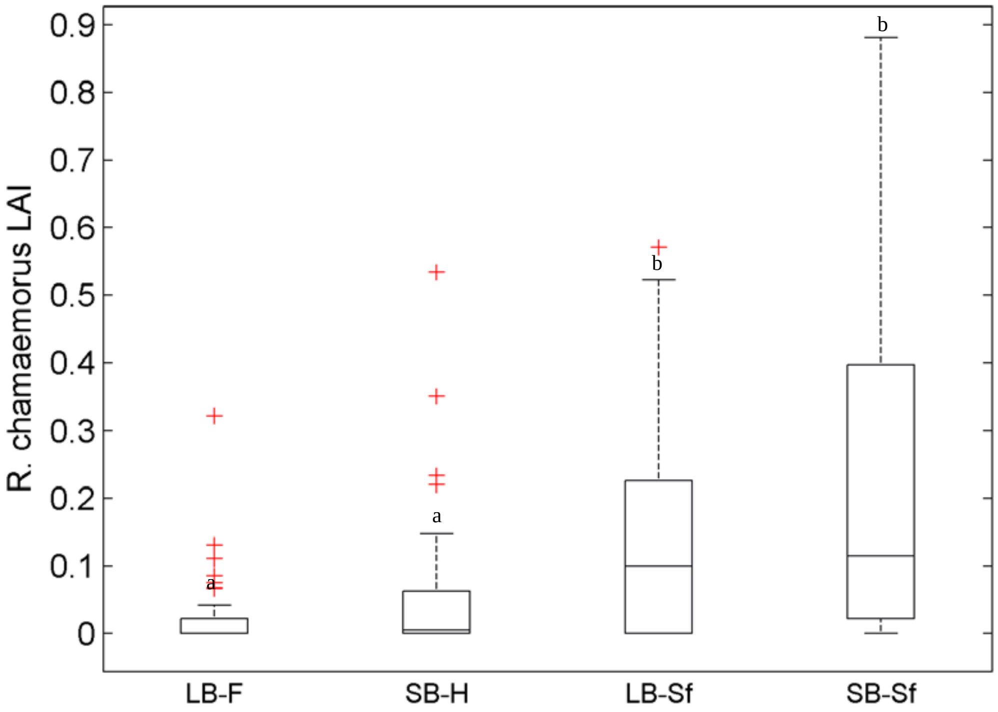

Figure 2.11: Total LAI of *R. chamaemorus* based on destructive sampling of above ground biomass within the plot by microform type.

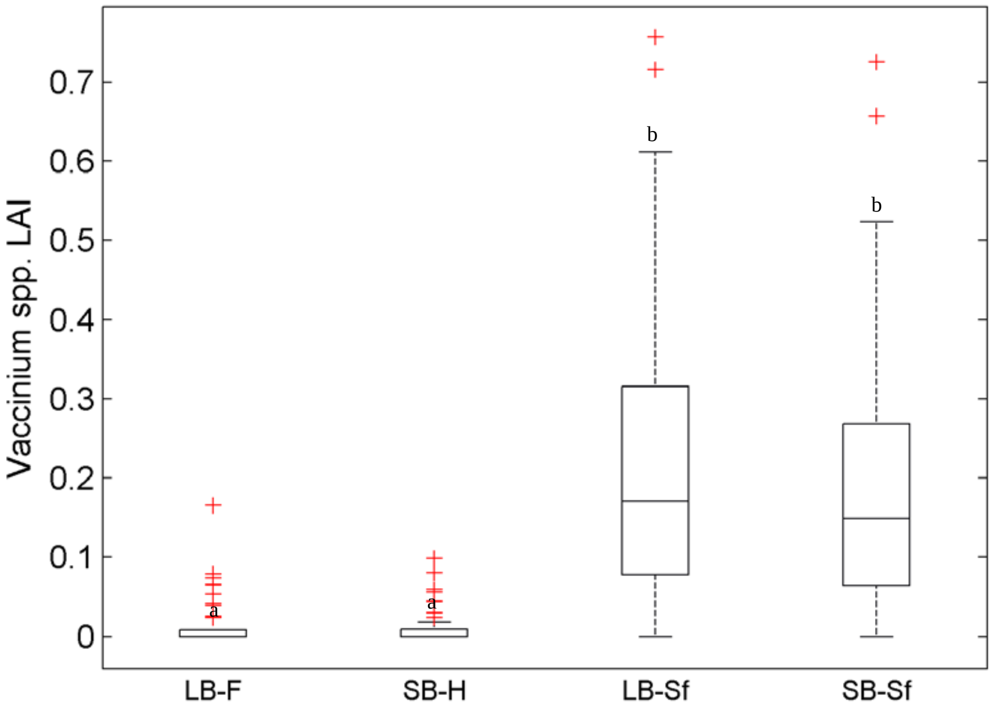

Figure 2.12: Combined Leaf Area Index (LAI) of *V. oxycoccos* (small bog cranberry) and *V. vitis-idaea* (bog cranberry) based on destructive sampling of above ground biomass within the plot by microform type.

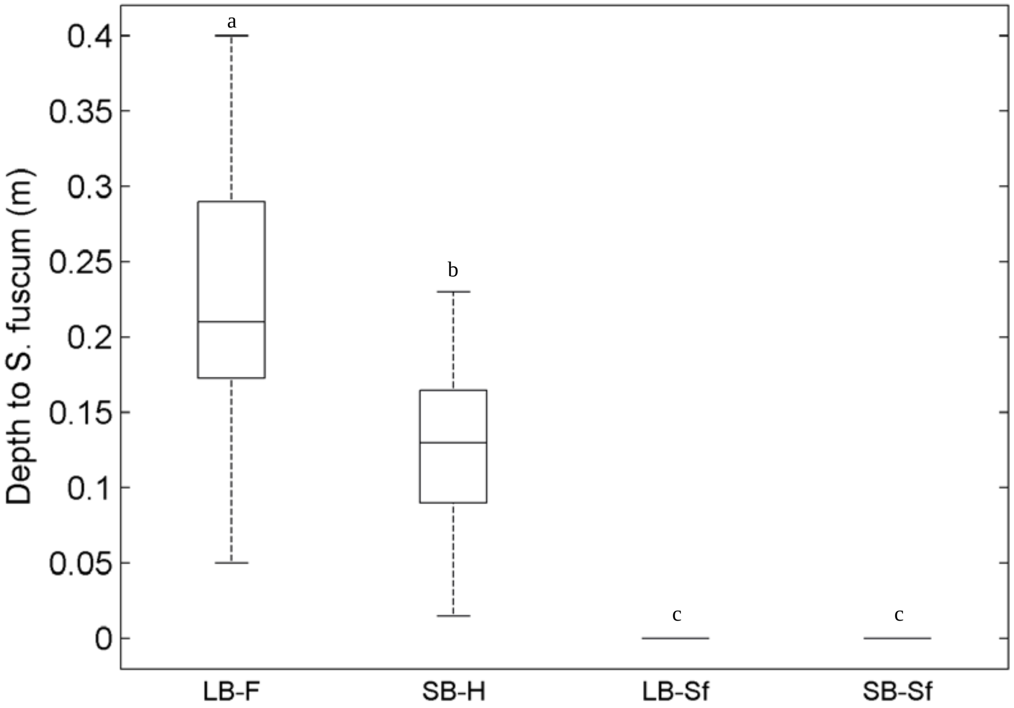

Figure 2.13: Depth below surface to *S. fuscum* botanical origin peat by microform type.

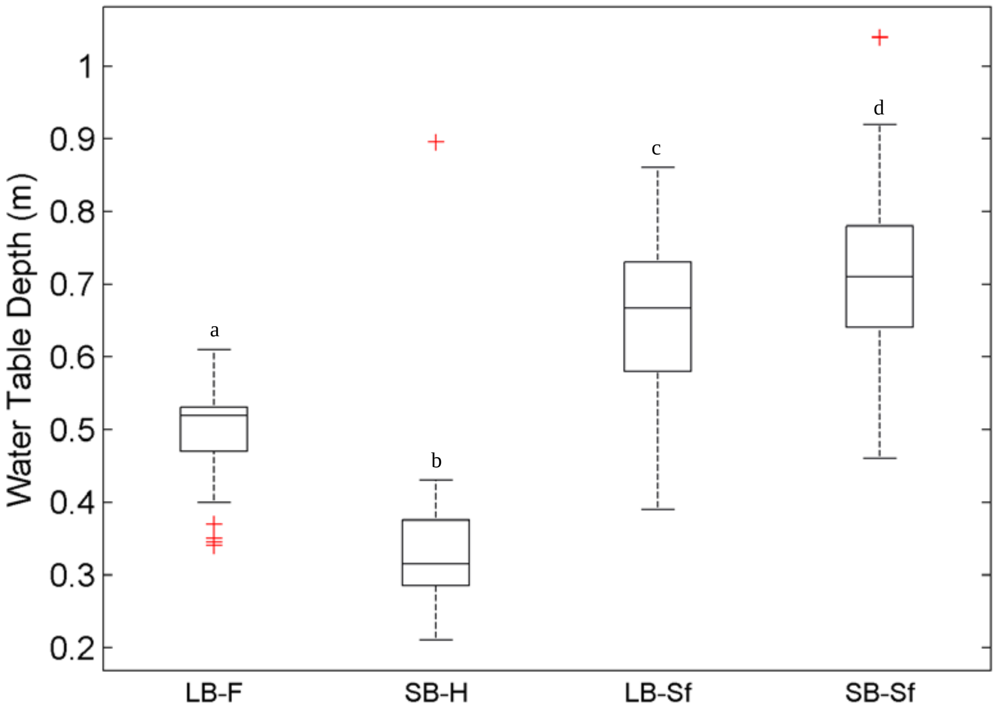

Figure 2.14: Average water table depth by microform type sampled.

