DISSERTATION

LESSONS IN ESTABLISHING PLANT COMMUNITIES ON CONSTRUCTED FENS FOR OIL SANDS MINE RECLAMATION

Submitted by

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ABSTRACT

LESSONS IN ESTABLISHING PLANT COMMUNITIES ON CONSTRUCTED FENS FOR OIL SANDS MINE RECLAMATION

The third-largest proven oil deposit in the world is in Alberta, underlying 142,000 square kilometers of Canada's boreal covered by forested uplands and peatland basins. The vast deposit is in the form of oil sands that consist of a mixture of sand, water, clay and oil. Where oil sands are near surface, they are excavated in open-pit mines that remove the overburden landscape to extract the resource. Reclamation is a legislative condition for oil sands operators to replace ecosystems that are lost. This involves recontouring the surface to recreate landscape processes and introducing plant species common in regional reference sites. Fen peatlands are the most dominant ecosystem type but provincial standards have allowed compensation with marsh wetland as they are easier to create. Oil sands extraction and reclamation is highly controversial with opponents suggesting that destroyed peatlands will not be restored. Scientists, operators and regulators are more aware that peatland reclamation is critical and despite the constraints, research is underway in two reclamation fens that have recently been constructed.

To effectively reclaim fens, we need to understand how plant species and communities respond to environmental gradients, the most effective methods to introduce species, and which success criteria are achievable. In the following chapters, I examine drivers of plant community assembly in natural and reclaimed fens and consequences of abiotic, biotic, and construction constraints on ecosystem structure and function. A major constraint in fen reclamation is achieving optimal surface topography and seasonal water table position to support desired plants. Moss-dominated fens are the most common regional peatland type and evaluating the response of mosses to submergence in natural fens provides insight into species selection and processes of recovery for reclaimed fens. I conducted a field experiment to determine the short and long-term tolerances of four fen mosses to submergence from 1 to 8 weeks. I

found that moss species vary in their responses to submergence duration and that shifts in community composition that support tolerant dominant species such as *Tomentypnum nitens* increased moss community resilience and provide stability in boreal fen ecosystems.

As part of a multi-stakeholder collaboration, the first self-sustaining reclamation fen and associated watershed was constructed within an oil sands mine site north of Fort McMurray, Alberta. To determine the most effective approach to establish fen plants, I designed and implemented a large-scale multifactorial field experiment that tested introducing moss layer transfer material (MLT), seeds, and seedlings under wood-strand mulch and with a *Typha latifolia* weeding treatment. Four years after planting, the MLT and *Juncus balticus* seedling treatment supported the highest fen bryophyte and vascular plant cover and species richness. Weeding did reduce *T. latifolia* cover but was not necessary in areas where seedlings or MLT was introduced. The most successful fen species to establish was *C. aquatilis*, which rapidly colonized but also reduced cover and richness of bryophytes and other vascular plants. To provide a broader context, I examined vegetation establishment across the two reclaimed fens that had different water level gradients and species introduction approaches. Despite differences, peat-accumulating bryophyte and vascular plant communities developed in both fens. Community convergence occurred due to dominance of *C. aquatilis*, and community divergence occurred in response to water level gradients. Dominant species adapted to site conditions can be introduce by basic approaches such as seeding. Intensive approaches such as planting seedlings or spreading MLT should be prioritized in areas of overlap along water level gradients between desirable and undesirable communities to deter establishment of non-peat forming species. Bryophyte cover and desirable species richness was highest following intensive approaches and where the summer water level was -10 cm to -40 cm from the soil surface. My research shows that it is possible to reclaim peat-accumulating bryophyte and vascular plant communities in the post-mining landscape of Alberta and that a range of successful outcomes are achievable. Previous assertions that fens cannot be reclaimed after mining activities are antiquated as large-scale construction designs and species introduction approaches are actively underway and the results are proven.

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journey with you. Lastly, to my little Emery. You are basically the sixth chapter to this dissertation and kept me up at night more than any other section, but you are by far the part of this stage in life that I am most proud of. I love watching you grow and can't wait to see who you become.

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1 INTRODUCTION

Humans have altered natural landscapes for thousands of years, but intensification over the last 50 years facilitated by advancements in technology and increases in demand have caused large-scale land conversions and substantial losses of ecological services (Bradshaw and Chadwick 1980, Millennium Ecosystem Assessment 2005, Suding 2011). The repercussions of degraded landscapes extend from irreversible losses in biodiversity to detrimental effects on human health (Millennium Ecosystem Assessment 2005). The importance of reversing degradation has never been greater and there is now consensus among land managers, policy makers, and the public that this century is in fact 'the era of restoration in ecology' (E.O. Wilson 1992). Aldo Leopold's work to reconstruct a sample of the native landscape at the University of Wisconsin Arboretum laid the foundation for restoring degraded ecosystems (Leopold 1934, Zedler 1999). As research progressed at the arboretum, the importance of ecological processes on the practice of restoration was revealed and integrated to recreate self-sustaining analogue ecosystems (Curtis and Partch 1948, Jordan et al. 1990). Ecological restoration is now defined as the process of assisting the recovery of damaged, degraded, or destroyed ecosystems (SER 2004). Despite being a relatively young scientific discipline, with publications in peer-reviewed journals beginning around 1990, ecological concepts such as succession, competition, disturbance, and ecosystem function are commonly explored and tested in restoration projects (Young et al. 2005).

Reclamation is like restoration but occurs where there has been a complete loss of ecosystem services and a replacement ecosystem is constructed (Lima et al. 2016). Reclamation projects are typically large in scale and constructed in heavily disturbed environments where connectivity to remnants of the pre-existing landscape is limited or nonexistent (Bradshaw and Chadwick 1980). Initial efforts improved degraded areas by surface recontouring, stabilization to prevent erosion or leaching of pollutants, and were vegetated with quick-growing ruderal plants or by economical approaches that depend on spontaneous recolonization (Walker et al. 2007, Skousen and Zipper 2014). Although collaborations and advancements in reclamation have occurred, many industry practitioners are engineers, contractors, and planners who follow antiquated legislative directives that do not integrate ecological concepts into project designs (SMCRA 1977, EPEA 2000). Some of the biggest challenges facing landscape-scale reclamation are the technologies and knowledge required, and the balance between desired ecological complexity and the enormous cost of implementation (Menz et al. 2013). The task is not easy because it involves understanding not only interactions between environment and plants, but also constraints specific to the site and disturbance type (Bradshaw and Chadwick 1980). Advocates supporting innovative approaches or scientific research are few in the industry, but the opportunity for improvement is enormous and relationships need to be forged. Collaborations with government agencies, industry partners, and scientists are key to developing evidence-based approaches that improve the practice of reclamation in heavily degraded landscapes. This is particularly important in the boreal region of Alberta, Canada where reclamation solutions are required after large-scale oil sands mining.

The third-largest proven oil deposit in the world is in Alberta, Canada, underlying 142,000 square kilometers and estimated to contain nearly 1.7 trillion barrels of bitumen or 10 % of the world's oil reserves (Natural Resources Canada 2017, Figure 1.1). The vast deposit was formed by estuarine and marine organisms and sediment and consist of a mixture of sand, water, clay and a type of oil called bitumen (Hein and Cotterill 2006). Bitumen is so viscous that it must be heated in situ to be pumped out or excavated in open-pit mines. Twenty percent of the oil sands deposit is near surface, within 70 m of the ground surface, and recoverable through conventional surface mining. The process involves removing and stockpiling the overburden to extract the oil sands resource. Bitumen is then separated from oil sands using hot water, NaOH and steam, a process that results in tailings slurry by-product. Tailings slurry consists of fine sand, clay, water, residual bitumen and dissolved organic and mineral compounds that is pumped into holding ponds to settle solid particles from water (Quagraine et al. 2005). Because of the regional geochemistry and the bitumen extraction process, resulting tailings sand used to recontour the landscape has high salinity, high pH, and contain residual hydrocarbon products (Hein and Cotterill 2006). This typically results in reclaimed sites having higher salinity levels than pre-mining ecosystems (Howat 2000, Purdy et al. 2005, Trites and Bayley 2009a, 2009b). The need for reclamation solutions is enormous as 10 mines are currently operating with leases that cover over 167,000 ha, with net impacts to peatlands estimated to be approximately 12,000 ha (Rooney et al. 2012). However, the nature of mining by-products material used for reclamation and extent of open-pit mining disturbance area make it challenging to reconstruct the entire landscape to resemble and function like the pre-existing ecosystems (Johnson and Miyanishi 2008, Rooney et al. 2012).

A legislative condition for oil sands project approvals includes implementing a reclamation plan following mine closure. The standard requires disturbed lands to be reclaimed to an 'equivalent land capacity', which encompasses land uses similar to predisturbance conditions but not necessarily identical to it (Government of Alberta 1999). Since the first large-scale mining operation near Fort McMurray began producing oil in 1967, reclamation has focused on marsh wetlands because they are easier to create and often spontaneously occur (Fung and Macyk 2000, Stolte et al. 2000, Rowland et al. 2009, Alberta Environment 2010, Daly et al. 2012). The time has come however to advance the practice of oil sands reclamation because the pre-existing landscape in the mineable area supported 62 % peatlands (Raine et al. 2002 in Rooney et al. 2012), the majority of which are groundwater supported fens (Vitt et al. 1998). Peatlands form where net primary production exceeds decomposition resulting in accumulation of partially decayed organic matter as peat and are a critical component in global carbon and nitrogen cycles (Wieder et al. 2006, Loisel et al. 2014). Oil sands reclamation is highly controversial with opponents suggesting that these destroyed peatlands will not be restored and the associated loss of stored carbon drastically amplifies CO_2 emission from oil production (Rooney et al. 2012). Regulators are more aware that restoring the essential function of carbon storage from peatlands in oil sands reclamation projects is key and new initiatives that inform development of regulatory guidelines are swiftly being prioritized (CEMA 2014, Environment and Parks 2017).

In the following chapters, I examine drivers of plant community assembly in natural and reclaimed peatlands and consequences of abiotic, biotic, and construction constraints on ecosystem structure and function (Figure 1.2). I studied these topics with bryophyte and vascular plants in the boreal oil sands region of Alberta, Canada. I conducted a field experiment in a natural fen, designed a large-scale multi-factorial experiment in a reclaimed fen, and synthesized results across two reclaimed fens to extract commonalities and provide guidance for future projects. I address concepts of species and community-levels of resistance and resilience to disturbance, effects of dominant species, ecological succession and niches, plant competition and facilitation, priority effects, and alternate states. My research aims to integrate ecological concepts into mine reclamation projects to advance the practice and increase the likelihood of successful outcomes. This work has been a collaboration with government agencies, industry partners, and colleagues in the United States and Canada to develop solutions that are evidence-based yet practical at large-scales.

Specifically, I address three main questions: (1) what is the tolerance of fen moss species and moss communities to submergence duration? (2) what methods are most effective to establish and support fen bryophyte and vascular plants in a constructed boreal fen? And, (3) what are the similarities in vegetation establishment between two regional reclaimed fens given differences in species introduction methods and water level gradients?

1.1 Chapter overviews

In Chapter 2, I examine the tolerance of four fen mosses and moss communities to submergence duration. This chapter was published in January 2018 in the Journal of Vegetation Science.

Borkenhagen, A. and Cooper, D.J., 2018. Tolerance of fen mosses to submergence, and the influence on moss community composition and ecosystem resilience. Journal of Vegetation Science, 29(2), pp.127-135.

Disturbances to moss dominated peatlands such as fires, large precipitation events, permafrost thaw, or human land use changes can create temporary or permanent submerged areas (Turetsky and Louis 2006, Stinson et al. 2011). Additionally, a major constraint in fen reclamation is achieving the optimal surface topography and seasonal water table position within the hydrologic niche of desired peatland plants. Flooding in peatlands can affect ecosystem function by altering greenhouse gas fluxes and reducing vegetation cover (Kelly et al. 1997, Roulet et al. 1997, St. Louis et al. 2000). Shifts in community composition and establishment of more tolerant species can limit these changes (Camill 1999, Beilman 2001, Granath et al. 2010), although recovery could depend on the frequency and duration of flooding, water chemistry, and an available submergence tolerant species pool. My goal with this research was to determine how moss species recover following flooding events and the implications on the resilience of boreal fens (Turetsky et al. 2012).

To assess the tolerances of fen mosses to submergence duration and the implications on moss community resilience, I conducted an experiment in a rich-fen near Fort McMurray, Alberta in 2014, 2015 and 2017. Four replicate plugs of four common fen moss species (*Hamatocaulis vernicosus*, *Sphagnum warnstorfii, Tomentypnum nitens,* and *Aulacomnium palustre*) were extracted from monospecific patches and submerged in dugout pits under 10 cm of water for 8, 6, 4, 2, or 1 week(s). All plugs were removed from the water and planted at the site in a bare peat area with near surface water tables (Figure 1.3). To determine the short and long-term effects of submergence, canopy cover of live (green) moss was visually estimated to the nearest percent in each plug to species six weeks after submergence and 11 months after submergence. Using this experiment, I tested the following hypotheses: (1) Fen moss species tolerance to submergence duration will coincide with their occurrence along a hummock-hollow gradient, (2) When intolerant species decline in abundance, other more tolerant species may establish, forming a diverse moss community that maintains or restores moss cover, and (3) Fen moss community resilience to disturbance from submergence duration is maintained by tolerant species and/or shifts in community composition.

In Chapter 3, I designed and implemented a multi-factorial experiment testing various species introduction approaches on a reclaimed fen and evaluated establishment of bryophytes and vascular plants over four years.

As part of a multi-stakeholder collaboration between Canadian and American Universities, industry partners, and government funding agencies, the first self-sustaining reclamation fen and associated watershed was constructed within the Millennium mine lease at Suncor Energy Inc. oil sands mining operations site, near Fort McMurray, Alberta (Price et al. 2010, Daly et al. 2012). The overall research program goal is to create a functioning fen comparable to natural fens in the region and to develop applicable methods for similar reclamation projects.