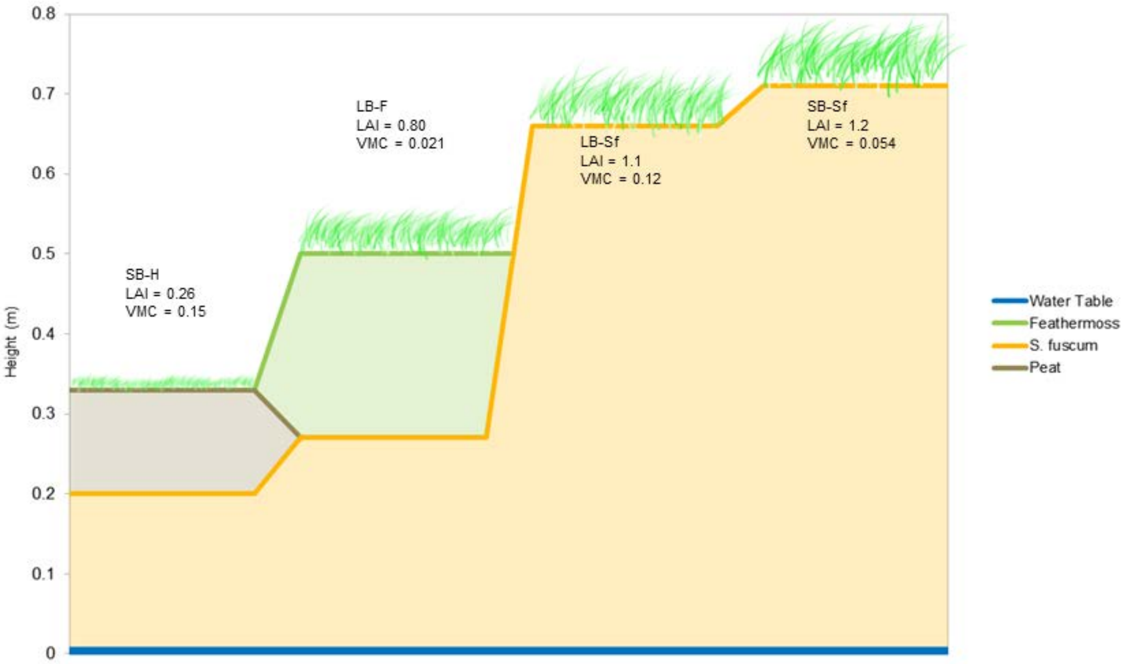

Figure 2.15: Conceptual diagram comparing the moisture influencing differences between microform types.

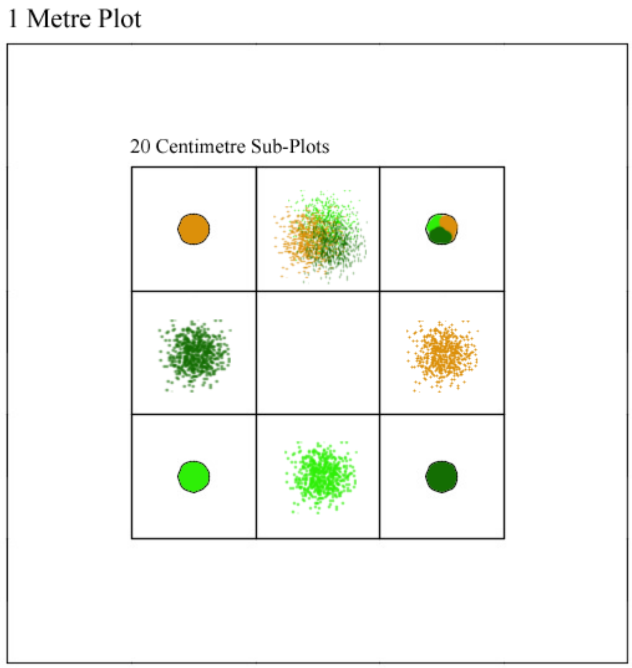

Figure 3.1: Diagram of transplanted colony and fragment plots. The 20 cm buffer area with same treatment as subplots can be seen around the 3 x 3 grid. The colonies can be seen in the corners of the 3 x 3 subplot grid with the fragments located between them. The central subplot is the control area to determine success of fragments over natural regeneration.

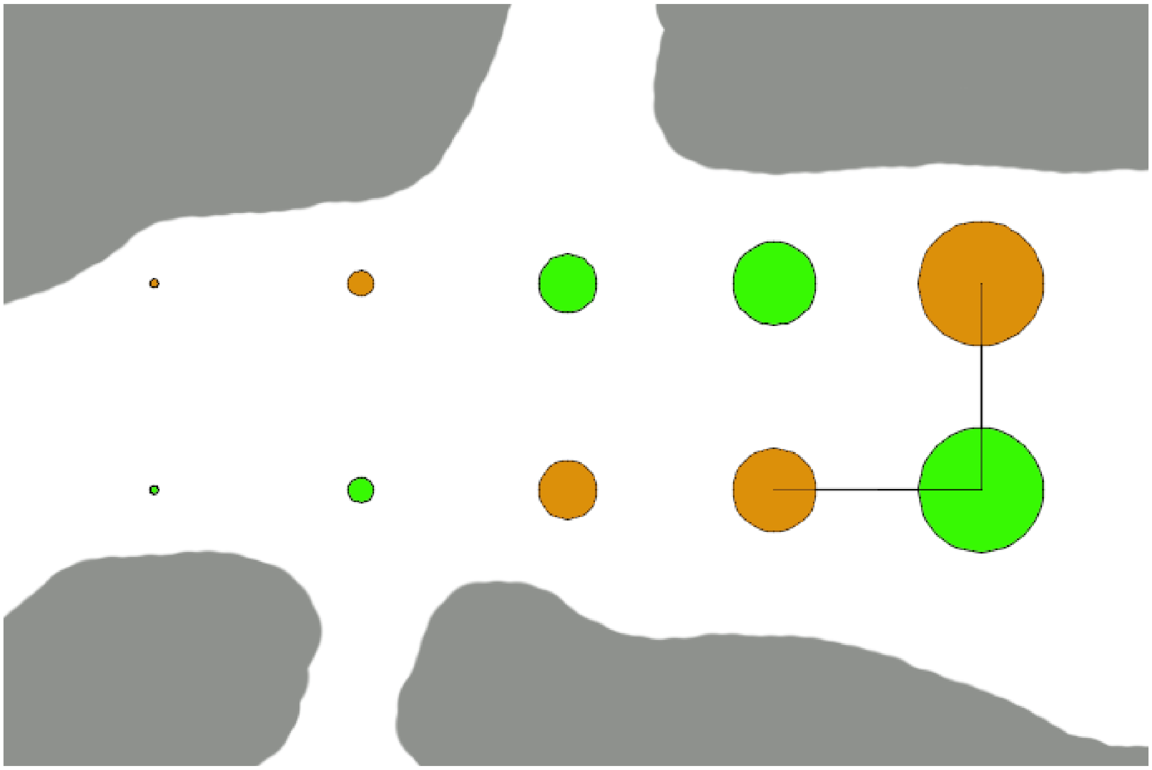

Figure 3.2: Microhabitat colony plot with the preferred arrangement of colonies in two parallel linear lines of increasing size. Species type along the size increase was random and this method only chosen if surface slope was negligible.

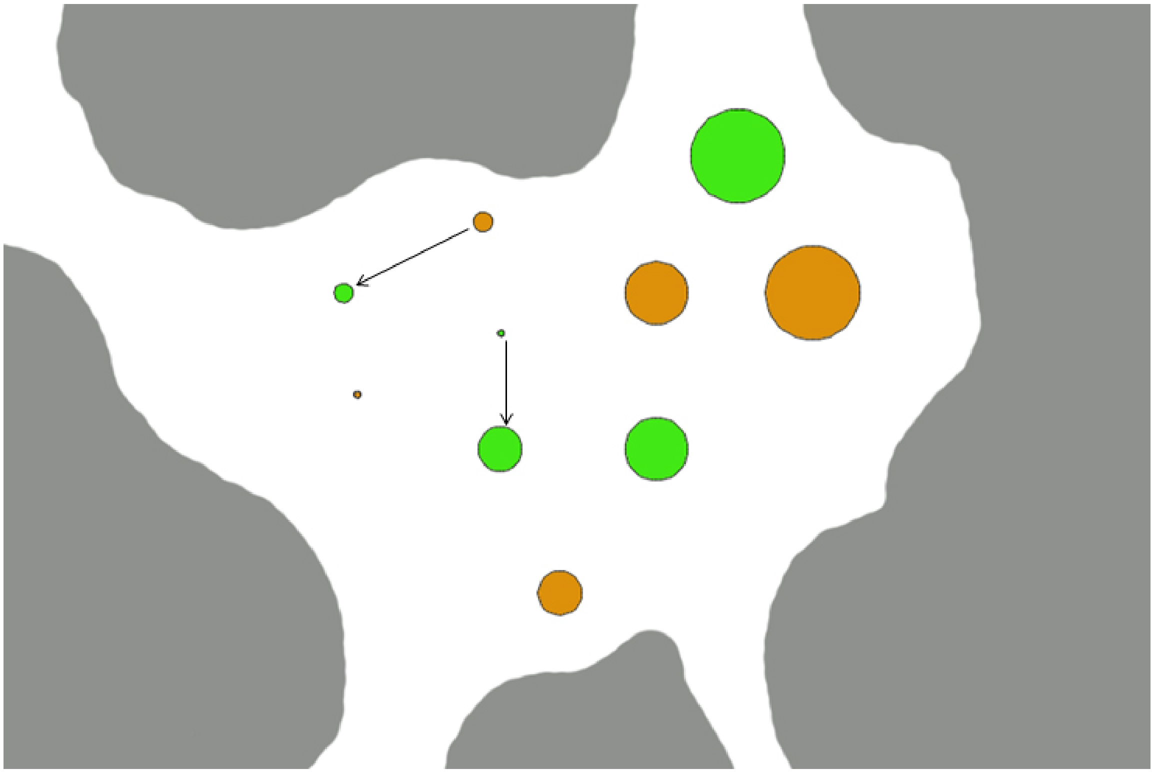

Figure 3.3: Microhabitat transplant plot arrangement if the colonies could not be transplanted in two linear parallel lines, while still abiding by the 20 cm limit between species type and size (shown with the black arrows).

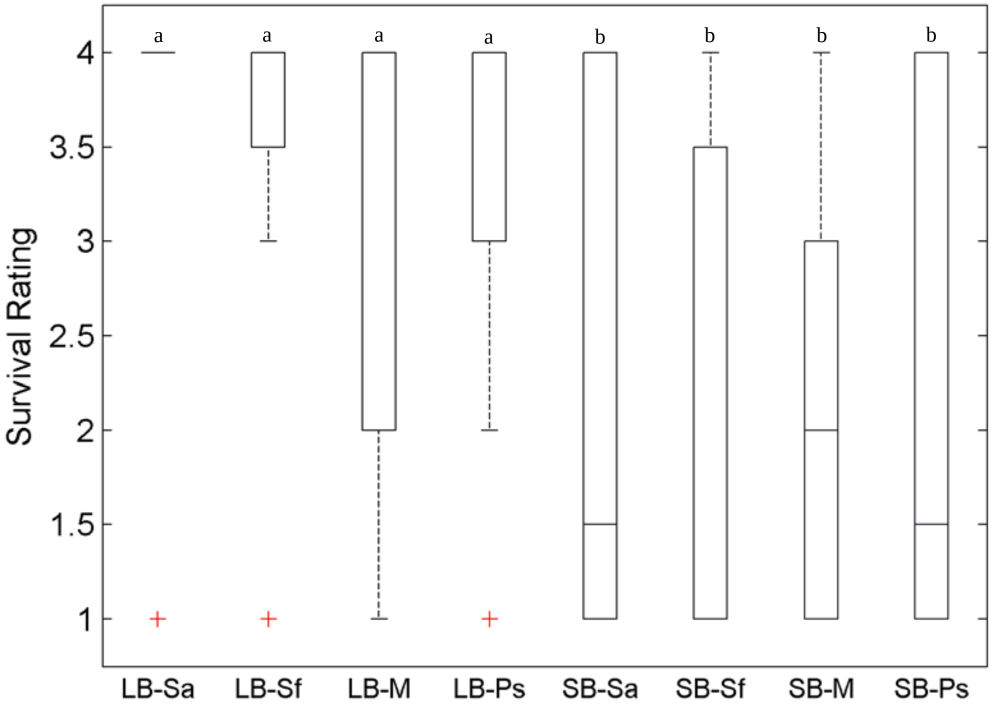

Figure 3.4: Transplanted colony survival rating average from the 1-4 scale at the end of summer, 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium,* Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

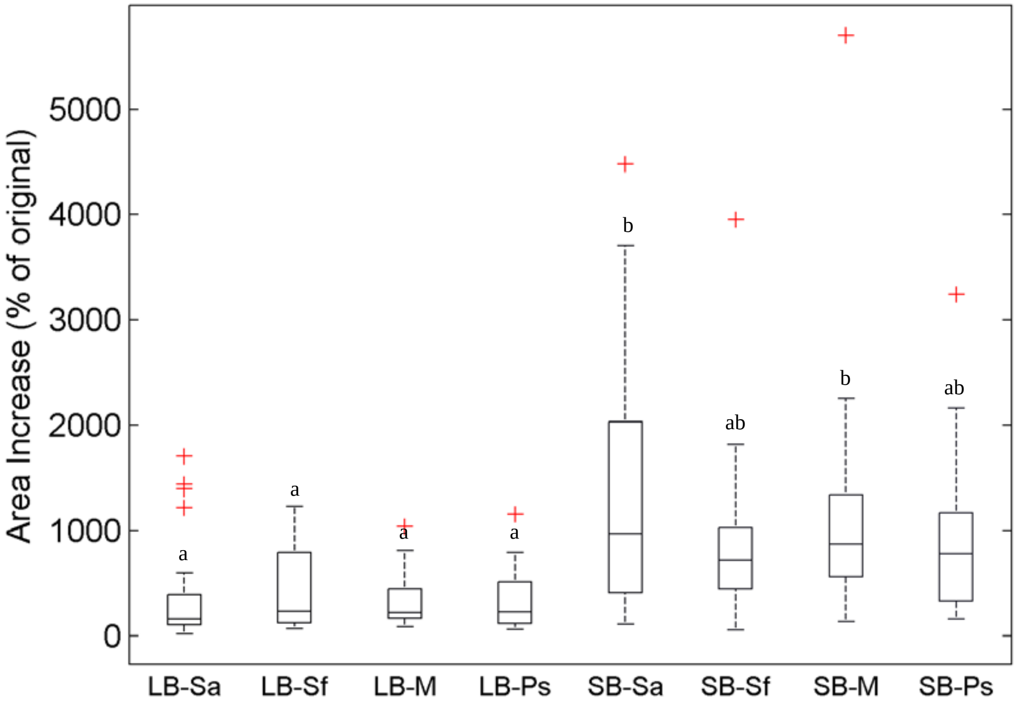

Figure 3.5: Colony surface area increase using the installation diameter of 5 cm and final diameters measured in August of 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

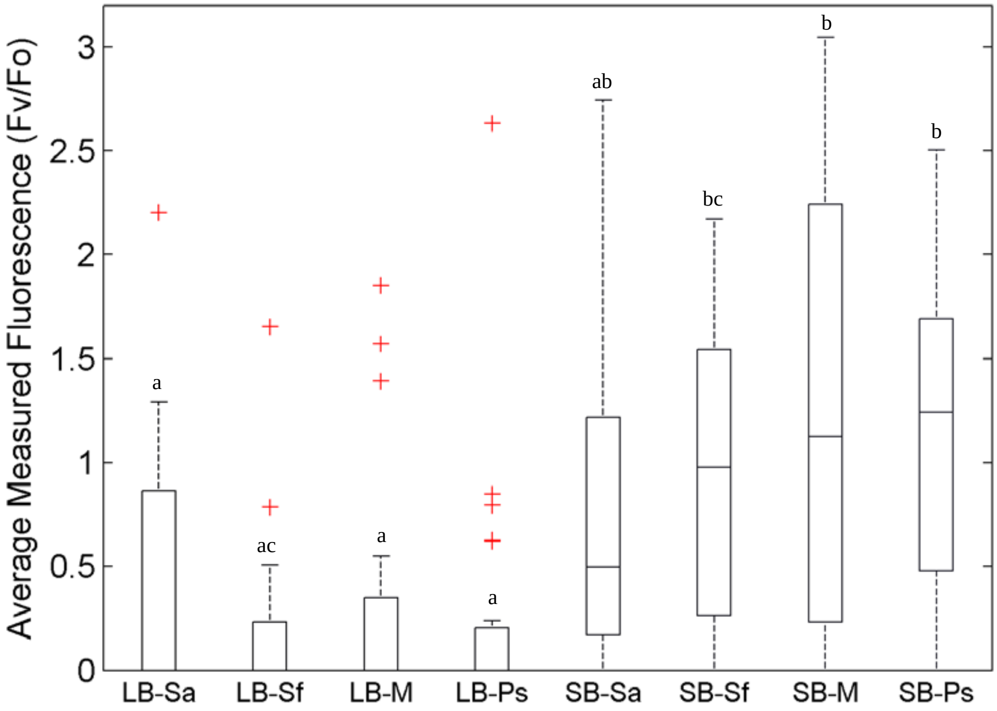

Figure 3.6: Colony average chlorophyll fluorescence during summer of 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

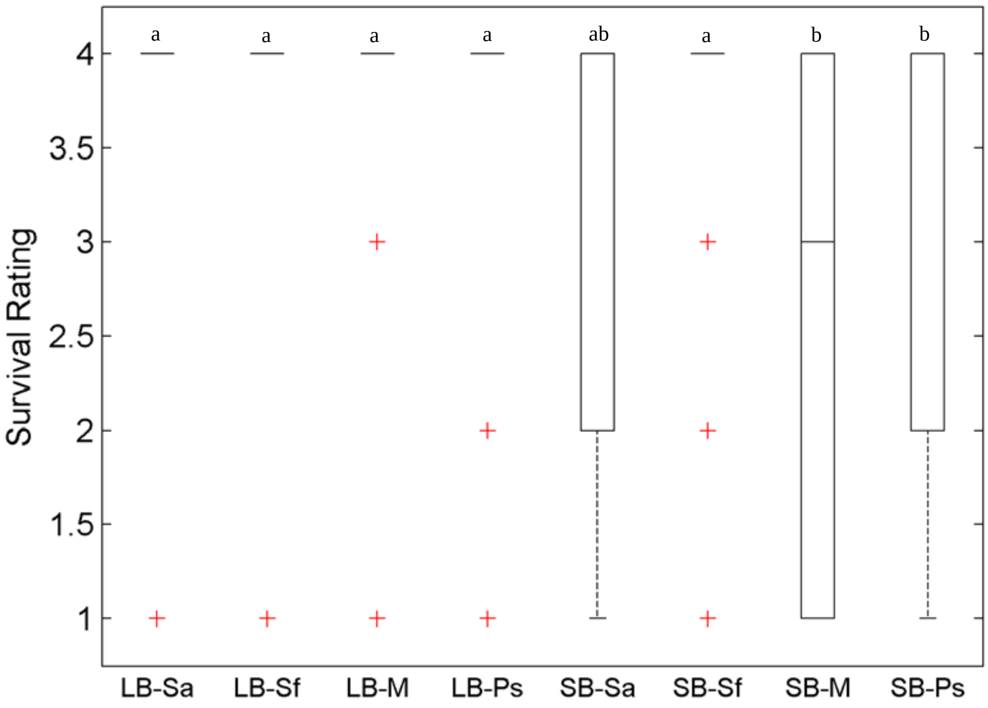

Figure 3.7: Fragment survival rating average from the 1-4 scale at the end of summer, 2014. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