Designed by Price et al. (2010), construction of the Nikanotee Fen (the Fen) was completed in 2013. The Fen was designed so that annual precipitation would be adequate to infiltrate and recharge the watershed to maintain hydrologic conditions suitable for fen vegetation establishment and peat accumulation (Price and Whitehead 2001, Price et al. 2010). The project includes an upland watershed of tailings material that provides surface and ground water flow into a fen basin constructed of salvaged donor rich fen peat substrate (Price et al. 2010). To accelerate the establishment of target species, the Fen was planted in 2013 with different methods of introduction for fen vascular plant and bryophyte species.

The experiment was a two-factor randomized block split-plot design (n = 12 blocks). Due to unexpected flooding and peat subsidence, only 5 replicate blocks were available for sampling and analysis (Figure 1.4). The blocks are divided into 7 whole-plot factor planting treatments; (1) *Carex aquatilis* seedlings; (2) *Juncus balticus* seedlings; (3) *C. aquatilis* seedlings + moss layer transfer (MLT) material; (4) *J. balticus* seedlings + MLT material; (5) MLT material; (6) mixed seeds; and (7) an unplanted control. Seeds were locally collected and sown or grown into seedlings by a local nursery. Seedlings were planted at a density of $3/m^2$. Material for the MLT was harvested from the top 10 cm of a rich-fen using a large rototiller mounted to an excavator and spread by hand at a 1:10 ratio of harvested to donor site (Rochefort et al. 2003). All species selected for introduction are regionally abundant in rich fens (Chee and Vitt 1989) and some exhibit suitable tolerances to elevated salinity levels (Pouliot et al. 2012, Pouliot et al. 2013). Each plot is further divided into 4 split-plot mulchweed treatments (mulch/no weed, mulch/weed, no mulch/weed, no mulch/no weed). To protect establishing bryophytes and reduce non-peatland species invasion, WoodStraw® mulch was applied to create 90% cover, similarly to agricultural straw applications in bog restorations (Rochefort et al. 2003). A *Typha latifolia* weeding treatment was implemented for the first three years to suppress establishment and allow desirable fen species to proliferate and potentially exclude future *T. latifolia* invasions (Lishawa et al. 2017).

My objectives with this research were to evaluate bryophyte, vascular plant, and *T. latifolia* percent cover and community diversity over four years in response to planting treatments, wood-strand mulch, weeding, and depth to water level. Using this experiment, I tested the following hypotheses: (1) Bryophyte cover would be greatest where the moss layer transfer was used and under wood-strand mulch, (2) Vascular plant cover would be highest where the moss layer transfer and seedlings were introduced, (3) *Typha latifolia* cover would be higher in unplanted areas and reduced by a weeding treatment, and (4) Bryophyte and vascular plant diversity would be highest where the moss layer transfer was used.

In Chapter 4, I collaborated with researchers at the Syncrude Sandhill Fen, led by Dr. Dale Vitt of Southern Illinois University, to compare vegetation establishment at both constructed fens in 2017 and evaluate outcomes from different species introduction methods and water level gradients.

New regulatory directives prompted initiatives by two large oil sands mine companies to design and construct fen peatlands (Daly et al. 2012, Wytrykush et al. 2012). Because peatland creation is a new concept, the development teams generated different designs. The Nikanotee Fen and watershed was constructed within the Millennium mine lease at Suncor Energy Inc. and completed in 2013. The design was based modeling of long-term climate data and vegetation moisture requirement thresholds and consisted of an isolated upland-fen system (Price and Whitehead 2001, Price et al. 2010, Daly et al. 2012, Ketcheson et al. 2016). The Sandhill Fen and watershed was constructed on Syncrude Canada Limited's Mildred Lake lease and completed in 2012. The design was based on regional groundwater exchange dynamics, and mimicked an undulating landscape of connected hummock uplands, ephemeral draws, and fen basins (Wytrykush et al. 2012, Ketcheson et al. 2016).

Species introduction approaches at the two reclaimed fens varied from natural regeneration in unplanted areas, basic approaches that included broadcast seeding, to intensive approaches of planting seedlings and harvested surface fen propagules using the moss layer transfer method (Rochefort et al. 2003). Evaluating outcomes of these methods across sites can help inform decisions about when we can rely on natural regeneration or basic approaches, where more intensive intervention is required, and whether selected species are appropriate or chosen targets are realistic given constraints (Prach and Hobbs

2008, Matthews and Spyreas 2010, Holl and Aide 2011). Abiotic and biotic factors may also influence species and community development, in particular water table depth and competition by dominant species. Understanding the factors that affect establishment of peat-forming vegetation is essential to efficiently allocate reclamation resources and maximize successful outcomes in future projects (Holl and Aide 2011). Vegetation surveys and depth to water table measurements were conducted at both fens in July 2017 (Figure 1.5). My goal with this research was to extract commonalities and determine the most effective strategies to establish peat-forming plants in reclaimed fens. In the chapter, I address the following five questions: (1) Which plants are abundant in the reclamation fens and how do these species respond to the water level gradient? (2) Do vegetation communities at the sites converge or diverge and how are they influenced by the species introduction approach and water level gradient? (3) Which communities support bryophytes and desirable fen species cover and species richness? (4) How does water level affect bryophyte cover and desirable fen species richness in each community? And (5) How does the ratio of desirable to undesirable species of each community vary in response to the species introduction approach and water level gradient?

1.2 FIGURES

Figure 1.1 Oil sands deposit in Alberta, Canada. Image courtesy of the Energy Resources Conservation Board and RB Capital Markets.

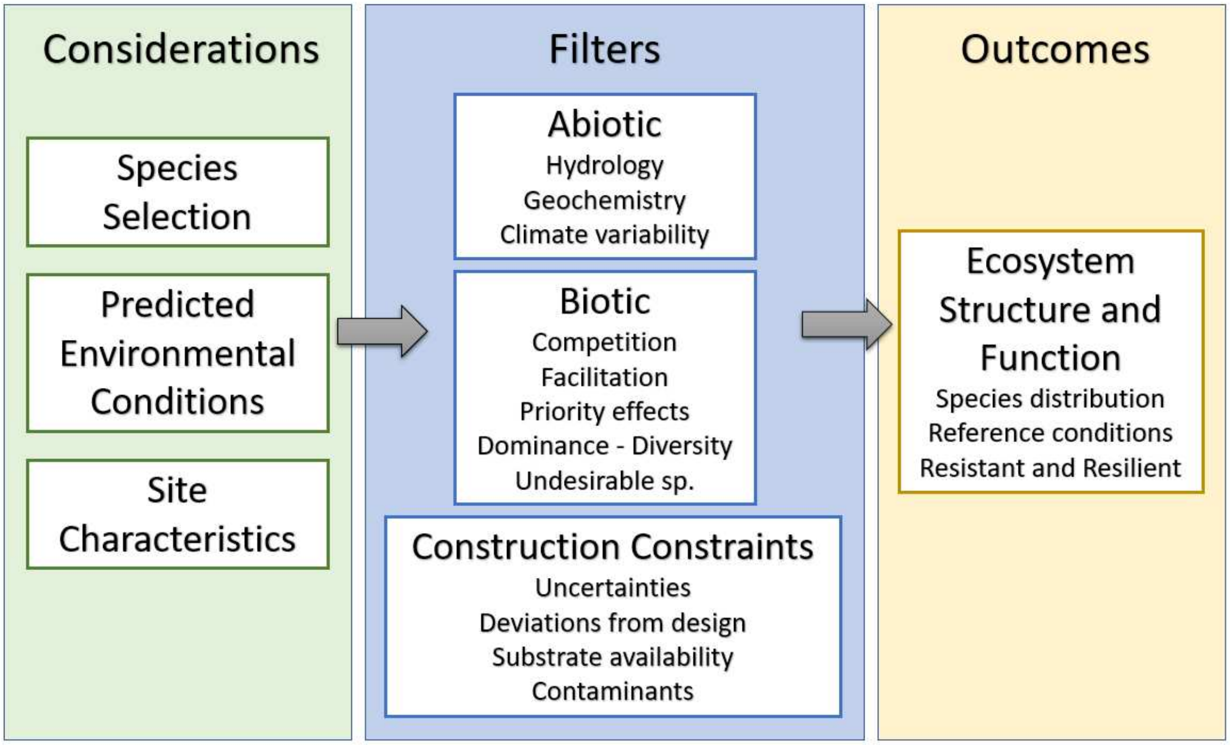

Figure 1.2 Conceptual model of the process and concepts that influenced experimental designs, analysis and interpretation of results during my dissertation research.

Figure 1.3 Moss community plugs planted in bare peat at a rich-fen site following 8 weeks of submergence in situ.

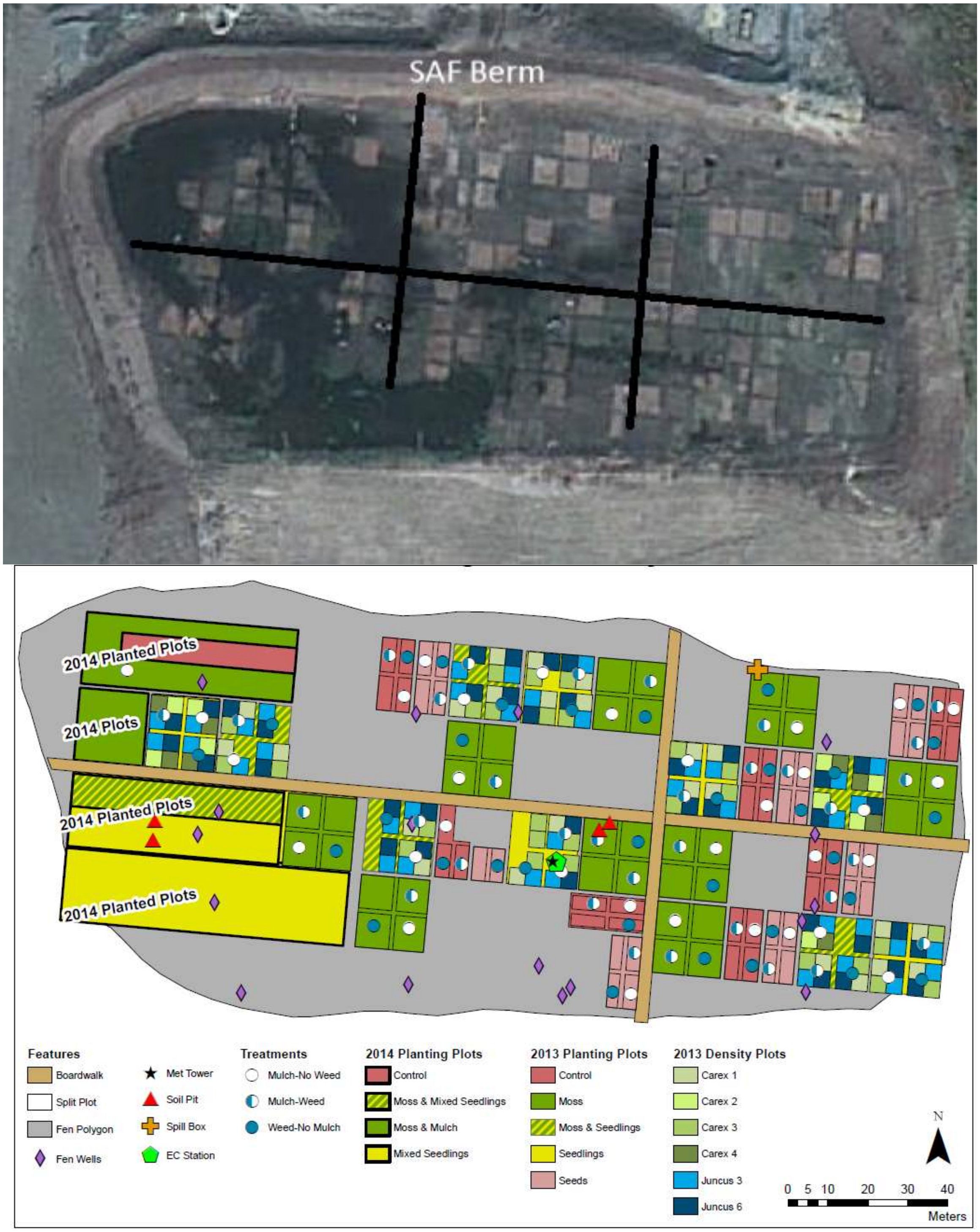

Figure 1.4 *Top* - Areal view of the Nikanotee Fen in 2013. The subplots are distinguishable as mulched (light squares) and un-mulched (dark squares) treatments. Pooled water is visible on the west side and in the middle of the fen. *Bottom* - Map of experimental design and treatment plots sampled at the Nikanotee Fen. Plots planted in 2014 were not sampled in this study.

Figure 1.5 Conducting vegetation surveys of bryophyte and vascular plant species at the Sandhill Fen in 2017. Pictured are from left to right, Dr. Dale Vitt, Melissa House, and Jeremy Hartsock.

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2 TOLERANCE OF FEN MOSSES TO SUBMERGENCE, AND THE INFLUENCE ON MOSS COMMUNITY COMPOSITION AND ECOSYSTEM RESILIENCE

2.1 INTRODUCTION

Northern peatlands contain one of the largest carbon pools in the biosphere (Yu 2012) and have been a critical component of the global carbon cycle throughout the Holocene (Harden et al. 1992; Vasander & Kettunen 2006; Yu et al. 2010). Disturbances to peatlands such as fires, peat-harvest, and flooding can lead to carbon loss as living biomass and stored peat is combusted, decomposed, or eroded (Turetsky & Louis 2006; Stinson et al. 2011). Increases in disturbance frequency and intensity in recent decades have shifted Canada's boreal forests from a carbon sink to a source, a trend that is projected to intensify (Goodale et al. 2002; Wieder et al. 2009; Metsaranta et al. 2010). Moss-dominated communities play a key role in the resilience and stability of boreal ecosystems, with ecological models clearly indicating their strong influence on the cycling of water, nutrients, energy, and carbon (Turetsky et al. 2012). Empirical evidence that identifies the resistance and resilience of moss species and communities to perturbations is limited, yet critical to inform model estimates and predict ecosystem responses to disturbances (Turetsky et al. 2012).

Flooding is the submergence of an area that is normally exposed due to water table rise above the soil surface. Flood disturbance caused by beaver damming, large precipitation events, permafrost thaw, or human land use changes can create temporary or permanent submerged areas (Turetsky & Louis 2006). Increases in the residence time of water above the soil surface or water alkalinity can lead to mortality of intolerant species and a transition from bog to fen or marsh communities with a reduction in plant cover and increase in open water (Asada et al. 2005; Granath et al. 2010; Vicherová et al. 2015). Flooding dramatically affects ecosystem function by increasing heat fluxes (Roulet et al. 1992), methane emissions (Roulet et al. 1997; Kelly et al. 1997), and dissolved organic carbon and nitrogen production (Kim et al. 2014), as well as decreases in carbon sequestration due to a reduction in vegetation cover (St. Louis et al. 2000). Shifts in community composition and establishment of more tolerant species can limit these changes (Camill 1999; Beilman 2001; Granath et al. 2010), although recovery could depend on the frequency and duration of flooding, water chemistry, and an available submergence tolerant species pool.

The frequency and duration of flooding in boreal peatlands due to climate change and human activities is expected to increase. Average annual precipitation and mean annual temperature rise (Price et al. 2013) have already increased the rate of permafrost melting and created positive feedbacks by increasing the number and extent of thermokarst ponds (Walter et al. 2006; Turetsky et al. 2007). Recent extreme fires have exceeded the known range of natural variation in the boreal forest (Kelly et al. 2013) and deep burning of peat soil has increased the extent and duration of flooding (Lukenbach et al. 2015). Direct anthropogenic disturbances from livestock-grazing (Worrall et al. 2007), roads (Joosten & Clarke 2002; Bocking et al. 2017), and water supply reservoir construction (Kelly et al. 1997; St. Louis et al. 2000) can also flood peatlands through surface compaction and water impoundment. Evaluating the effect of these alterations on peatland moss species and communities is vital for assessing impacts to ecosystem function and the processes of recovery.

Flooding may not always alter a peatlands function because moss communities exist along a broad range of hydrologic conditions (Zoltai & Vitt, 1995; Asada et al. 2005; Granath et al. 2010). The species that persist in saturated sites are resistant to some duration of submergence and shifts in community composition that include these species could prevent carbon losses after a disturbance. Changes have been observed when fen vegetation cover increases around thermokarst ponds and within collapse scars in bogs with thawing permafrost (Camill 1999; Payette et al. 2004), post-fire depressions (Benscoter et al. 2005; Lukenbach et al. 2015), and experimentally flooded complexes (Asada et al. 2005). The vegetation in these features typically shift from Sphagnum spp. dominated to true mosses (Vitt & Chee 1990), with increased peat accumulation rates compared to adjacent drier or frozen areas (Robinson & Moore 1999; Turetsky et al. 2000). Fen mosses have defined niches along microtopographic gradients relative to water level (Gignac et al. 1991; Rydin & Jeglum 2013; Vitt 2014), and the desiccation tolerance of many moss species has been extensively studied (Proctor 2001; Oliver et al.

2005; Proctor et al. 2007; Manukjanová et al. 2014), but their tolerance to submergence remains poorly known.

The tolerance of a species or community to disturbance can be measured as its resistance and resilience (Holling 1973; Rykiel 1985). Resistance is the short-term degree of change caused by a disturbance and can be calculated as the ratio of a measured variable in perturbed and unperturbed samples. Resilience is the long-term change of a measured variable returning to its predisturbance level and can be measured as the ratio between the perturbed and control samples (Griffiths & Philippot 2013). Together, they are important metrics of ecosystem stability and recovery toward the pre-disturbance condition or displacement towards an alternate state (Pimm 1984; Gunderson 2000). Losing dominant species that are intolerant of a certain type or level of disturbance can erode ecosystem function (Smith & Knapp 2003; Sasaki & Lauenroth 2011; Winfree et al. 2015; Lohbeck et al. 2016), but if other tolerant species can establish, the community may shift and retain a similar function (Robinson & Moore 1999; Camill et al. 2001; Turetsky et al. 2007).

Understanding the tolerance of fen moss species and moss communities to a submergence duration gradient will provide guidance for predicting the effects of natural or human caused flooding variation and ecosystem resilience. I conducted a field experiment to assess the short and long-term responses to submergence duration of four regionally abundant fen moss species and the communities they dominate, Aulacomnium palustre (Hedw.) Schwaegr., Hamatocaulis vernicosus (Mitt.) Hedenäs, Sphagnum warnstorfii Russ., and Tomentypnum nitens (Hedw.) Loeske. In this paper I test the following hypotheses: (1) Fen moss species tolerance to submergence duration will coincide with their occurrence along a hummock-hollow gradient, (2) When intolerant species decline in abundance, other more tolerant species may establish, forming a diverse moss community that maintains or restores moss cover, and (3) Fen moss community resilience to disturbance from submergence duration is maintained by tolerant species and/or shifts in community composition.

2.2 METHODS

STUDY SITE

The study was conducted in a treed rich fen dominated by *Larix laricina*, *Betula glandulosa*, *Carex aquatilis*, *Tomentypnum nitens,* and *Sphagnum angustifolium*. The fen is located 25 km north of Fort McMurray, Alberta, Canada (56° 56' 34" N, 111° 33' 9" W) and has an average annual precipitation of 419 mm, approximately 316 mm falling as rain and 134 cm as snow. Mean daily temperature during the growing season (May to September), is 13.3° C (local weather station at 56° 39' N, 111° 13' W; Environment Canada 2016a). The experiment was conducted from June 10, 2014 to June 12, 2015. For the growing months during the length of the study, the mean temperature was 14.5 °C and total rainfall was 224 mm (Environment Canada 2016b). Additional experimentation was conducted from July 10 to August 19, 2017, when the mean temperature was 18 °C and total rainfall was 35 mm (Environment Canada 2017b).

Fen water samples were collected on August 8, 2014, and July 11, 2015, from three hand-dug pits where the submergence experiment was conducted. Electrical conductivity (EC) and pH was measured at the time of sampling using a Thermo Scientific™ Orion™ Conductivity and Temperature probe. Water samples were taken in clean 60 ml high density polyethylene vials, filtered in the lab within 24 hours through 0.45 µm nitrocellulose filters, decanted, and frozen until analysis at the Biotron Experimental Climate Change Research Facility at Western University, London, Ontario, Canada. Major anions and cations analyzed included F^-, Cl^-, Br^-, NO_3^-, PO_4^{3-}, SO_4^{-2}, Na^+, NH^{4+}, K^+, Mg^{2+}, and Ca^{2+} (Appendix I). Elemental concentrations were within ranges observed for other regional rich fens (Vitt & Chee 1990). Mosses were submerged in hand-dug pits within a few meters of where they were naturally growing in the fen.

EXPERIMENTAL DESIGN AND SAMPLING

Four moss species were selected for this study based on their commonality and abundance in regional fens (Chee & Vitt 1989) and occurrence along a hummock-hollow gradient (Vitt & Andrus 1977;

Gignac et al. 1991; Hedenäs & Kooijman 1996; Hájková & Hájek 2004). *Hamatocaulis vernicosus* (Mitt.) Hedenäs is found in lawns or wet depressions, *Sphagnum warnstorfii* Russ. forms low hummocks or lawns, *Tomentypnum nitens* (Hedw.) Loeske creates hummocks, and *Aulacomnium palustre* (Hedw.) Schwaegr. occurs on hummock tops (Vitt 2014). These species are most abundant in regional fens where the water table is 10 to 30 cm below the surface (Gignac et al. 1991; Vitt 2014). Four replicate plugs (10 cm diameter and 5 cm deep) of each species were extracted in PVC tubes from monospecific patches (>95% of target species) in the fen over the course of eight weeks. The plugs were named to represent the original dominant moss; 1) TnOM: *Tomentypnum nitens* Original Moss, 2) HvOM: *Hamatocaulis vernicosus* Original Moss, 3) ApOM: *Aulacomnium palustre* Original Moss, and, 4) SwOM: *Sphagnum warnstorfii* Original Moss. The plugs were submerged at the same site in dugout pits under 10 cm of water for 8, 6, 4, 2, or 1 week(s) from June 1-July 30, 2014. All moss plugs were removed from the water on August 6, 2014 and planted at the site in bare peat where the water table averaged 7 cm below the soil surface at the time of planting. This area was selected to ensure that none of the species would be water stressed and they would not undergo a second submergence event. Moss cores were sheltered from full-sun using a suspended 50 % black shade cloth from the date of planting to the end of the 2014 growing season. To determine the short and long-term effects of submergence, absolute percent canopy cover of live (green) moss was visually estimated to the nearest percent in each plug to species on September 23, 2014 (six weeks after submergence) and June 12, 2015 (11 months after submergence) (Appendix II). My original experiment did not include a non-submergence control. I conducted additional experimentation from July 10 to August 19, 2017 to test control transplants for their short-term response of planting without submergence (zero-weeks). Four replicate plugs of each moss species were extracted and planted as in the original experiment. After six weeks, the effect of planting was evaluated by estimating the absolute percent canopy cover of live moss. In the original experiment, the plugs were planted from August 6, 2014 and evaluated on September 23, 2014. The climatic conditions during these 6 weeks differed slightly as 2017 averaged 5 ^{o} C warmer and received 24 mm less precipitation, which is a disadvantage for desiccation prone mosses. In contrast, moss cover either did not decline or declined to the same degree as the submerged population (*H. vernicosus*) suggesting that the additional experiment successfully tested the effect of planting despite the difference in year and seasonal period. Statistical analysis did not directly compare responses of submergence controls to the submerged plugs.

DATA ANALYSIS

To evaluate the short and long-term effects of submergence duration on moss percent cover, I used a one-way nonparametric Kruskal and Wallis Rank Sum Test, because variances were non-constant and data were not normally distributed. A Conover multiple comparison test was performed post-hoc to determine statistical significance (α = 0.05) between the sample medians (Conover & Iman 1979; Conover 1999). To evaluate the change from short and long-term evaluation periods, I used a two-way ANOVA with weeks of submergence and evaluation period as fixed factors. A Tukey-adjusted least squares means test was applied when the main effect was significant to determine statistical significance (α=0.05) between the marginal means. All analyses were performed in R using the Stats and PMCMR packages (R Core Team 2016). One replicate plug of *T. nitens* that had been submerged for 4 weeks was disturbed during planting and removed from the analysis *a priori*.

Resistance and resilience were calculated as change relative to the control (Kaufman 1982) using log response ratios (Hedges et al. 1999), which is the effect size of a log-proportional change between the treatment and control. I refer to resistance as the species' or moss communities' short-term (6 weeks after submergence) percent cover over the non-submergence control percent cover, and resilience as the long-term (11 months after submergence) percent cover of formerly submerged plugs over the pre-submergence canopy cover, standardized as 100 percent cover. Values greater than -0.7 indicate high resistance/resilience and represent less than a 50 % proportional change in response to disturbance compared to control values.

2.3 RESULTS

The four tested fen moss species differed in their response to duration of submergence, but tolerances did not completely coincide with their occurrence along a hummock-hollow gradient (Figure

2.1; Appendix III). During our experiment, *H. vernicosus* was the only tested moss whose cover did not decline in response to submergence duration or change between short and long-terms. *Tomentypnum nitens* was the second most tolerant species, declining significantly after 8 weeks of submergence in the short-term but having some recovery over the long-term. *Aulacomnium palustre* cover declined in the short-term after 4 or more weeks of submergence, however recovery over the long-term was limited in plugs submerged for 6 weeks or longer. *Sphagnum warnstorfii* was most negatively affected by submergence, declining to less than 2 % across all submergence durations over the long-term.

The reduction in moss cover when intolerant species were lost due to submergence was buffered in some cases by changes in moss species composition over-time (Figure 2.1; Appendix III). *Tomentypnum nitens* maintained dominance in the TnOM plugs, contributing 98 % of total moss cover over the long-term. In the HvOM plugs, *H. vernicosus* contributed 85 - 90 % of total moss cover as *Ptychostomum pseudotriquetrum* (Hedw.) D.T. Holyoak et N. Pedersen (common in fen lawns)*, T. nitens*, and *Calliergon giganteum* (Schimp.) Kindb. (common in fen pools) established in plugs across all submergence durations. *Aulacomnium palustre* contributed 90 % of total moss cover over the short-term in the ApOM plugs, but only 75 % in the long-term after *T. nitens* established. Despite the relative decline in *A. palustre*, total moss cover increased to a greater extent over the long-term because of *T. nitens*, particularly in plugs submerged for up to 6 weeks. *Sphagnum warnstorfii* in the SwOM plugs declined from 54 % to 5 % of total moss cover over time after *T. nitens* and *A. palustre* established. Despite the near loss of *S. warnstorfii* the establishment of other moss species restored moss cover in plugs that had been submerged for up to 2 weeks.