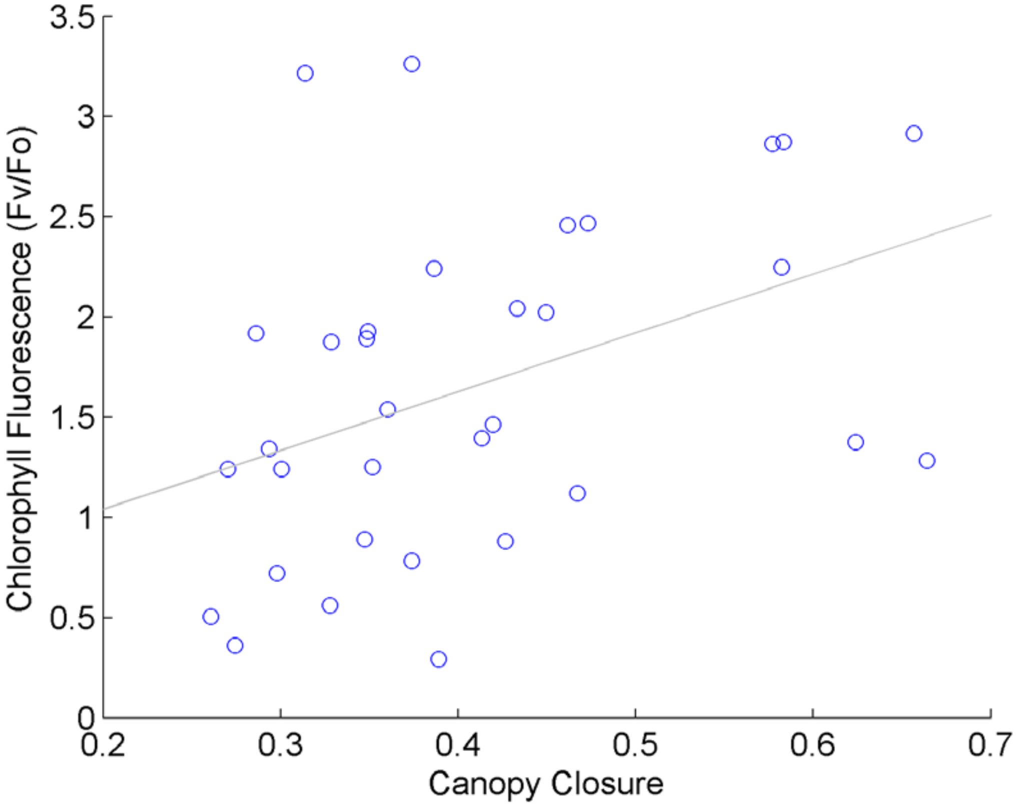

Figure 3.8: The relationship between canopy closure and chlorophyll fluorescence of all active colonies on August 19th, 2014 in lightly burned feathermoss lawns. The trend is statistically significant with $p<0.001$.

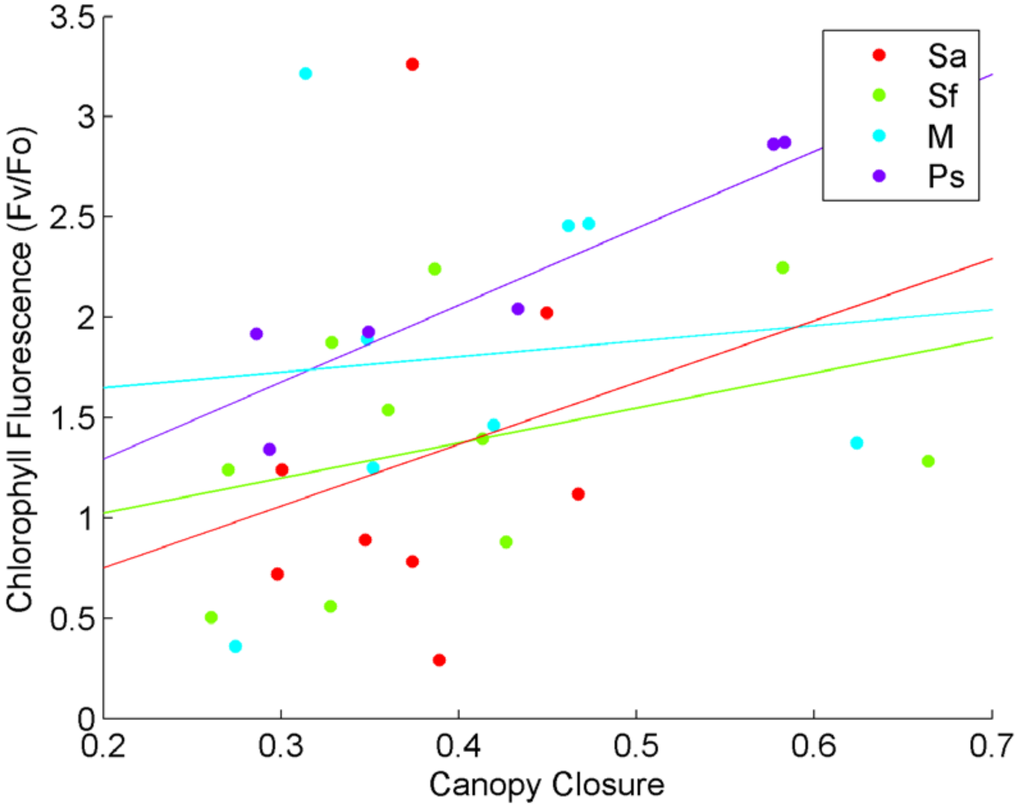

Figure 3.9: The relationship between canopy closure and chlorophyll fluorescence of all active colonies by type in lightly burned feathermoss lawns on August 19th, 2014. *Polytrichum strictum* is statistically significant ($p<0.001$) however the other three species are not. Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

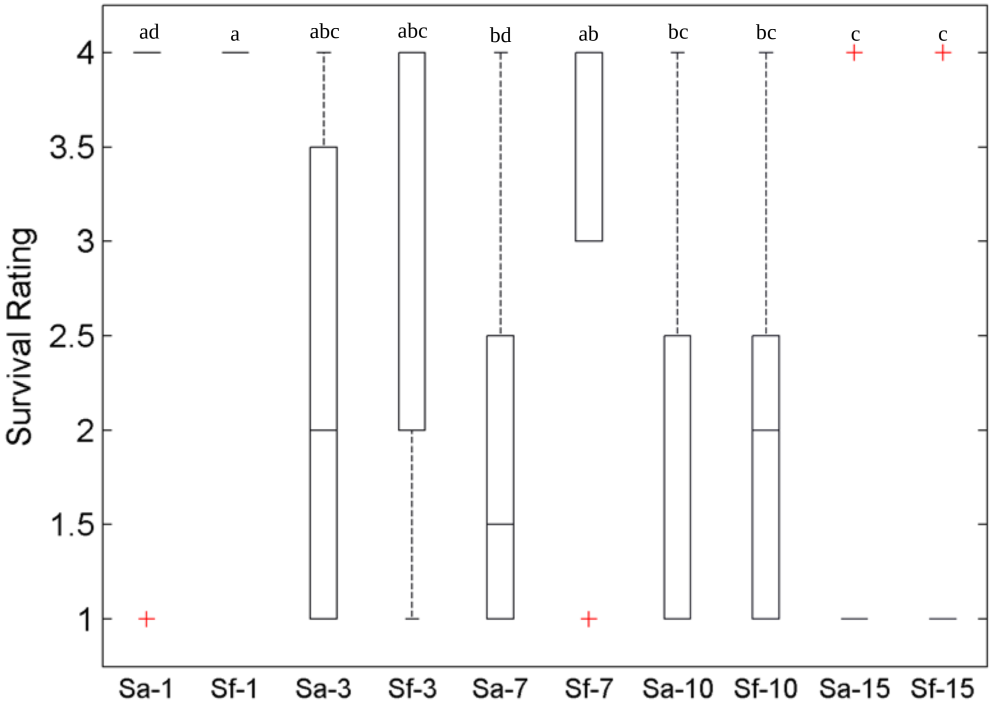

Figure 3.10: Survival rating of the transplanted microhabitat colonies by species and size from the 1-4 scale at the end of summer, 2014. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the colony diameter at installation.