Fen moss community resilience to submergence duration was maintained by tolerant species and shifts in community composition (Figure 2.2). *Hamatocaulis vernicosus* and the moss community in HvOM plugs were resistant and resilient to all submergence durations, averaging 4 % and 19 % proportional change from control in the short (resistance metric (RsM) = 0.04) and long-term (resistance metric (RlM) = -0.21). *Tomentypnum nitens* and the moss community in TnOM plugs were resistant and resilient to up to 6 weeks of submergence, and recovered from 25 % to 15 % proportional change from control over time (RsM = -0.29 and RlM = -0.17). *Aulacomnium palustre* was only resistant to up to 2 weeks of submergence, and decreased from 23 % to 35 % proportional change from control over time (RsM = -0.27 and RlM = -0.42). However, increases in *T. nitens* cover in the moss community in the ApOM plugs provided resilient to up to 4 weeks of submergence. Despite a shift in composition in the SwOM plugs, the moss community was not resilient to submergence. This suggests that if losses of intolerant species are significant, recovery of moss cover may take more than one growing season.

2.4 Discussion

Hamatocaulis vernicosus and *T. nitens* were most tolerant of submergence and when intolerant species declined in abundance, *T. nitens* commonly established to restore moss cover and increase moss community resilience. The shift in species composition suggests that ecosystem resilience can be maintained by diversity if tolerant species are present and able to replace intolerant species following a disturbance (Elmqvist et al. 2003). This provides insight into how disturbance tolerant species increase ecosystem stability and resilience in response to environmental changes (Smith & Knapp 2003; Sasaki & Lauenroth 2011).

As expected, *H. vernicosus* tolerated submergence as it occurs in areas of rich fens that experience recurring flooding (Vitt 2014). *Tomentypnum nitens* was also resistant to submergence, an unexpected result for this species that typically forms hummocks in boreal fens (Vitt 2014). This suggests that submergence tolerance may be an evolutionary innovation as *Tomentypnum* and *Hamatocaulis* are in the same clade of *Amblystegiaceae* mosses (Hedenäs & Kooijman 1996). Although *T. nitens* typically occurs higher above the water table, its hydrologic niche overlaps that occupied by *H. vernicosus* (Hedenas & Kooijman 1996). *Tomentypnum nitens'* adaption to relatively dryer habitats may be the derived trait (Hedenas & Kooijman 1996) and biotic interactions at the water level could restrict it to hummocks (Gignac 1992; Robroek et al. 2007; Udd et al. 2015).

Aulacomnium palustre was tolerant of short periods of submergence, possibly due to its morphological characteristics. *Aulacomnium palustre* is considered semi-aquatic and may persist for short periods under water as stem tomentum and tightly clustered leaves can trap air bubbles and provide a source of carbon dioxide (Vitt & Glime 1984). However, its high leaf density and thick cell walls (Crum & Anderson 1981) would likely reduce carbon assimilation under longer durations of submergence, especially in stagnant water (Rice & Schuepp 1995; Jenkins & Proctor 1985). Semi-aquatic species of *Sphagnum* can also tolerate short periods of submergence (Vitt & Glime 1984; Rice & Schuepp 1995; Granath et al. 2010), although the loss of *S. warnstorfii* in our experiment could have resulted from changes in growth form. *Sphagnum* survival has been shown to be impeded by etiolation and dissociation into loose mats when subjected to long-term flooding, making them more prone to desiccation when the water receded (Rochefort et al. 2002).

The chemistry of flood waters can also directly affect moss species persistence, which is normally buffered by subsurface water tables. High levels of calcium and bicarbonate, and high pH in rich fen waters has been shown to limit intolerant *Sphagnum* spp. due to saturation of cell wall exchange sites and their insufficient control over the balance of intracellular Ca^{2+} concentrations (Vicherová et al. 2015). In contrast, submergence for three weeks under high Ca^{2+} and pH waters enhanced the growth of *T. nitens, H. vernicosus, A. palustre*, and other true mosses (Vicherová et al. 2015). Rich fen waters can therefore limit intolerant species from persisting and encourage tolerant species to dominate in areas that undergo frequent or long duration flood events (Granath et al. 2010; Vicherová et al. 2015).

Changes in moss community composition over-time varied depending on the original species tolerance of submergence and the other species' ability to establishment. Submergence tolerant species can maintain their dominance, whereas less tolerant species decline in abundance and open canopy space for other species. The processes that allowed other species to establish in the plugs were not investigated, however the immigration potential of mosses is high because of their high fecundity and dispersal ability (Campbell et al. 2003), multiple asexual reproduction strategies (Frey & Kürschner 2011), and abundance of propagules retained in surface peat layers (Campeau & Rochefort 1996).

Establishment potential and competitive ability differs by moss species (Li & Vitt 1995; Borkenhagen & Cooper 2016), and our findings suggest that a few dominant resistant and resilient species, not species richness, were most important for recovering ecosystem function following disturbance (Winfree et al. 2015). *Tomentypnum nitens* established in plugs of all tested mosses and may have reduced the recovery of the less tolerant *S. warnstorfii* and *A. palustre*. Similar interactions have been observed in multi-species experiments where *Sphagnum* spp. was outcompeted by true mosses in wet rich fens (Udd et al. 2015) and *A. palustre* was competitively excluded in mature communities (Li & Vitt 1995). In boreal rich fens, *T. nitens* is one of the most abundant and has the widest habitat niche breath among true peatland moss species (Gignac 1992; Vitt et al. 2009). The ability of *T. nitens* to establish and dominate is likely influenced by its resistance and resilience to submergence along with its tolerance of desiccation (Manukjanová et al. 2014; Goetz & Price 2015; Borkenhagen & Cooper 2016). My results stress the importance of *T. nitens* in maintaining ecosystem function and resilience to drying and flooding disturbances.

If flood duration increases in fens, particularly where flood tolerant species such as *T. nitens* or *H. vernicosus* are absent, ecosystems could lose moss cover and transition into an alternative state. Legacy effects that persist after flood-waters receded could include bare areas devoid of live mosses or the invasion of aquatic vascular plant species that alter ecosystem function because they contribute little to carbon sequestration (Camill 1999; Beilman 2001; Asada et al. 2005; Rochefort et al. 2002; Campeau et al. 2004). My experiment evaluated the effect of a one-time flood but suggests that the limitations on moss persistence would be further exacerbated by an increase in flood frequency and/or duration.

The restoration of peatlands disturbed by natural or anthropogenic caused flooding could expedite the recovery of natural structure and function (Falk et al. 2006; Suding 2011) and offset carbon losses (Vasander et al. 2003; Andersen et al. 2016; Chimner et al. 2016). However, species selection must consider water chemistry and the potential for recurring floods, especially in future climates scenarios (Harris et al. 2006; Price et al. 2013). Efforts to restore peatland mosses on oil and gas well pads, and oil-sands and peat mines have been successful, but not in areas that regularly flood due to construction constraints and land settling (Rochefort et al. 2002; Campeau et al. 2004; Caners & Lieffers 2014; Ketcheson et al. 2016). Legacy effects of flooding have also been observed where *Sphagnum* spp.

recolonization was limited in areas with deep flooding (Campeau et al. 2004) and efforts to recreate pools in restored peatlands have failed to establish plant communities similar to those of natural peatland pools (Fontaine et al. 2007). To limit the effects of future flood disturbance, introducing moss species that are tolerant of a range of submergence durations and water chemistry could allow restored ecosystems to reverse carbon losses.

Disturbances have increased recently in boreal peatlands, altering their carbon balance from a sink to a source (Turetsky & Louis 2006; Metsaranta et al. 2010). Because peatlands are hydrologically and climatically regulated (Zoltai & Vitt 1995; Yu et al. 2010), many disturbances exacerbate drying or flooding. Mosses are a critical component of boreal peatland vegetation and although their response to desiccation has been documented (Manukjanová et al. 2014), the consequences of flooding is poorly understood. My research indicates that some fen moss species are resistant to submergence and resilient to at least two months of submergence. The abundance of less tolerant species may decline in response to submergence, but rapid shifts in community composition can provide resilience to recovery where flooding is of shorter duration or frequency. Species diversity is often reported to provide ecosystem stability (Loreau et al. 2002; Hooper et al. 2005), but our results indicate that recovery by the dominant tolerant species is more important. I provide empirical evidence of the tolerances of certain dominant fen mosses to submergence disturbance, substantiating the critical role moss communities have in maintaining the functional stability of boreal ecosystems.

2.5 FIGURES

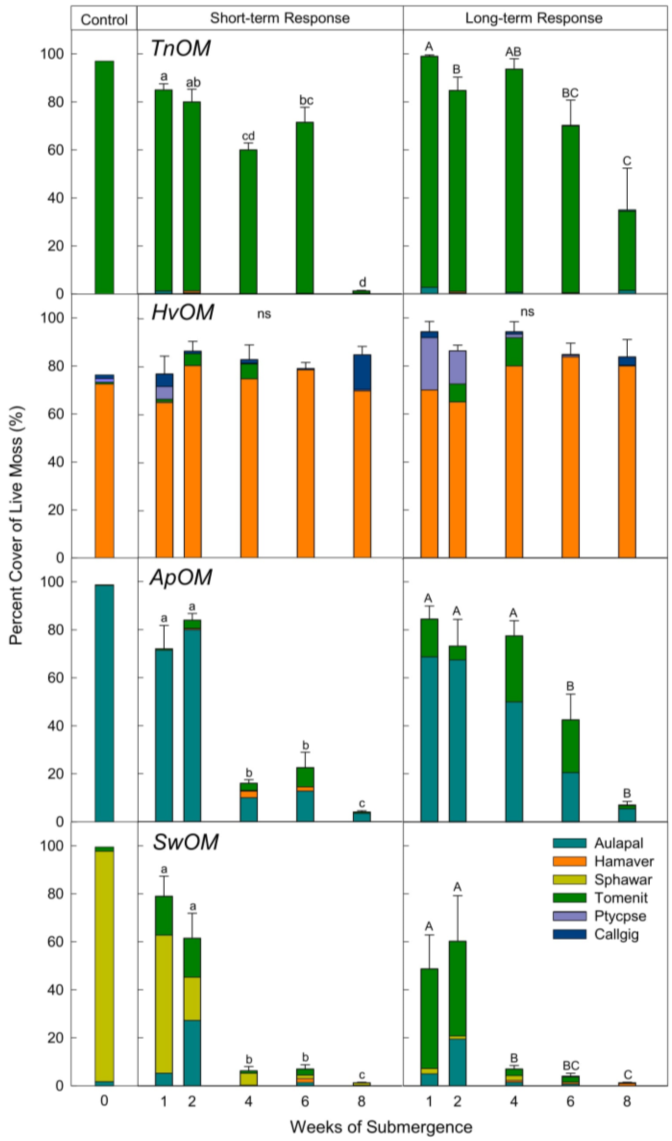

Figure 2.1. Percent cover of live moss from plugs named to represent the original dominant moss; TnOM= *Tomentypnum nitens* Original Moss, HvOM = *Hamatocaulis vernicosus,* ApOM = *Aulacomnium palustre* Original Moss, and, SwOM= *Sphagnum warnstorfii* Original Moss. Additional species that opportunistically established during the experiment include *Ptychostomum pseudotriquetrum* and *Calliergon giganteum.* Responses to the five durations of submergence were evaluated six weeks after submergence (Short-term response) and 11-months after submergence (Long-term response). Bars represent means values with standard error. Means with different letters represent total moss covers that are significantly different within short and long-term response periods (Kruskal-Wallis Conover's test for multiple comparison, $P < 0.05$; $n = 4$ for each species, except *T. nitens* in Week 4 has $n = 3$). Multiple comparisons data for original moss species from each plug are presented in Appendix III, Table A3.2.

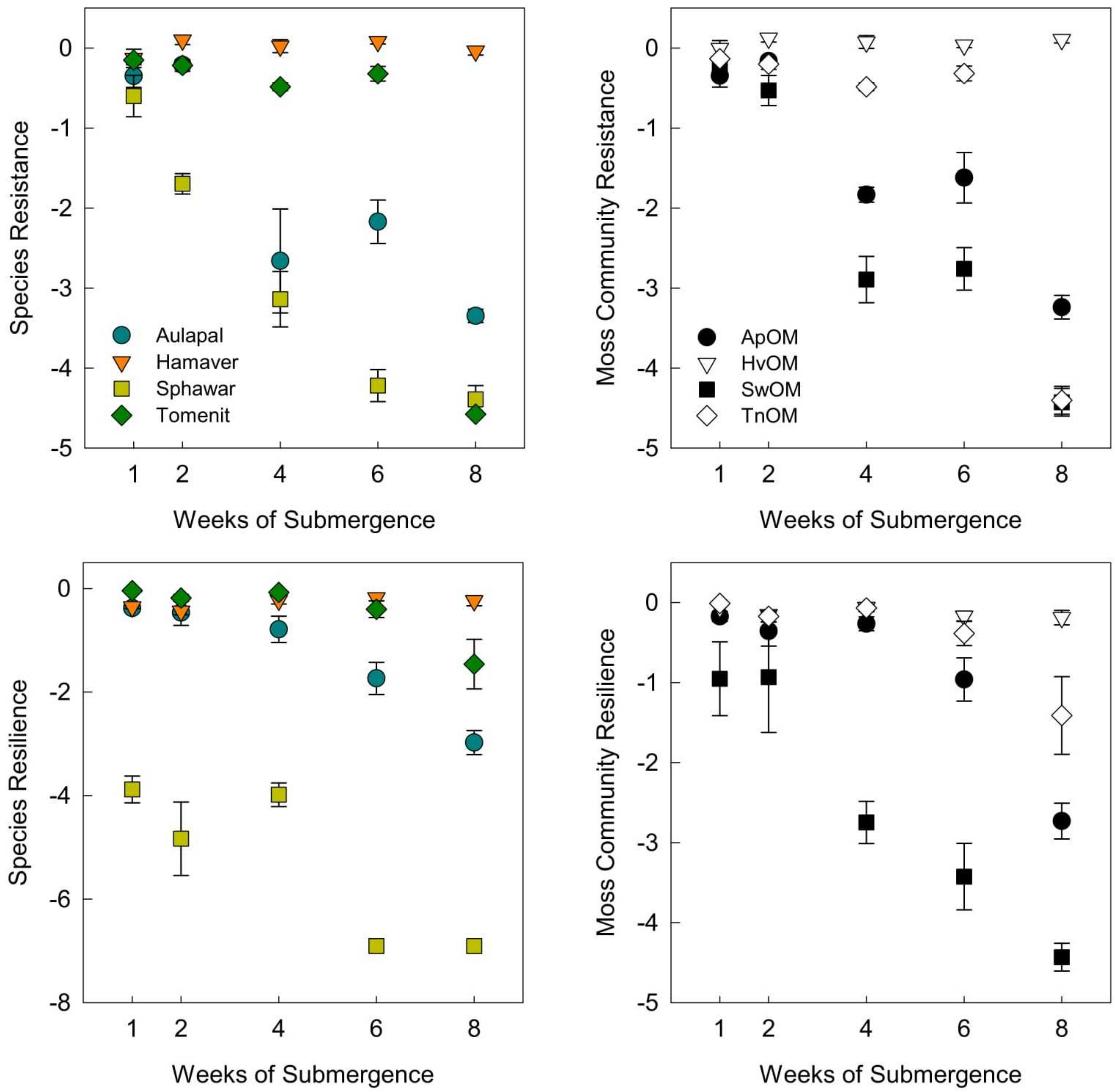

Figure 2.2. Effect of submergence duration on the resistance and resilience of four fen mosses and the moss communities within plugs named to represent the original dominant moss; TnOM= *Tomentypnum nitens* Original Moss, HvOM = *Hamatocaulis vernicosus,* ApOM = *Aulacomnium palustre* Original Moss, and, SwOM= *Sphagnum warnstorfii* Original Moss. Resistance is a log ratio of the short-term (6 weeks after submergence) percent cover response over a non-submergence control, and resilience is the log-ratio of the long-term (11 months after submergence) percent cover response of formerly submerged plugs over the pre-submergence canopy cover. Points represent means values with standard error bars.

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3 METHODS FOR VEGETATION ESTABLISHMENT IN A CONSTRUCTED FEN IN ALBERTA'S OIL SANDS REGION

3.1 INTRODUCTION

Impacts to wetland ecosystems have been a persistent outcome of human land-use. Although avoidance of wetland loss should be prioritized during development, restoration actions are required as compensation where degradation occurs (Government of Canada 1991, Alberta Environment and Sustainable Resource Development 2013). Ecological restoration is the process of assisting the recovery of damaged, degraded, or destroyed ecosystems and its complexity is undeniable (SER 2004). Constraints vary with location, ecosystem type, abiotic and biotic resistances, regulations, and project goals (Suding 2011). Reclamation is like restoration but occurs where there has been a complete loss of ecosystem services and the land surface is recontoured to construct a replacement ecosystem (EPEA 2000, Lima et al. 2016). Reclamation projects have additional constraints as they are typically large in scale, can have novel geochemical conditions, and are conducted in heavily disturbed environments where connectivity to remnants of the pre-existing landscape is limited or nonexistent (Bradshaw and Chadwick 1980). Challenges associated with reclamation of a specific wetland type are particularly applicable in Alberta, Canada, where large-scale disturbances to peatlands are occurring due to resource extraction and standardized reclamation methods do not exist (Foote 2012).

Mining activities in Alberta's oil sands region has resulted in the removal of large areas of boreal forest, of which 50 - 60 % is dominated by peatlands (Vitt et al. 1998, Rooney et al. 2012). Conservative estimates of the net impacts to peatland area from mining activities total approximately 12,000 hectares and no current closure plan proposes to mitigate this loss with peatland reclamation projects (Rooney et al. 2012). Wetland reclamation efforts over the last 30 years have focused on marshes and shallow water wetlands because they are easier to create in a post-mining landscape and were within the regulatory guidelines of restoring to 'equivalent land capacity' (Conservation and Reclamation Regulation 1993, Daly 2011, Alberta Environment 2014). Regulators and operators are now aware that this land-conversion has broad implications of altered structure and function, and efforts have shifted towards developing methods that reconstruct the pre-existing peatlands (Daly et al. 2012, Wytrykush et al. 2012).

Peatlands are challenging wetlands to reclaim as they take hundreds to thousands of years to develop into the mature ecosystems that dominate the region (Halsey et al. 1998). However, recent advances in reclamation approaches and hydrologic modeling suggested that landscape-scale fen peatland creation is possible in a post-mining landscape (Price et al. 2010). This compensation is appropriate as groundwater supplied fens are the most common peatland type in the region (Vitt et al. 1998, Halsey et al. 2003). To comply with a new regulatory framework and test conceptual designs, a watershed supporting a pilot reclamation fen was constructed in an oil sands mine north of Fort McMurray, Alberta (Price et al. 2010, Daly et al. 2012, Ketcheson et al. 2016). The goal was to create a self-sustaining carbon-accumulating fen that supports vegetation typical of natural regional fens (Daly et al. 2012). Here, I present results from a large-scale experiment that was conducted over four years to test various species introduction approaches to establish bryophytes and vascular plants on the constructed fen.

Fen restoration has been conducted in a variety of different regions, most of which has focused on restoring hydrologic regimes and introducing plant species in intact but disturbed sites (Cooper et al. 1998, Cooper et al. 2000, Patterson and Cooper 2007, Mälson and Rydin 2007, Mälson et al. 2010, Cooper et al. 2017, Graf and Rochefort 2008, Graf and Rochefort 2010, Chimner et al. 2016, Andersen et al. 2016, Bess et al. 2014, Cobbaert et al. 2004, Lamers et al. 2015). Challenges of fen reclamation in constructed sites are more complex due to uncertainties in water budgets, surface topography, substrate heterogeneity, diversions from conceptual specifications, and absence of a propagule seed bank (Daly et al. 2012, Ketcheson et al. 2016). Landscapes reclaimed after oil sands mining are also commonly saline because the extraction exhumes marine deposits that are reused for construction (Howat 2000, Daly et al. 2012). Species introduction strategies must consider these constraints and selected species tolerant of anticipated reclamation conditions.

Bryophyte establishment is a key step in reassembling boreal fen communities as they are important peat-formers, provide stability and resilience to disturbance, and are the dominant ground cover

(Turetsky 2003, Turetsky et al. 2012). Mosses have been successfully reestablished in harvested bogs (Bugnon et al. 1997, Gorham and Rochefort 2003, Andersen et al. 2013, González and Rochefort 2014) and hydrologically restored fens (Cobbaert et al. 2004, Graf and Rochefort 2008, Graf and Rochefort 2010, Bess et al. 2014) using the moss layer transfer method (Rochefort et al. 2003, Rochefort and Lode 2006). The moss layer transfer method harvests a peatland's entire surficial propagule bank, which is then spread in the restoration or reclamation area. Previous research has shown that mosses have higher regeneration rates under shade due to microclimate moderation of temperature, light, relative humidity, and soil moisture (Graf and Rochefort 2010, Price et al. 1998). Shade can be provided by mulch, considered essential for *Sphagnum* spp. regeneration (Rochefort et al. 2003), or nurse plants such as the pioneer moss *Polytrichum strictum* or herbaceous plant *Scirpus cyperinus* (Groeneveld et al. 2007, Graf and Rochefort 2010). Microclimate moderation by mulch or planted nurse seedlings is important for *Sphagnum* spp. establishment in bog restoration, but fens have higher water levels and vascular plant cover so the need to implement a shade treatment may not be as critical for true-moss regeneration in reclaimed fens.

Efforts to limit the establishment of species that do not occur in regional fens may also be required to suppress development of an alternate marsh wetland community (Beisner et al. 2003, Suding et al. 2004). Poulin et al. (2012) showed that eight years after bog restoration, *Typha latifolia* cover was higher in restored sites compared to reference bogs and suggested that targeted weeding could prevent invasion. *Typha latifolia* is undesirable in reclaimed fens because it is not a peatland species and can form dense monocultures that reduce diversity and alter ecosystem function (Zedler and Kercher 2004, Shih and Finkelstein 2008, Koropchak and Vitt 2012).

My research is the first to experimentally test multiple plant introduction methods on a constructed fen in a post-mining landscape. This research aims to promote fen bryophyte and vascular plant establishment, reduce *T. latifolia* cover, and establish a diverse assemblage of fen plant species. My objectives were to evaluate bryophyte, vascular plant, and *T. latifolia* percent cover and species diversity over four years in response to planting treatments, wood-strand mulch, weeding, and depth to water level.

I tested the hypotheses that: (1) bryophyte cover would be greatest where the moss layer transfer was used and under wood-strand mulch, (2) vascular plant cover would be highest where the moss layer transfer and seedlings were introduced, (3) *Typha latifolia* cover would be higher in unplanted areas and reduced by a weeding treatment, and (4) bryophyte and vascular plant diversity would be highest where the moss layer transfer was used.

3.1 METHODS

STUDY SITE

This project is located on the Millennium mine lease at Suncor Energy Inc., 35 km north of Fort McMurray, Alberta (56°43′35″N, 111°22′49″W). Average annual precipitation is 419 mm, with approximately 316 mm falling as rain and 134 cm as snow. Growing season (May to September) daily temperatures average 13.3 ° C with 287 mm of precipitation (weather station at 56° 39' N, 111° 13' W, Environment Canada 2018). Growing season precipitation has varied from near-average (2013 = 310 mm, 2014 = 281 mm, and 2016 = 331) to below average (2015 = 203 mm and 2017 = 210 mm). Peak July temperatures exceeded local averages by 3 to 5 degrees (Table A4.1).

Following the design guidelines of Price et al. (2010), construction of the Nikanotee Fen (the Fen) was completed in 2013. The design was based on numerical modelling of climate normals and fen vegetation water requirements (Price and Whitehead 2001). The site was constructed to maintain a range of hydrologic conditions suitable for fen vegetation establishment and peat accumulation (Price et al. 2010). The project includes an upland watershed of tailings material that provides surface and ground water flow into a fen basin constructed of salvaged donor rich fen peat substrate (Price et al. 2010). The peat was derived from horizons that would have no living seeds, roots or rhizomes. The fen construction approach, design, and hydrologic processes are described in Ketcheson et al. (2016).

EXPERIMENTAL DESIGN

My experiment was a two-factor randomized block split-plot design (n = 5 blocks). The blocks are divided into 7 whole-plot factor planting treatments; (1) *Carex aquatilis* seedlings (*Carex*); (2) *Juncus balticus* seedlings (*Juncus*); (3) *C. aquatilis* seedlings and moss layer transfer (MLT + *Carex*); (4) *J. balticus* seedlings and moss layer transfer (MLT +*Juncus*); (5) moss layer transfer (MLT); (6) mixed seeds (Seeds); and (7) unplanted bare-peat control (Unplanted) plots. Each plot is further divided into 4 split-plot mulchweed treatments (mulch/no weed, mulch/weed, no mulch/weed, no mulch/no weed). Whole-plots were 17 X 18 m and split-plots were 8 X 8.5 m. Seed and Unplanted plots had a higher potential to be colonized by invasive species so were designed to be half the size of other treatments plots to limit the overall abundance of invasive species in the Fen (whole-plots were 8 X 18 m and split-plots were 4 X 8.5 m). Planting occurred from mid-June to mid-July, 2013.

I selected two species for planting as seedlings that are regionally common in rich and saline fens. *Carex aquatilis* is among the most common Cyperaceae species found in boreal peatlands, particularly rich fens, across Canada (Gignac et al. 2004). It is an ideal candidate for boreal peatland restoration due to its ability to colonize following disturbance, wide distribution along climatic, water table, pH, electrical conductivity (EC), and substrate type gradients and ease of propagation (Gignac et al. 2004, Koropchak 2010, Mollard et al. 2012). *Juncus balticus* is a rhizomatous species common throughout Canada, occurring in wet meadows and fens (Cooper et al. 2006). It is tolerant of mild to moderate soil salinities, with EC values ranging from 0.1 to 20.1 mS/cm (mean 3.3 mS/cm; Kantrud et al. 1989) and is highly rated for seed availability and establishment potential for alkaline and saline fen reclamation (Ross et al. 2014). Seedlings were grown over the 2012-2013 winter in the nursery and planted at a density of $3/m^2$. In one block, *Carex* was incorrectly planted at $4/m^2$. The plots were combined as analysis showed no statistical differences in response variables between these densities.

Restoration methods typically focus on planting seedlings to ensure rapid establishment, however germination of seeds applied to soils is an alternate method that is less expensive and time consuming than planting seedlings (van der Valk et al. 1999). To determine the germination success and establishment potential, species common in regional saline and rich fens were introduced in a seed mixture. The number of pure live seeds per square meter (pls/m^2) varied by species due to availability and viability. Presented values for seeding rates were calculated estimates based on percent viability determined by TZ test and number of seeds per dry weight. The species mixture included *C. aquatilis* (710 pls/m^2), *Betula pumila* (620 pls/m^2), *Calamagrostis inexpansa* (390 pls/m^2), *Sarracenia purpurea* (290 pls/m^2), *Triglochin martima* (170 pls/m^2*), J. balticus* (110 pls/m^2), and *Vaccinium oxycoccos* (30 pls/m^2). The seeds were locally collected in 2012, stratified in a greenhouse, and sown in a sand/vermiculite mixture with a broadcast seed spreader in July, 2013.

Material for the MLT was collected from a rich fen located 12 km west of the Fen (56° 56' 34" N, 111° 33' 9" W). The site is dominated by regionally abundant bryophyte and vascular species (Chee and Vitt 1989) that have suitable tolerances to oil sands process water (Pouliot et al. 2012, Pouliot et al. 2013) including *Tomentypnum nitens, Aulacomnium palustre, Sphagnum warnstorfii, S. angustifolium, Betula pumila*, and *Carex aquatilis*. From June 19 to 26, 2013, the top 5-10 cm of the donor site was harvested using a large rototiller mounted on an excavator. The MLT was loaded into trucks and delivered and stockpiled at the Fen. The MLT was spread by hand from late-June to mid-July, 2013 at a 1:10 ratio of harvested to donor site (Rochefort et al. 2003).