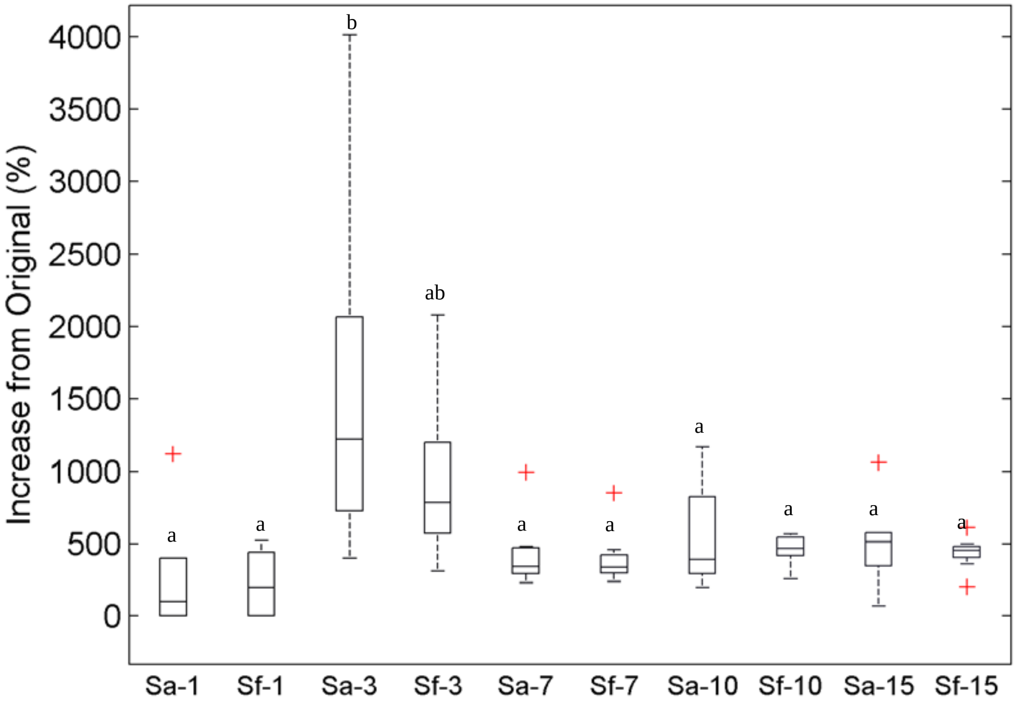

Figure 3.11: The percent increase from the original size until end of summer 2014. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation.

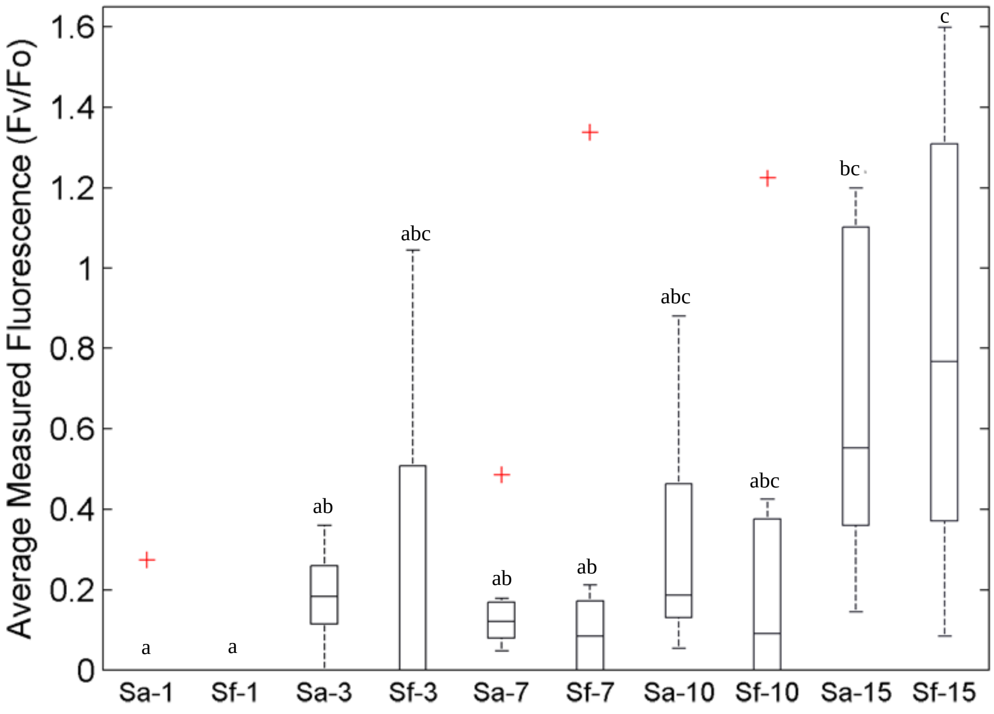

Figure 3.12: Measurement averaged chlorophyll fluorescence for the summer of 2014. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation.

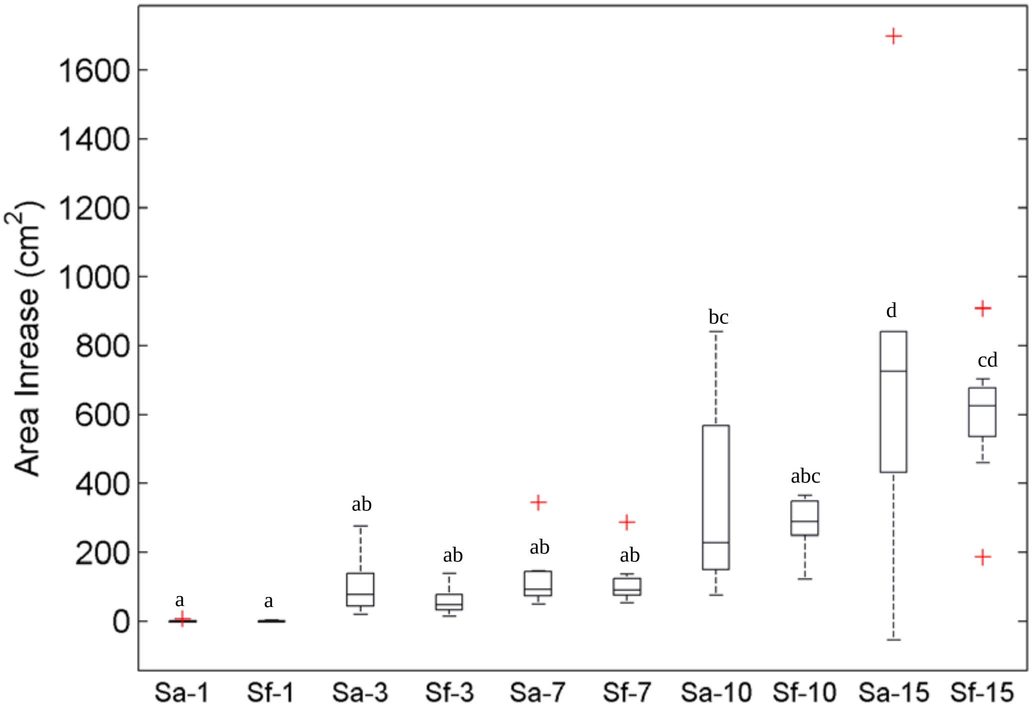

Figure 3.13: Absolute area increase from original size of the microhabitat colonies. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation.

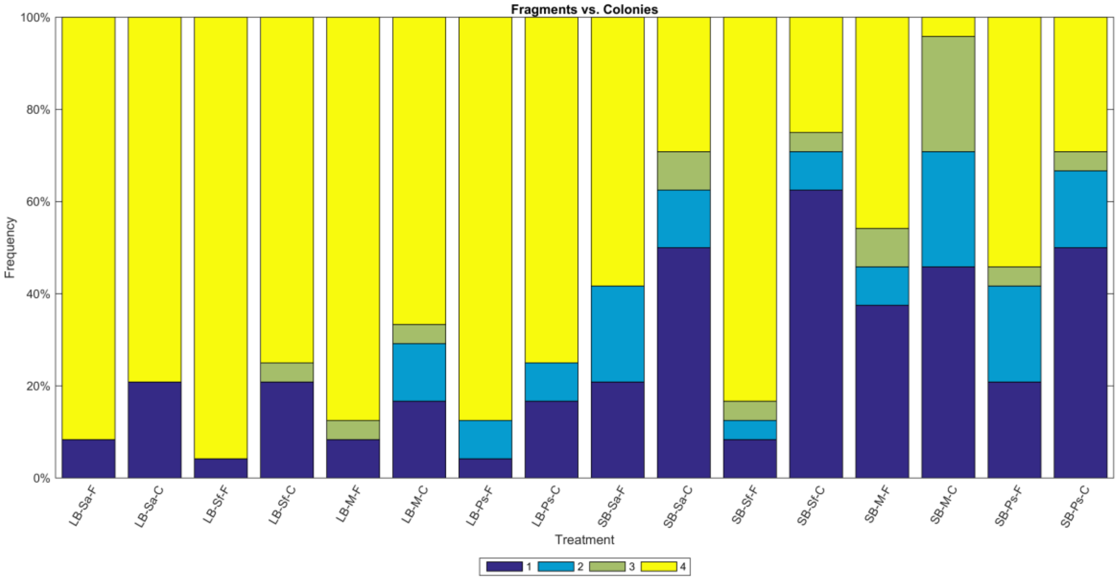

Figure 3.14: Frequency of survival rating of the applied fragments (F) and colonies (C). LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.* (n = 24 per column).

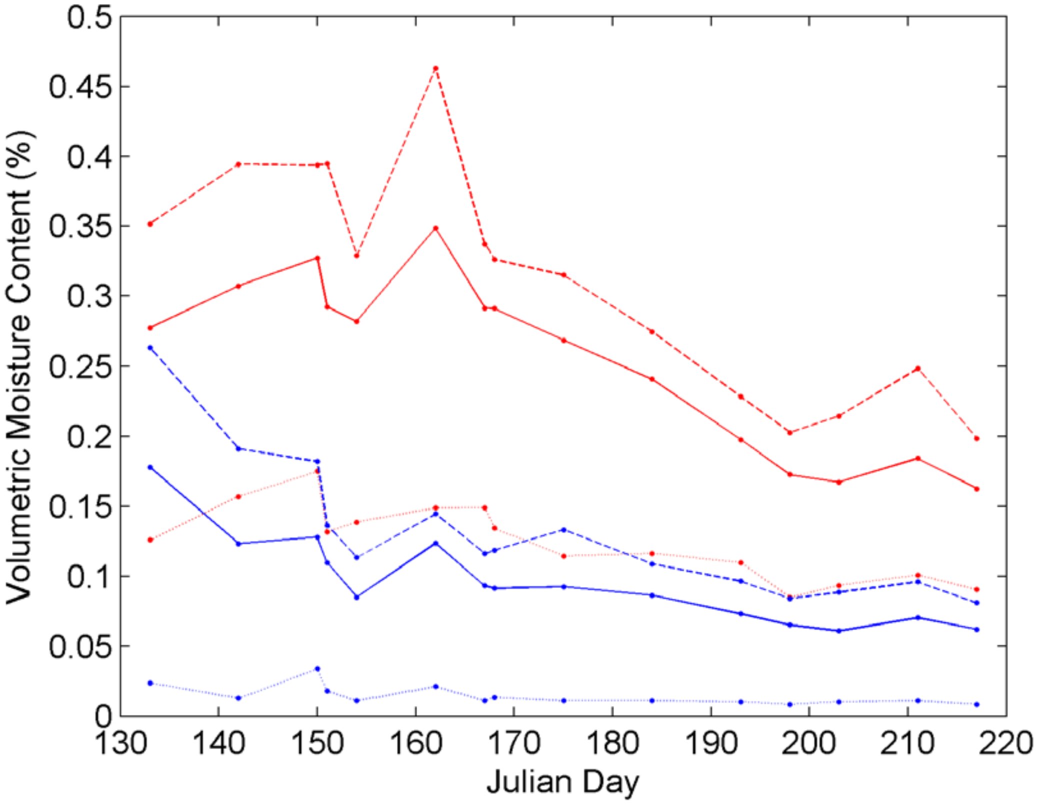

Figure 3.15: Summer of 2014 moisture trend for the lightly burned (blue) and severely burned (red) microforms that the colonies were transplanted into. The solid line represents the mean, dotted line representing the first quartile, and dashed line representing the third quartile limits.