The application of agricultural straw mulch has facilitated moss regeneration in bogs (Rochefort et al. 2003) but was thought to be less desirable in the Fen as it can be susceptible to displacement by flowing water (Foltz and Dooley 2003). In contrast, wood-strand mulch (WoodStraw® ECM 2012) is more resistant to dispersal and degradation and has been shown to delay runoff and reduce sediment loss from bare soils (Foltz and Dooley 2003, Yanosek et al. 2006). The product is created from low grade debarked and water bathed Douglas fir veneer from British Columbia. To protect establishing bryophytes and reduce non-peatland species invasion, wood-strand mulch was applied to create 90% cover, similar to agricultural straw applications in bog restorations (Rochefort et al. 2003).

Long-term mechanical removal of undesirable species is not viable in most large-scale reclamation settings, but intensive management of young developing stands may be possible in the initial stages to limit establishment. A *T. latifolia* weeding treatment was implemented for the first three years to suppress priority effects and allow desirable fen species to establish and competitively exclude future *T. latifolia* invasions (Lishawa et al. 2017). Emerging *T. latifolia* shoots were clipped at the base and removed from the site 3-4 times throughout the first three growing seasons (2013-2015).

FIELD DATA COLLECTION

Vegetation surveys were conducted in mid-July from 2014 to 2017. Aerial cover was visually estimated to the nearest 1 percent for each species separately in the same 4 m^2 quadrat that was inset 1 m from the edge of each split-plot. My design included 5 replicate blocks, but some plots were not planted because of flooding associated with unexpected site heterogeneity and others were misplanted with *Scirpus microcarpus* instead of *C. aquatilis* because of a delivery error. Sampling effort of split-plots in 2014 (n = 82) was limited because of constraints on survey duration, but efforts increased in 2015 (n = 123), 2016 (n = 121), and 2017 (n = 128) to capture variation in vegetation establishment and water level gradients. This resulted in different samples sizes for certain treatment plots. All treatments were samples with some replication and the percentage of treatments surveyed with 4 or more replicates were 43 % in 2014, 86 % in 2015 and 2016, and 89 % in 2017.

Depth to water level and water pH, electrical conductivity and temperature were measured in mid-July of 2015, 2016, and 2017 in a shallow soil pit dug adjacent to the plot. Absolute positions and elevations (mASL; ± 0.5 cm vertical accuracy) of plots were determined in 2015 using a Leica Geosystems Viva GA14 GNSS RTK GPS system. The Fen was equipped with a grid of wells with Odyssey Capacity Water Level Loggers and Pressure Transducer loggers that recorded daily water levels from March 31, 2013 to September 21, 2017.

I was not able to sample water and soil chemistry in treatment plots each year but did subsample in 2014 and 2015 to provide a snapshot of differences and changes over time in solute concentrations. Water samples were collected from soil pits and soil samples were collected from the top 2 cm of the surface on August 8, 2014 and July 11, 2015 at the Fen and rich fen where the MLT was harvested. Using a Thermo Scientific™ Orion™ Conductivity and Temperature probe, water electrical conductivity (EC) and pH were measured at the time of sampling and soil EC and pH were measured ex-situ in a soil-water extract based on a fixed soil:solution ratio of 1:5 (Dellavalle 1992). Water samples were taken in clean 60 ml high density polyethylene vials, filtered in the lab within 24 hours through 0.45 µm nitrocellulose filters, decanted, and frozen until analysis at the Biotron Experimental Climate Change Research Facility at Western University, London, Ontario, Canada. Major anions and cations analyzed included Cl-, SO_4^{-2}, NO_3^-, Na^+, K^+, Mg^{2+}, and Ca^{2+} (Table A4.2).

SITE CONDITIONS

Season and annual water level variation is presented for three plots that represent a range in water level depths across the Fen in Figure A4.1 with hydrologic parameters in Table A4.3. Water level varied in plots across the Fen throughout the season, and inter-annually between wetter and drier years. Water levels was highest in 2013 after planting and has generally decreased each year. Water levels in the driest plots maintain relatively similar hydrographs despite variation in annual precipitation. Plots with near surface water levels had submergence events where water regularly rose above the ground surface ranging for a total of 40 to 80 days per season.

Average solute concentrations were found to be higher in the soil compared to the water and generally increased from 2014 to 2015. Concentrations in the water and soil at the Fen exceeded levels at the donor rich-fen and ranges observed regional fens (Vitt and Chee 1990, Halsey 2008) but are similar to slightly saline fens and reclaimed boreal communities (Purdy et al. 2005). The EC (avg = 4242 ± 1350 uS/cm) and NA^+ (avg = 398.89 ± 291.57 mg/L in 2015) values are high and concentrations are considered "Fair" for reclamation, but establishment of some fen vascular plant species may be reduced based on the suggested threshold range of 300-600 mg/L for Na^+ (Howat 2000, Koropchak and Vitt 2012). These levels may also limit bryophytes as only a few species inhabit regional saline fens, including *Ptychostomum pseudotriquetrum*, *Campylium stellatum*, and *Drepanocladus aduncus* (Vitt et al. 1993). These mosses and *Sphagnum warnstorfii* and *Tomentypnum nitens* can tolerate high saline levels for up to

100 days of exposure (Pouliot et al. 2013), but the effect of exposure over multiple growing seasons, high fluxes in drier years, and increased accumulation of surface levels over time is unknown (Kessel et al. 2018).

Data Analysis

To evaluate differences between the treatments, I used linear mixed effect models with planting and mulchweed treatments as fixed factors and treatments within replicate blocks as random factors for the split-plot error. To evaluate the effect of mulch on bryophyte cover, comparisons were conducted by averaging across weed treatment split-plots that were determined not to have a significant overall effect (mulch/weed and mulch/no weed vs. no mulch/weed and no mulch/no weed). Weed plots where evaluated similarly by averaging over mulch treatment split-plots to determine the effect of weeding on *T. latifolia* cover as mulch had no significant overall effect (weed/mulch and weed/no mulch vs. no weed/mulch and no weed/no mulch). Satterthwaite's (1946) method was applied to account for unequal sample sizes and variances. I also conducted repeated measures as survey quadrats were placed in the same location within split-plots in each of the four years. Repeated measures mixed models were run to determine difference across years using a 1st order ante-dependence covariance structure that was selected by AIC model comparison. A Tukey-Kramer adjusted least squares means test was applied to determine statistical significance (α=0.05) between the marginal means of unequal sample sizes. I also evaluated the effect of depth to water level in each planting treatment using a mixed effects model with depth to water level as a random variable and treatments within replicate blocks as random factors for the split-plot error. Linear mixed effects model analyses were performed in SAS using proc MIXED (Version 9.3, Cary, NC, USA).

3.2 Results

Bryophyte Cover

Bryophytes established in all experimental plots, at first from the MLT treatment and then through spontaneous colonization over time either via dispersal from MLT plots or from indigenous sources outside the Fen (Figure 3.1). Bryophyte cover was affected by planting treatment from 2014 to 2017 and by mulchweed treatment in 2014 (Table A4.4). In 2014, when averaged across weed treatments, bryophyte cover in unmulched plots differed by planting treatments ($F_{6,31} = 2.6$, $p = 0.036$) but multiple comparison estimates were more conservative and differences between treatments were not observed. Planting treatment in mulched plots did affect bryophyte cover in 2014 ($F_{6,29} = 28.6$, $p < 0.001$), with bryophytes only occurring in MLT + *Juncus*, MLT + *Carex*, and MLT plots. In 2015, the mulch treatment no longer affected bryophyte cover ($F_{3,103} = 0.6$, $p = 0.454$) but planting treatment was still significant ($F_{6,106} = 32.3$, $p < 0.001$) and bryophyte cover was again highest in MLT + *Juncus*, MLT + *Carex*, and MLT plots. In 2016, bryophyte cover was also affected by planting treatment ($F_{6,23} = 6.8$, $p < 0.001$) but changes in the direction of effect occurred. Bryophyte cover in unmulched MLT plots was no longer similar to the MLT + *Juncus* plots and instead had as much bryophyte cover as Unplanted plots. In mulched plots, bryophyte cover in MLT and MLT + *Carex* was similar to MLT + *Juncus* but also did not differ from plots that did not received the MLT. In contrast, bryophyte cover in unmulched *Juncus* and Seed plots became similar to MLT + *Juncus* and MLT + *Carex* plots. In 2017, bryophyte cover continued to differ between planting treatments ($F_{6,26} = 5.3$, $p = 0.001$). In unmulched plots, bryophyte cover was again higher in MLT + *Juncus* plots and lower in MLT and Unplanted plots. In mulched plots, bryophyte cover was higher in MLT + *Juncus* plots and lower in *Carex*, Seed, and Unplanted plots.

In unmulched plots, bryophyte cover changed over the four years in all planting treatments except *Carex* ($F_{3,196} = 2.3$, $p = 0.075$) and Unplanted plots ($F_{3,175} = 2.0$, $p = 0.113$). Bryophyte cover in MLT + *Juncus* plots peaked in 2015, declined in 2016, but recovered in 2017. Bryophyte cover also peaked in 2015 and declined in 2016 in MLT and MLT + *Carex* plots but then remained low in 2017 with covers similar to 2014. Bryophyte cover increased overtime in Seed and *Juncus* plots, with higher values from 2015 to 2017 compared to 2014 in Seed plots, and from 2016 to 2017 compared to 2014 and 2015 in *Juncus* plots.

In mulched plots, bryophyte cover remained unchanged over the four years in all planting treatments except MLT + *Carex* ($F_{3,193} = 6.1$, $p < 0.001$) and *Juncus* plots ($F_{3,192} = 3.3$, $p = 0.021$).

Bryophyte cover in MLT + *Carex* was highest in 2015, declined in 2016 and remained low 2017. In contrast, bryophyte cover in *Juncus* plots increased over time and was higher in 2017 compared to 2014.

Bryophyte cover declined as the water level became closer to the surface in certain planting treatments from 2015 to 2017 (Figure 3.2, Table A4.5). Bryophyte cover was affected by water level in *Juncus* plots in all years, in MLT + *Juncus* plots in 2015 and 2016, in *Carex* plots in 2015 and 2017, and in MLT + *Carex* plots in 2017.

VASCULAR PLANT COVER

Vascular plant cover varied in response to planting treatment from 2014 to 2017 and was not affected by mulchweed treatments (Figure 3.3, Table A4.6). In 2014, vascular plant cover in MLT plots was higher than all other planting treatments except MLT + *Carex* ($F_{3,15} = 4.5$, $p = 0.009$). In 2015, vascular plant cover was higher in MLT and MLT + *Carex* plots compared to Unplanted, *Carex* and *Juncus* plots ($F_{6,25} = 4.2$, $p = 0.005$). In 2016, vascular plant cover was highest in MLT, MLT + *Juncus* and Seed plots and lowest in *Carex* and *Juncus* ($F_{6,22} = 8.3$, $p = 0.023$). Vascular plant cover in MLT + *Carex* plots declined and became similar to *Carex* and *Juncus* plots, whereas cover in Seed plots increased and became comparable to MLT and MLT + *Juncus*. In 2017, vascular plant cover was highest in MLT + *Juncus* plots and lowest in *Carex* plots ($F_{6,22} = 8.3$, $p = 0.023$).

Vascular plant cover changed over time in all treatments and mostly increased from 2014 to 2016. Responses were more variable from 2016 to 2017 as vascular plant cover declined in Unplanted and Seed plots, increased in MLT + *Juncus* plots, and remained in unchanged in other treatments. Vascular plant cover is additive due to the vertical layering of species canopies, so the lack of change or a decline in cover may represent no new recruitment or loss of species. Conversely, an increase in cover could be due to an increase in the number of contributing species. Vascular plant cover was only influenced by water level in MLT + *Carex* plots in 2015 and 2016 with decreasing covers at near surface water levels (Figure A4.2, Table A4.5).

Carex aquatilis percent varied by planting treatment in all years and increase over time in all treatments except MLT + Carex ($F_{3,198} = 0.8$, $p = 0.503$) (Figure 3.4A, Table A4.6). By 2017, all treatments, except in *Juncus* plots, has similar cover of *C. aquatilis* and was higher than any other species, ranging from 40 % to 80 % cover in treatment plots. *Juncus balticus* cover was affected by planting treatment in all years and by mulchweed in 2016 and 2017 (Figure 3.4B, Table A4.6). *Juncus balticus* cover increased in *Juncus* ($F_{3,124} = 8.2$, $p < 0.001$) and Seed plots ($F_{3,137} = 21.5$, $p < 0.001$) from 2014 to 2017, and increased in Unplanted plots ($F_{3,142} = 3.7$, $p = 0.014$) from 2014 to 2016 but then declined in 2017 to similar cover values as 2014. Average percent cover of *J. balticus* in 2017 was 55 % in plots where it was planted as seedlings and 17 % cover (se = 2.0) in Seed plots. *Carex aquatilis* cover in MLT and *Carex* plots increased where the water level was closer to the surface in 2016 and 2017, whereas cover decreased where the water level was closer to the surface in Seed plots in 2016 (Table A4.5). *Juncus balticus* cover was only affected by water level in *Carex* plots in 2015 and 2017, where it declined at near surface water levels, perhaps due to the increase in *C. aquatilis* cover (Table A4.5).

The other introduced species did not establish as well (data not shown). *Calamagrostis inexpansa* failed to germinate in Seed plots but did reach an average of 7 % cover (se = 1.7) by 2017 in MLT plots. *Triglochin martima* successfully germinated in Seed plots, averaging 30 % cover (se = 2.5) in 2014, but declined to 17 % cover (se = 2.3) by 2017. *Betula pumila, Sarracenia purpurea* and *Vaccinium oxycoccos* did not establish in Seed plots but *B. pumila* and *V. oxycoccos* occurred sporadically (< 1 % cover) in MLT and MLT + *Juncus* plots from 2015 to 2017.