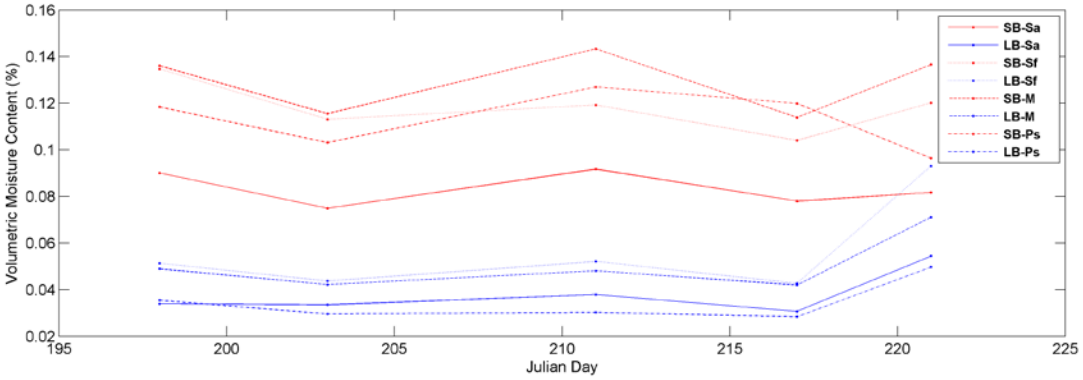

Figure 3.16: Summer of 2014 moisture trend within the transplanted colonies. LB = Light Burn, SB = Severe Burn, Sa = *Sphagnum angustifolium*, Sf = *Sphagnum fuscum*, M = mixed species, Ps = *Polytrichum strictum.*

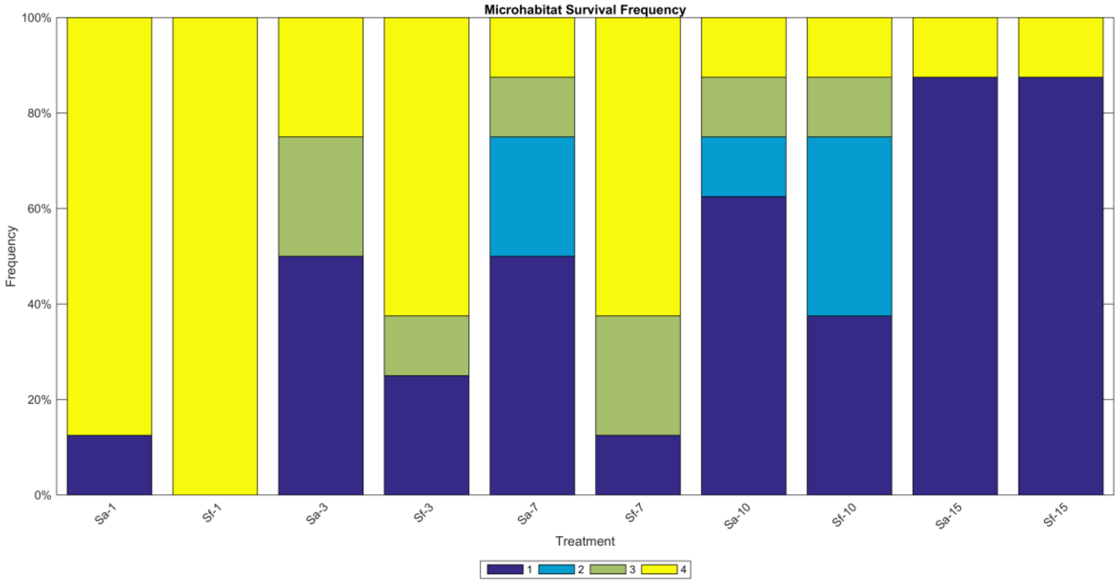

Figure 3.17: Frequency of survival rating based on the diameter of the colony. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation. (n = 8 for each column).

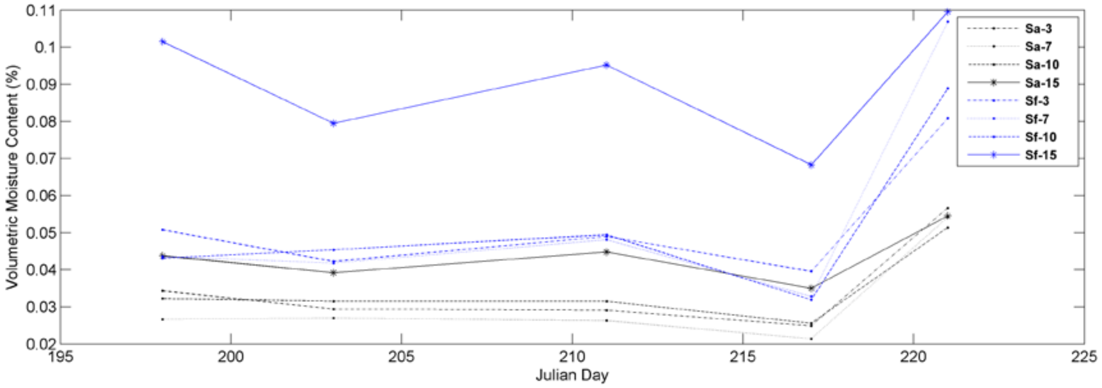

Figure 3.18: Summer of 2014 moisture trend of the microhabitat colonies by species and size. The 1cm diameter was omitted as the moisture probe prongs would have been in the surface peat and not the colony. Sa = *Sphagnum angustifolium* and Sf = *Sphagnum fuscum*. Numbers after the species denote the diameter at installation.

REFERENCES

ADC BioScientific Ltd. (2013). OS-30p+ - Rapid plant stress screening device. Retrieved April 2, 2015, from http://www.adc.co.uk/products/os30p-plant-stress-meter/.

Agren, J. (1989). Seed size and number in *Rubus chamaemorus*: between-habitat variation, and effects of defoliation and supplemental pollination. Journal of Ecology, 77(4), 1080-1092.

Aickin, M., and Gensler, H. (1996). Adjusting for multiple testing when reporting research results: the Bonferroni vs Holm methods. American Journal of Public Health, 86(5), 726-728.

Andrus, R. E. (1986). Some aspects of *Sphagnum* ecology. Canadian Journal of Botany, 64(2), 416-426.

Baker, N. R., and Rosenqvist, E. (2004). Applications of chlorophyll fluorescence can improve crop production strategies: an examination of future possibilities. Journal of Experimental Botany, 55(403), 1607-1621.

Benscoter, B. W. (2006). Post-fire bryophyte establishment in a continental bog. Journal of Vegetation Science, 17, 647-652.

Benscoter, B. W., and Vitt, D. H. (2008). Spatial patterns and temporal trajectories of the bog ground layer along a post-fire chronosequence. Ecosystems, 11(7), 1054-1064.

Benscoter, B. W., and Wieder, R. K. (2003). Variability in organic matter lost by combustion in a boreal bog during the 2001 Chisholm fire. Canadian Journal of Forest Research, 33(12), 2509-2513.

Benscoter, B. W., Vitt, D. H., and Wieder, R. K. (2005). Association of postfire peat accumulation and microtopography in boreal bogs. Canadian Journal of Forest Research, 35(9), 2188-2193.

Benscoter, B. W., Wieder, R., and Vitt, D. H. (2005). Linking microtopography with post-fire succession in bogs. Journal of Vegetation Science, 16(4), 453-460.

Berendse, F., Van Breemen, N., Rydin, H., Buttler, A., Heijmans, M., Hoosbeek, M. R., Lee, J. A., Mitchell, E., Saarinen, T., Vasander, H., and Wallen, B. (2001). Raised atmospheric CO_2 levels and increased N deposition cause shifts in plant species composition and production in *Sphagnum* bogs. Global Change Biology, 7(5), 591-598.

Berglund, K., and Persson, L. (1996). Water repellence of cultivated organic soils. Acta Agriculturae Scandinavica B-Plant Soil Sciences, 46(3), 145-152.

Bisbee, K. E., Gower, S. T., Norman, J. M., and Nordheim, E. V. (2001). Environmental controls on ground cover species composition and productivity in a boreal black spruce forest. Oecologia, 129(2), 261-270.

Campbell, D. R., Rochefort, L., and Lavoie, C. (2003). Determining the immigration potential of plants colonizing disturbed environments: the case of milled peatlands in Quebec. Journal of Applied Ecology, 40(1), 78-91.

Campeau, S. and Rochefort, L. (1996). Sphagnum regeneration on bare peat surfaces: field and greenhouse results. Journal of Applied Ecology, 33, 599-608.

Certini, G. (2005). Effects of fire on properties of forest soils: a review. Oecologia, 143(1), 1-10.

Chen, J. M., Blanket, P. D., Black, T. A., Guilbeault, M., and Chen, S. (1997). Radiation regime and canopy architecture in a boreal aspen forest. Agricultural and Forest Meteorology, 86(1), 107-125.

Chong, M., Humphreys, E., and Moore, T. R. (2012). Microclimatic response to increasing shrub cover and its effect on Sphagnum CO2 exchange in a bog. Ecoscience, 19(1), 89-97.

Cobbaert, D., Rochefort, L., and Price, J. S. (2004). Experimental restoration of a fen plant community after peat mining. Applied vegetation Science, 7(2), 209-220.

Corradini, P., and Clement, B. (1999). Growth pattern and modular reiteration of a hardy coloniser *Polytrichum commune* Hedw. Plant Ecology, 143(1), 67-76.

Crockford, H., Topalidis, S., and Richardson, D. P. (1991). Water repellency in a dry sclerophyll eucalypt forest – measurements and processes. Hydrological Processes, 5(4), 405-420.

Cumming, S. G. (2001). Forest type and wildfire in the Alberta boreal mixedwood: what do fires burn? Ecological Applications, 11(1), 97-110.

Debano, L. F. (2000). The role of fire and soil heating on water repellence in wildland environments: a review. Journal of Hydrology, 231, 195-206.

DeJonge, L. W, Jacobsen, O. H., and Moldrup, P. (1999). Soil water repellency: effects of water content, temperature, and particle size. Soil Science Society of America Journal, 63(3), 437-442.

Dekker, L. W., and Ritsema, C. J. (2000). Wetting patterns and moisture variability in water repellent Dutch soils. Journal of Hydrology, 231, 148-164.

Dekker, L. W., DeBano, L. F., Oostindie, K., and van den Elsen, E. (2003). More than one thousand references related to soil water repellency. In Soil Water Repellency: Occurrence, Consequences, and Amelioration, Ritsema, C. J., and Dekker, L. W. (eds). Elsevier Science: Amsterdam; 315-352.

Dekker, L. W., Ritsema, C. J., and Oostindie, K. (2000). Extent and significance of water repellency in dunes along the Dutch coast. Journal of Hydrology, 231, 112-125.

Dekker, L. W., Ritsema, C. J., Oostindie, K., and Boersma, O. H. (1998). Effect of drying temperature on the severity of soil water repellency. Soil Science, 163(10), 780-796.

Devito, K., Mendoza, C., and Qualizza, C. (2012). Conceptualizing Water Movement in the Boreal Plains: Implications for Watershed Reconstruction Rep., 164 pp, Synthesis report prepared for the Canadian Oil Sands Network for Research and Development. Environmental and Reclamation Research Group., Alberta, Canada.

Doerr, S. H. (1998). On standardizing the ‘water drop penetration time’ and the ‘molarity of an ethanol droplet’ techniques to classify soil hydrophobicity: a case study using medium textured soils. Earth Surface Processes and Landforms, 23(7), 663-668.

Doerr, S. H., and Moody, J. A. (2004). Hydrological effects of soil water repellency: on spatial and temporal uncertainties. Hydrological Processes, 18(4), 829-832.

Doerr, S. H., Dekker, L. W., Ritsema, C. J., Shakesby, R. A., and Bryant, R. (2002). Water repellency of soils: the influence of ambient relative humidity. Soil Science Society of America Journal, 66(2), 401-405.

Doerr, S. H., Shakesby, R. A., and Walsh, R. (2000). Soil water repellency: its causes, characteristics, and hydro-geomorphological significance. Earth-Science Reviews, 51(1), 33-65.

Elumeeva, T. G., Soudzilovskaia, N. A., During, H. J., and Cornelissen, J. H. (2011). The importance of colony structure versus shoot morphology for the water balance of 22 subarctic bryophyte species. Journal of Vegetation Science, 22(1), 152-164.

Farrick, K. K. (2008). The role of ericaceous shrubs in the surface water balance and soil water availability of a cutover peatland, Quebec. Master’s Thesis, University of Waterloo, Waterloo, Ontario.

Farrick, K. K., and Price, J. S. (2009). Ericaceous shrubs on abandoned block-cut peatlands: implications for soil water availability and *Sphagnum* restoration. Ecohydrology, 2(4), 530-540. DOI: 10.1002/eco.77

Fenton, N. J., and Bergeron, Y. (2006). Facilitative succession in a boreal bryophyte community driven by changes in available moisture and light. Journal of Vegetation Science, 17(1), 65-76.

Ferone, J. M., and Devito, K. J. (2004). Shallow groundwater – surface water interactions in pond – peatland complexes along a Boreal Plains topographic gradient. Journal of Hydrology, 292, 75-95.

Flannigan, M. D., and Wotton, B. M. (2001). Climate, weather, and area burned. In Forest fire: behaviour and ecological effects. Edited by E. Johnson and K. Miyanishi. Academic Press, San Diego, Calif. Pp. 351-373.

Flannigan, M. D., Logan, K. A,, Amiro, B. D., Skinner, W. R., and Stocks B. J. (2005). Future area burned in Canada. Climatic Change, 72, 1-16. DOI: 10.1007/s10584-005-5935-y

Flannigan, M. D., Stocks, B. J., and Wotton, B. M. (2000). Climate change and forest fires. Science of the Total Environment, 262(3), 221-229.

Flannigan, M., Amiro, B., Logan, K., Stocks, B., and Wotton, B. (2005). Forest fires and climate change in the 21st century. Mitigation and Adaptation Strategies for Global Change, 11(4), 847-859.

Flannigan, M., Stocks, B., Turetsky, M., and Wotton, M. (2009). Impacts of climate change on fire activity and fire management in the circumboreal forest. Global Change Biology, 15(3), 549-560. DOI: 10.1111/j.1365-2486.2008.01660.x

Frolking, S. (1997). Sensitivity of spruce/moss boreal forest net ecosystem productivity to seasonal anomalies in weather. Journal of Geophysical Research, 102(D24), 29053-29064.

Gonsamo, A., Walter, J. N, and Pellikka, P. (2011). CIMES: a package of programs determining canopy geometry and solar radiation regimes through hemispherical photographs. Computers and Electronics in Agriculture, 79, 207-215.

Gonzalez, E., Henstra, S. W., Rochefort, L., Bradfield, G. E., and Poulin, M. (2013). Is rewetting enough to recover Sphagnum and associated peat-accumulating species in traditionally exploited bogs? Wetlands Ecology and Management, 22, 49-62. DOI: 10.1007/s11273-013-9322-6

Graf, M. D., and Rochefort, L. (2008). Techniques for restoring fen vegetation on cut-away peatlands in North America. Applied Vegetation Science, 11(4), 521-528.

Graf, M. D., and Rochefort, L. (2010). Moss regeneration for fen restoration: field and greenhouse experiments. Restoration Ecology, 18, 121-130.

Granath, G., Gonsamo, A., Walter, J. N. (2014). Cimesr: analysing canopy geometry and solar radiation regimes using hemispherical photographs. https://github.com/ggranath/cimesr.

Groeneveld, E. V., Masse, A., and Rochefort, L. (2007). *Polytrichum strictum* as a nurse-plant in peatland restoration. Restoration Ecology, 15(4), 709-719.

Groeneveld, E., and Rochefort, L. (2002). Nursing plants in peatland restoration: on their potential use to alleviate frost heaving problems. Suo, 53(3-4), 73-85.

Hajek, T., and Beckett, R. P. (2008). Effect of water content components on desiccation and recovery in *Sphagnum* mosses. Annals of Botany, 101(1), 165-173. DOI: 10.1093/aob/mcm287

Hayward, P. M., and Clymo, R. S. (1982). Profiles of water content and pore size in *Sphagnum* peat, and their relation to peat bog ecology. Proceedings of the Royal Society of London B: Biological Sciences, 215(1200), 299-325.

Hayward, P. M., and Clymo, R. S. (1983). The growth of *Sphagnum*: experiments on, and simulation of, some effects of light flux and water-table depth. Journal of Ecology, 71, 845-863.

Hebert, F., and Thiffault, N. (2011). The biology of Canadian weeds. 146. *Rhododendron groenlandicum* (Oeder) Kron and Judd. Canadian Journal of Plant Science, 91, 725-738. DOI: 10.4141/CJPS2010-012

Heijmans, M. M., Arp, W. J., and Berendse, F. (2001). Effects of elevated CO_2 and vascular plants on evapotranspiration in bog vegetation. Global Change Biology, 7(7), 817-827.

Hokanson, K. J. (2014). Influence of hydrogeological setting on peatland burn severity. Master's Thesis, McMaster University, Hamilton, Ontario.

Holm, S. (1979). A simple sequentially rejective multiple test procedure. Scandinavian Journal of Statistics, 6(2), 65-70.

Hope, G., Whinam, J., and Good, R. (2005). Methods and preliminary results post-fire experimental trials of restoration techniques in the peatlands of Namadgi (ACT) and Kosciuszko National Parks (NSW). Ecological Management & Restoration, 6(3), 214-217.

Johnson, E. A. (1981). Vegetation organization and dynamics of lichen woodland communities in the Northwest Territories, Canada. Ecology, 62, 200-215.

Karlin, E. F., and Bliss, L. C. (1983). Ecology of *Ledum groenlandicum* and *Ledum palustre* ssp. Decumbens. Arctic and Alpine Research, 15, 397-404.

Kasischke, E., and Turetsky, M. (2006). Recent changes in the fire regime across the North American boreal region – spatial and temporal patterns of burning across Canada and Alaska. Geophysical research letters, 33(9).