TYPHA LATIFOLIA *COVER*

Typha latifolia cover was affected by planting treatment in 2016 and mulchweed treatment from 2015 to 2015 (Table A4.7). In unweeded treatments, cover of *T. latifolia* varied across planting treatments from 2014 to 2016 (Figure 3.5). Unplanted unweeded plots ($F_{3,271} = 31.9$, $p < 0.001$) had the highest cover of *T. latifolia* from 2014 to 2016, followed by Seed plots ($F_{3,262} = 5.6$, $p = 0.001$) from 2014 to 2016 and MLT plots ($F_{3,282} = 4.0$, $p = 0.008$) from 2014 to 2015. In unweeded Unplanted and Seed plots, *T. latifolia* cover increased from 2014 to 2016, but declined in 2017 to the same cover as 2014. The reason for the change may be related to water availability as annual precipitation and water levels were high in 2016 when *T. latifolia* increased and were lower in 2017 when *T. latifolia* declined (Table A4.1, Figure A4.1). *Typha latifolia* increased where water levels were closer to the surface in MLT + *Carex* plots in 2015 and then in Unplanted, MLT and MLT + *Carex* plots in 2016 (Table A4.8). The effect of water level was greatest in 2017 where *T. latifolia* cover increased at near surface water levels in Unplanted, MLT + *Carex*, MLT + *Juncus*, and Seed plots.

DIVERSITY

Bryophyte species richness was affected by planting treatment from 2014 to 2017 and the mulchweed treatment in 2014 and 2016 (Table A4.9). Bryophyte species richness was highest in MLT + *Juncus,* MLT + *Carex* and MLT plots in 2015 and 2016 but declined in MLT plots by 2017 to values similar to Unplanted plots (Figure 3.7A). By 2017, bryophyte species richness in *Juncus* plots increased significantly and was similar to MLT + *Juncus* plots. Bryophyte evenness was affected by planting treatment in all years but multiple comparison estimates were not computed for 2014 due to small sample sizes because bryophyte cover was so low (Table A4.9). From 2015 to 2016, evenness was highest in Unplanted, *Juncus, Carex*, and Seed plots because only a few common bryophytes colonized these areas (Figure 3.7B). Conversely, bryophyte evenness was lowest in MLT, MLT + *Carex* and MLT + *Juncus* plots because of increased number of species propagules in the MLT. By 2017, significant differences in bryophyte evenness occurred only between Unplanted and MLT + *Juncus* plots as species were gained and lost in treatments over the years and evenness converged.

Bryophyte species richness increased in MLT plots in 2015 at near surface water levels, but then decreased at near surface water levels in MLT + *Juncus* and MLT + *Carex* plots in 2016, and MLT + *Carex*, *Juncus*, Seed, and Unplanted plots in 2017 (Figures A4.3, Table A.10). The effect of water level on bryophyte evenness was similar to species richness but with inverse relationships, decreasing in MLT plots in 2015, and increased in MLT + Carex and MLT + Juncus plots in 2016 and in *Carex* and *Juncus* plots in 2017 (Figure A4.4, Table A.10).

Vascular plant species richness was affected by planting treatment and mulchweed in all years (Table A4.9). Across mulchweed treatments, vascular species richness was highest in MLT plots and lowest in *Carex* and *Juncus* plots from 2014 to 2016 (Figure 3.7C). Vascular plant species richness changed over time in all treatments except for *Carex* ($F_{3,89} = 2.3$, $p = 0.084$) and *Juncus* plots ($F_{3,89} = 2.2$, $p = 0.097$). Vascular plant species richness declined in MLT, Unplanted, and Seed plots, increased in MLT + *Juncus*, and was variable was remained unchanged between 2014 and 2017 in MLT + *Carex*.

Vascular plant evenness was affected by planting treatment in all years (Table A4.9) and was highest in Unplanted and Seed plots from 2014 and 2015 and then in Seed and *Juncus* plots by 2017 (Figure 3.7D). Evenness remained unchanged in most treatments from 2014 to 2017, except in *Juncus* ($F_{3,131} = 13.1$, $p < 0.001$) and *Carex* plots ($F_{3,131} = 9.2$, $p < 0.001$), which increased in 2016, and MLT plots ($F_{3,171} = 2.8$, $p = 0.041$) that increased slightly in 2015 but was similar in 2017 to 2014 values.

Vascular plant species richness declined at near surface water levels in all years and in many treatments (Figure A4.5, Table A4.10). Vascular plant species richness was affected by water level from 2015 to 2017 in MLT + *Carex* and MLT + *Juncus* plots, and to a greater extent in 2017 in all treatments except MLT and *Juncus*. Vascular plant evenness had the opposite relationship with increasing evenness at near surface water tables in MLT plots in 2015 and in Seed plots in 2016. In 2017, vascular plant evenness decreased at near surface water levels in MLT and increased at near surface water levels in Seed, MLT + *Carex*, and MLT + *Juncus* plots (Figures A4.6, Table A4.10).

3.4 Discussion

My research demonstrates that it is possible to establish fen bryophyte and vascular species in a constructed boreal fen. The moss layer transfer and *Juncus balticus* seedling planting treatment (MLT + *Juncus*) was most effective at maintaining bryophyte and vascular plant cover and species richness and excluding *Typha latifolia*. Plots with MLT initially supported the highest bryophyte and vascular plant richness and may have acted as source populations for bryophyte dispersal to areas that did not receive the treatment. Over the four years of this study, bryophyte richness increased in many treatments whereas vascular plant richness declined or remained unchanged. The most successful vascular fen species to establish was *Carex aquatilis*, by both seed and seedling, and it spread into all planted and unplanted plots. Of the other introduced fen species, *Juncus balticus* was most effectively introduced as seedlings, *Triglochin maritima* by seed, and *Calamagrostis inexpansa, Betula pumila,* and *Vaccinium oxycoccos* by propagules in the MLT. Within the water level gradient of our plots (+1 cm to -42 cm from 2014 to 2017), weeding *T. latifolia* reduced its cover in the highly affected Unplanted and Seed plots, but was not necessary in other treatments where it was likely competitively excluded by planted species. Wood-strand mulch did not affect bryophyte cover beyond the first year and the effect of water level increased in importance for many measured variables each year. At near surface water levels, there was lower cover and richness of bryophytes and vascular plants, and higher *T. latifolia* cover.

Bryophytes have been successfully established in peatland restoration sites using similar MLT methods (Rochefort et al. 2003). Unlike the recommended straw mulch that improves establishment for boreal bog restoration (Price et al. 1998), I found that wood-strand mulch did not affect fen bryophyte establishment from the MLT over time and also reduced spontaneous colonization to new areas that did not receive the MLT. Straw mulch provides the shade and microclimate moderation essential for *Sphagnum* spp. regeneration in bogs (Price et al. 1998, Rochefort et al. 2003). However, the use of straw mulch in fen restoration projects has had either a negative or no effect on fen species or bryophytes cover (Cobbaert et al. 2004, Graf and Rochefort 2008, Graf and Rochefort 2010, Bess et al. 2014). In our reclaimed fen, high soil moisture is maintained by the near surface water tables (Scarlett et al. 2017) and shade was provided by the abundance of vascular plants that emerge from the MLT, likely resulting in the wood-strand mulch providing little additional benefit. As I observed in unmulched *Juncus* plots, mosses have also successfully established without being introduced or with a mulch treatment in suitably wet and shady sedge communities in restored montane fens (Cooper et al. 2017). In addition, wood decomposes at a slower rate than straw and could have altered soil surface C:N ratios or deterred recruitment of species with a preference for peat substrates. Based on these results, I do not recommend using wood-strand mulch at the tested application density to support bryophyte establishment in future fen reclamation projects, however further experimentation could improve our understanding of situations in which it could provide a benefit. I recommend using the MLT to facilitate rapid colonization and increase diversity of bryophytes and found that *J. balticus* functioning as a nurse plant was most effective at supporting bryophyte cover. This is similar to responses achieved with *Scirpus cyperinus* in bog restoration experiments (Graf and Rochefort 2010).

Contrary to results from other field experiments (Bugnon et al. 1997, Mälson and Rydin 2007, Graf and Rochefort 2010), a near surface water table at the Fen was less favorable for bryophyte establishment and richness. This is likely due to greater annual and season variation with the water table regularly rising above the ground surface that produced submergence events ranging from 40 to 80 days per season over the four years. Previous research has shown that bryophyte species vary in their tolerances to submergence and frequent or reoccurring events severely limited bryophyte persistence and richness (Borkenhagen and Cooper 2018). Bryophytes also vary in their tolerances to different water chemistries (Vicherová et al. 2015) and soil and water salinity levels at the Fen exceeded conditions and differed in chemical composition to the rich-fen donor site where the MLT material was harvested. These novel conditions favored tolerant species such as *Ptychostomum pseudotriquetrum* and likely limited the survival of intolerant species present in the MLT such as *Sphagnum warnstorfii* (Poulin et al. 2013).

Vascular plants rapidly spread across the Fen, established more rapidly than bryophytes, and were successfully introduced by seeds, seedlings, and MLT. *Carex aquatilis* was highly successful across the entire hydrologic gradient, readily established by several introduction methods, and produced more above ground biomass than any other species (Messner et al. in prep). *Juncus balticus* was more modest in its growth and dispersal from where it was planted but supported more bryophyte cover and species richness. Although not as regionally common as moss-dominated fens, herbaceous fens containing *C. aquatilis* or *J. balticus* naturally exist and may be appropriate reference sites for constructed fens in Alberta's oil sands region (Vitt et al. 1998, Halsey 2008, Environment and Parks 2017). *Carex aquatilis* has also successfully colonized the reclaimed Syncrude Fen (Vitt et al. 2016) and been proposed as an ideal species based on its wide tolerances of water table depths and salinity (Koropchak 2010). Despite the advantages in a reclaimed setting, *C. aquatilis* is highly successful, and its height, dense litter production, and high density of culms suppressed bryophytes and other fen vascular plants. Depending on the goals of the project its dominance could be desirable or undesirable. If biomass production and carbon sequestration are the goals, then *C. aquatilis* is an excellent species to propagate. If diversity and bryophyte establishment are targets, then approaches that limit *C. aquatilis* abundance and support *J. balticus* communities should be considered.

Typha latifolia was suppressed by weeding but also by the MLT and seedling planting treatments. This is a key finding because mechanical weeding of an undesirable species is not feasible on large-scale project sites. *Typha latifolia* is a ubiquitous marsh plant in our study region that is difficult to exclude from restored wetlands due to its prolific seed production, dispersal, good germination, rapid growth of rhizomes and competitive tall growth (Shih and Finkelstein 2008). Poulin et al. (2012) found that *T. latifolia* densely colonized wetter areas for years after bog restoration. Vascular plant cover in the Fen exceeded that typically found in bog restorations (Rochefort et al. 2013), suggesting that *T. latifolia* exclusion may be due to competition with other fen plants. Exclusion is particularly important in highly susceptible zones and targeted planting of *C. aquatilis,* which is also adapted to shallow standing water, is likely an effective approach to suppress *T. latifolia* stands in ponded areas of future reclamation projects (Funk et al. 2008). *Typha latifolia* cover in the Fen was highest in wetter areas, but also increased during a wet year in 2016 and declined during a dry year in 2017. This suggests that competition with planted vascular plants may limit invasion but interannual climate and water availability variability likely also have an effect.

I observed an increase in bryophyte richness in some plots and convergence of evenness over time as new species were observed each year and some common species proliferated. The most common species in 2017 was *Ptychostomum pseudotriquetrum*, which occurred in all treatment plots and had the highest cover under *J. balticus*. *Ptychostomum pseudotriquetrum* is a ubiquitous fen moss with a slender and unbranched erect form and grows intermixed with other mosses (BFNA 2014). Despite producing lower biomass than other fen species, *P. pseudotriquetrum* may be a key pioneer species because of its tolerance of saline conditions and elevated elemental concentrations conditions (Vitt and Chee 1990, Pouliot et al. 2012, Pouliot et al. 2013, Vitt and House 2015). The donor site was selected for the MLT in part because it contained *Tomentypnum nitens* and *Sphagnum warnstorfii*, shown to also be tolerant of elevated salinity levels (Pouliot et al. 2013). *Tomentypnum nitens* initially propagated from the MLT but persisted primarily under *J. balticus* and generally declined in abundance each year. *Sphagnum warnstorfii* was never observed in the Fen, likely due to its low tolerance to high salt concentrations and submergence duration (Pouliot et al. 2013, Borkenhagen and Cooper 2018). Bryophytes that established and increased in occurrence over time across the Fen included *Calliergon giganteum, Campylium stellatum*, *Drepanocladus aduncus*, *Drepanocladus polygamous, Leptobryum pyriforme*, and *Funaria hygrometrica,* common fen or ruderal species tolerant of elevated salinity levels and drier or wetter fen conditions (Vitt et al. 1993, Pouliot et al. 2013, BFNA 2014).

Vascular plant species richness generally declined or remained unchanged while evenness was relatively stable in most treatments over the four years. Species known to tolerate elevated salinity levels, such as *Triglochin maritima*, *Triglochin palustre*, and *Calamagrostis inexpansa*, established and persisted in the Fen (Pouliot et al. 2012). In contrast, the occurrence of other wet meadow and fen species, such as *Carex aurea*, *Carex disperma*, *Eleocharis palustris*, *Deschampsia caespitosa*, and *Hierochloe hirta*, peaked in 2015 but then were limited or absent by 2017. The loss of these non-saline or subdominant species was likely influenced by increases in soil and water salinity over time or competition with aggressive species such as *C. aquatilis*, especially in areas with shallow water tables. Identifying biotic and abiotic constraints on establishment is a critical component of species selection for reclamation, however, these constraints may not be initially evident or predictable (Suding 2011, Foote 2012).