Kettridge, N., Humphrey, R. E., Smith, J. E., Lukenbach, M. C., Devito, K., Petrone, R. M., and Waddington, J. M. (2014). Burned and unburned peat water repellency: implications for peatland evaporation following wildfire. Journal of Hydrology, 513, 335-341.

Kettridge, N., Turetsky, M. R., Sherwood, J. H., Thompson, D. K., Miller, C. A., Benscoter, B. W., Flannigan, M. D., Wotton, B. M., and Waddington, J. M. (2015). Moderate drop in water table increases peatland vulnerability to post-fire regime shift. Scientific Reports, 5:8063. DOI: 10.1038/srep08063

Kovaleva, N. M., and Ivanova, G. A. (2013). Recovery of ground vegetation at the initial stage of fire succession. Contemporary Problems of Ecology, 6(2), 162-169.

Koyama, A., and Tsuyuzaki, S. (2010). Effects of sedge and cottongrass tussock on plant establishment patterns in a post-mined peatland, northern Japan. Wetlands Ecology and Management, 18(2), 135-148. DOI: 10.1007/s11273-009-9154-6

Krawchuk, M. A., Cumming, S. G., Flannigan, M. D., and Wein, R. W. (2006). Biotic and abiotic regulation of lightning fire initiation in the mixedwood boreal forest. Ecology, 87(2), 458-468.

Lukenbach, M. C. (2015). Hydrogeological and ecohydrological controls on peatland resilience to wildfire. PhD Thesis, McMaster University, Hamilton, Ontario.

Lukenbach, M. C., Devito, K. J., Kettridge, N., Petrone, R. M., and Waddington, J. M. (2015). Burn severity alters peatland moss water availability: implications for post-fire recovery. Ecohydrology, 9, 341–353. DOI: 10.1002/eco.1639

MacDonald, L. H., and Huffman, E. L. (2004). Post-fire soil water repellency: persistence and soil moisture thresholds. Soil Science Society of America Journal, 68(5), 1729-1734.

Mallik, A. U., and Gimingham, C. H. (1985). Ecological effects of heather burning: II. Effects on seed germination and vegetative regeneration. Journal of Ecology, 73(2), 633-644.

Maltby, E., Legg, C. J., and Proctor, M. C. F. (1990). The ecology of severe moorland fire on the North York moors: effects of the 1976 fires, and subsequent surface and vegetation development. Journal of Ecology, 490-518.

Morneau, C., and Payette, S. (1989). Postfire lichen-spruce woodland recovery at the limit of the boreal forest in northern Quebec. Canadian Journal of Botany, 67(9), 2770-2782.

Nyman, P., Sheridan, G. J, Smith, H. G., and Lane, P. N. (2014). Modeling the effects of surface storage, macropore flow and water repellency on infiltration after wildfire. Journal of Hydrology, 513, 301-313.

O'Donnell, J. A., Turetsky, M. R., Harden, J. W., Manies, K. L., Pruett, L. E., Shetler, G., and Neff, J. C. (2009). Interactive effects of fire, soil, climate, and moss on CO_2 fluxes in black spruce ecosystems of interior Alaska. Ecosystems, 12, 57-72. DOI: 10.1007/s10021-008-9206-4

Price, J. S., Rochefort, L., and Campeau, S. (2002). Use of shallow basins to restore cutover peatlands: hydrology. Restoration Ecology, 10(2), 259-266.

Price, J. S., Whitehead, G. S. (2001). Developing hydrological thresholds for Sphagnum recolonization on an abandoned cutover bog. Wetlands, 21, 32–40.

Price, J., Rochefort, L., and Quinty, F. (1998). Energy and moisture considerations on cutover peatlands: surface microtopography, mulch cover and *Sphagnum* regeneration. Ecological Engineering, 10(4), 293–312.

Proctor, M. C., and Smirnoff, N. (2000). Rapid recovery of photosystems on rewetting desiccation-tolerant mosses: chlorophyll fluorescence and inhibitor experiments. Journal of Experimental Botany, 51(351), 1695-1704.

Reed, M. S., Bonn, A., Evans, C., Glenk, K., and Hansjurgens, B. (2014). Editorial: Assessing and valuing peatland ecosystem services for sustainable management. Ecosystem Services, 9, 1-4.

Rochefort, L. (2000). *Sphagnum* – a keystone genus in habitat restoration. The Bryologist, 103(3), 503-508.

Rochefort, L., Isselin-Nondedeu, F., Boudreau, S., and Poulin, M. (2013). Comparing survey methods for monitoring vegetation change through time in a restored peatland. Wetlands Ecology and Management, 21, 71-85.

Rochefort, L., Quinty, F., Campeau, S., Johnson, K., and Malterer, T. (2003). North American approach to the restoration of Sphagnum dominated peatlands. Wetlands Ecology and Management, 11, 3-20.

Rupp, T. S., Starfield, A. M., Chapin III, F. S., and Duffy, P. (2002). Modeling the impact of black spruce on the fire regime of Alaskan boreal forest. Climatic Change, 55, 213-233.

Rydgren, K., Okland, R. H., Hestmark, G. (2004). Disturbance severity and community resilience in a boreal forest. Ecology, 85(7), 1906-1915.

Salonen, V. (1987). Relationship between the seed rain and the establishment of vegetation in two areas abandoned after peat harvesting. Ecography, 10(3), 171-174.

Schouwenaars, J. M., and Gosen, A. M. (2007). The sensitivity of *Sphagnum* to surface layer conditions in a re-wetted bog: a simulation study of water stress. Mires and Peat, 2(2), 1-19.

Seaman, M. A., Levin, J. R., and Serlin, R. C. (1991). New developments in pairwise multiple comparisons: some powerful and practicable procedures. Psychological Bulletin, 110(3), 577-586.

Shetler, G., Turetsky, M., Kane, E., and Kasischke, E. (2008). *Sphagnum* mosses limit total carbon consumption during fire in Alaskan black spruce forests. Canadian Journal of Forest Research, 38(8), 2328-2336. DOI: 10.1139/X08-057

Silvola, J., and Aaltonen, H. (1984). Water content and photosynthesis in the peat mosses *Spaghnum fuscum* and *S. angustifolium*. Annales Botanici Fennici, 21, 1-6.

Smolders, A. J. P., Tomassen, H. B. M., van Mullekom, M., Lamers, L. P. M., and Roelofs, J. G. M. (2003). Mechanisms involved in the re-establishment of *Sphagnum*-dominated vegetation in rewetted bog remnants. Wetlands Ecology and Management, 11(6), 403-418.

Sundberg, S. (2013). Spore rain in relation to regional sources and beyond. Ecography, 36(3), 364-373. DOI: 10.1111/j.1600-0587.2012.07664.x

Sundberg, S., and Rydin, H. (2000). Experimental evidence for a persistent spore bank in *Sphagnum*. New Phytologist, 148, 105-116.

Sundberg, S., and Rydin, H. (2002). Habitat requirements for establishment of *Sphagnum* from spores. Journal of Ecology, 90(2), 268-278.

Thompson, D. K., and Waddington, J. M. (2013). Wildfire effects on vadose zone hydrology in forested boreal peatland microforms. Journal of Hydrology, 486, 48-56.

Turetsky, M. R., Donahue, W. F., and Benscoter, B. W. (2011a). Experimental drying intensifies burning and carbon losses in a northern peatland. Nature Communications, 2, 514. DOI: 10.1038/Ncomms1523

Turetsky, M. R., Kane, E. S., Harden, J. W., Ottmar, R. D., Manies, K. L., Hoy, E., and Kasischke, E. S. (2010). Recent acceleration of biomass burning and carbon losses in Alaskan forests and peatlands. Nature Geoscience, 4(1), 27-31. DOI: 10.1038/NGEO1027

Turetsky, M. R., Kane, E. S., Harden, J. W., Ottmar, R. D., Manies, K. L., Hoy, E., and Kasischke, E. S. (2011b). Recent acceleration of biomass burning and carbon losses in Alaskan forests and peatlands. Nature Geoscience, 4, 27-31. DOI: 10.1038/Ngeo1027

Turetsky, M., Wieder, K., Halsey, L., and Vitt, D. (2002). Current disturbance and the diminishing peatland carbon sink. Geophysical Research Letters, 29(11).

Valat, B., Jouany, C., and Riviere, L. M. (1991). Characterization of the wetting properties of air dried peats and composts. Soil Science, 152(2), 100-107.

Vitt, D. H., Halsey, L. A., Bauer, I. E., and Campbell, C. (2000). Spatial and temporal trends in carbon storage of peatlands of continental western Canada through the Holocene. Canadian Journal of Earth Sciences, 37(5), 683-693.

Waddington, J. M., Warner, K. D., and Kennedy, G. W. (2002). Cutover peatlands: a persistent source of atmospheric CO_2. Global Biogeochemical Cycles, 16(1), 1-7. DOI: 10.1029/2001GB001398

Wein, R., and MacLean, D. (Eds.) (1983). The role of fire in north circumpolar ecosystems. (SCOPE 18). Nature and Resources. UNESCO, 19(3), 322.

Whinam, J., and Buxton, R. (1997). *Sphagnum* peatlands of Australasia: an assessment of harvesting sustainability. Biological Conservation, 82, 21-29.

Wieder, R. K., Scott, K. D., Kamminga, K., Vile, M. A., Vitt, D. H., Bone, T., Xu, B. I. N., Benscoter, B. W., and Bhatti, J. S. (2009). Postfire carbon balance in boreal bogs of Alberta, Canada. Global Change Biology, 15, 63-81. DOI: 10.1111/j.1365-2486.2008.01756.x

Zhang, Y., Chen, J. M., and Miller, J. R. (2005) Determining digital hemispherical photograph exposure for leaf area index estimation. Agricultural and Forest Meteorology, 133, 166-181. DOI: 10.1016/j.agrformet.2005.09.009