Dominance and suppression of fen bryophytes and most vascular plant species by *C. aquatilis* was an unintended outcome in the Fen. Compared to the rich fen where the MLT was harvested and regional fens references (Chee and Vitt 1989), the water table at the Fen is closer to the surface and the water and soil have higher salinity. Abiotic filters and species functional traits determine whether a species can colonize a habitat, and biotic interaction filters can either constrain or facilitate a species persistence within the community (Keddy 1992, Keddy 1999, Weiher et al. 2011). Reclaimed sites typically have abundant resources and lack pre-existing biotic constraints resulting in novel conditions and interactions as communities assemble. This can lead to dominance by aggressive resource aquisitors and the loss of more uncommon or slower growing species (Doherty and Zedler 2014). The Fen surface was also designed to be flat to retain water (Price et al. 2010), which resulted in a nearly homogeneous surface that may have facilitated *C. aquatilis* expansion compared to natural fens that are typically topographically, hydrologically, and geochemically variable (Chee and Vitt 1989, Sampath et al. 2016).

My research demonstrates the successful introduction of fen bryophyte and vascular plant species within a constructed boreal fen over four years. Bryophyte establishment was highest under *J. balticus*, vascular plants established rapidly, *C. aquatilis* dominated across all treatments, and *T. latifolia* cover was reduced by planted species. Soil and water chemistry and abundance of vascular plants at the Fen are more similar to regional saline fens then the intended target of a moss-dominated rich-fens (Chee and Vitt 1989, Wells and Price 2015). An herbaceous saline fen is still within the range of acceptable outcomes although consideration must be given to potential differences in production and carbon dynamics between these two fen types (Volik et al. 2017). To prioritize plant diversity and bryophyte establishment, future projects should be designed to have more topography and limit areas with mid-summer near surface water levels, select species tolerant of expected abiotic conditions, and introduce co-dominant seedling nurse plants. Methods of plant introduction are important to prioritize, but species selection that consider the characteristics of reclaimed system is critical. These characteristics can be difficult to predict where novel reclamation approaches are used, but an understanding of expected hydrologic and geochemical conditions will provide the bases for predicting the successful establishment of target species.

3.5 FIGURES

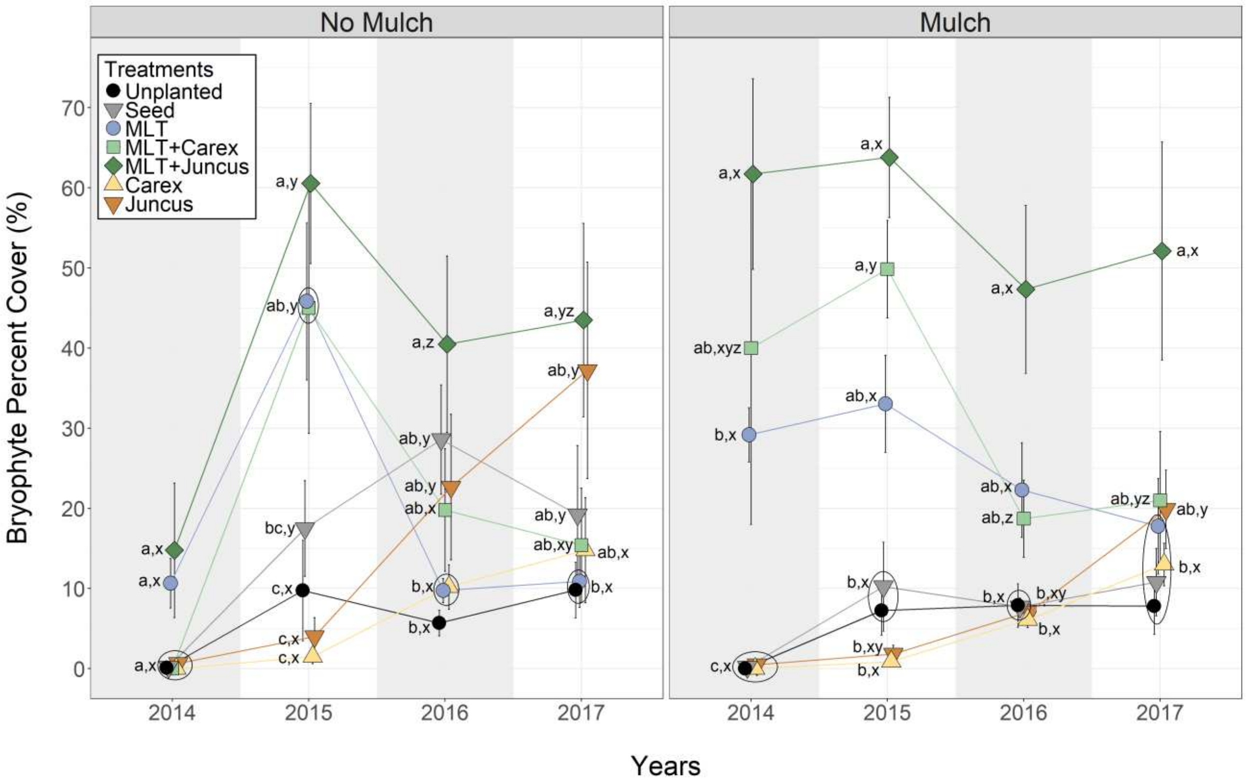

Figure 3.1 Effects of planting treatments on bryophyte percent cover from 2014 to 2017. Bars represent mean ±1 standard error. Differences among treatments within a year indicated by letters a–c, differences over time within a treatment indicated by letters x–z. Means with the same letter (a–c or x–z) were not significantly different (Tukey-Kramer adjusted comparison of least squares means, α=0.05).

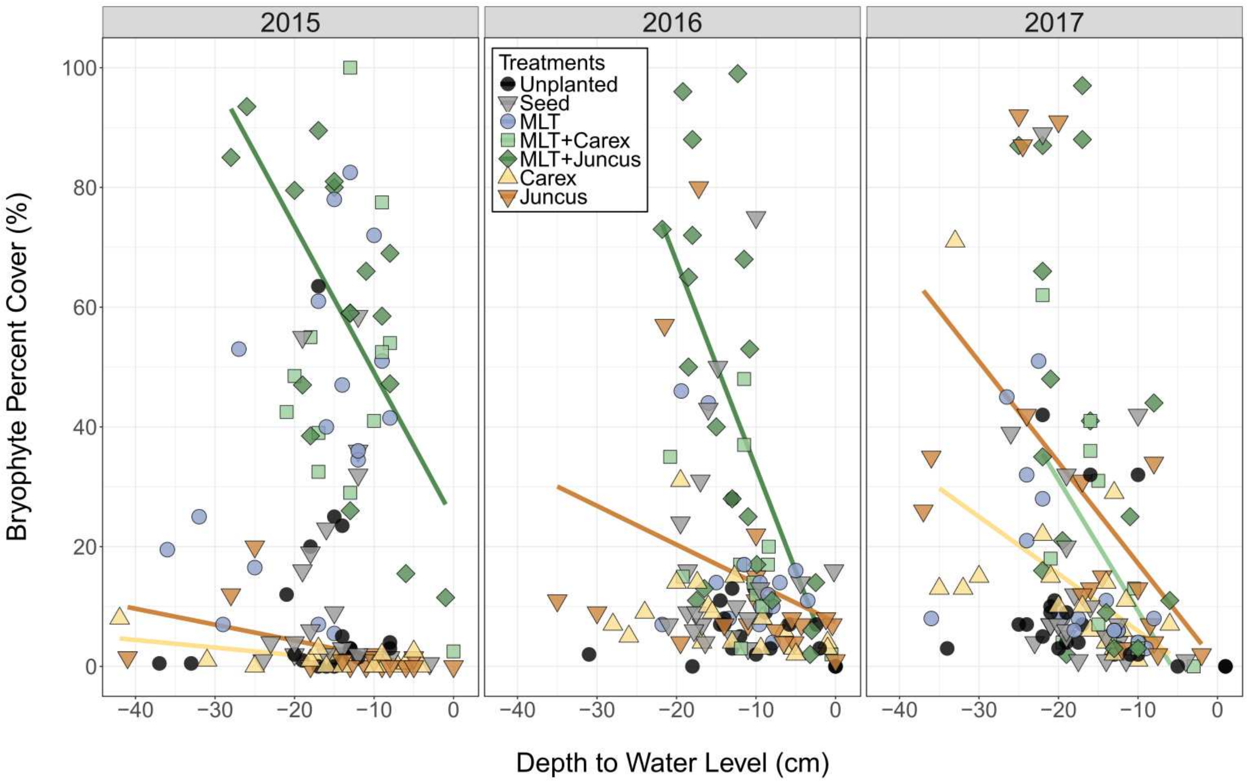

Figure 3.2 Effects of depth to water level and planting treatments on bryophyte percent cover from 2015 to 2017. Significant linear regressions relationships between percent cover and depth to water level are presented for each planting treatment by color.

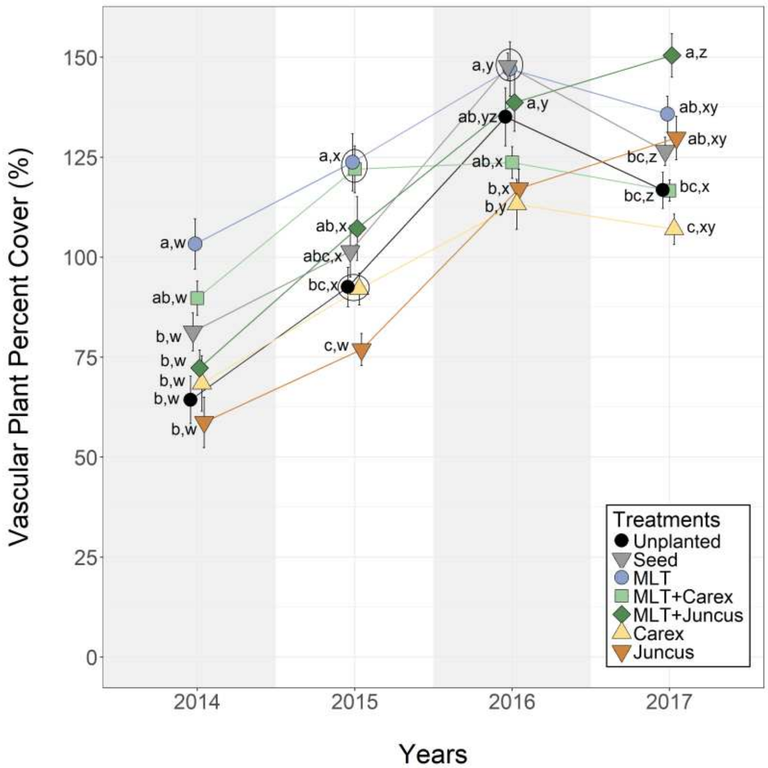

Figure 3.3 Effects of planting treatments on vascular plant percent cover from 2014 to 2017. Bars represent mean values with +1 standard error. Differences among treatments within a year indicated by letters a–c, differences over time within a treatment indicated by letters w–z. Means with the same letter (a–c or w–z) were not significantly different (Tukey-Kramer adjusted comparison of least squares means, α=0.05).

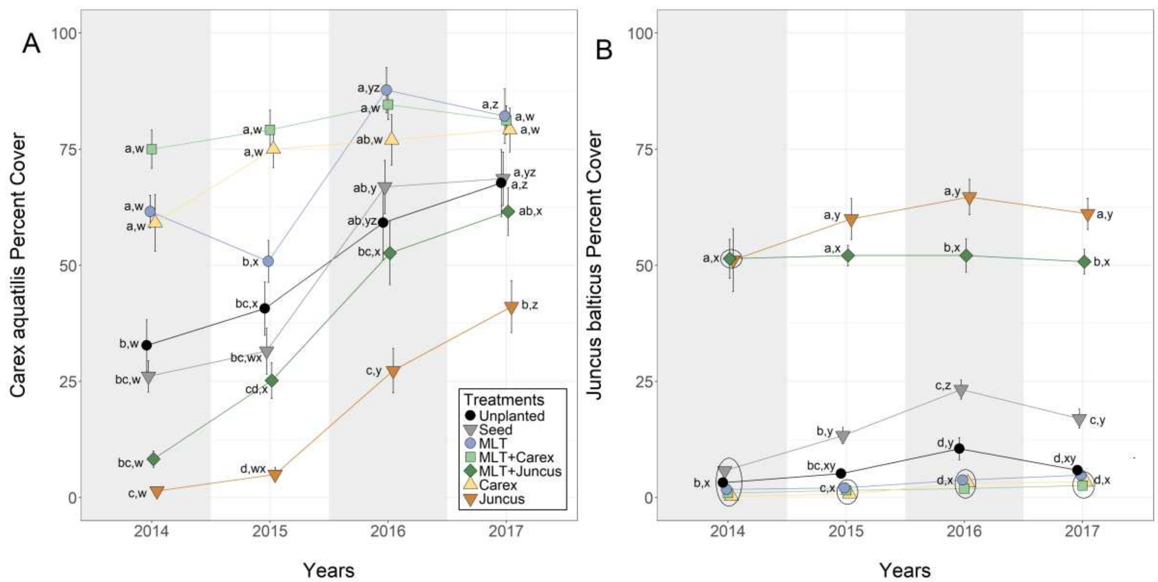

Figure 3.4 Effects of planting treatments on percent cover of *Carex aquatilis* (A) and *Juncus balticus* (B) from 2014 to 2017. Bars represent mean values with ±1 standard error. Differences among treatments within a year indicated by letters a–d, differences over time within a treatment indicated by letters w–z. Means with the same letter (a–d or w–z) were not significantly different (Tukey-Kramer adjusted comparison of least squares means, α=0.05).

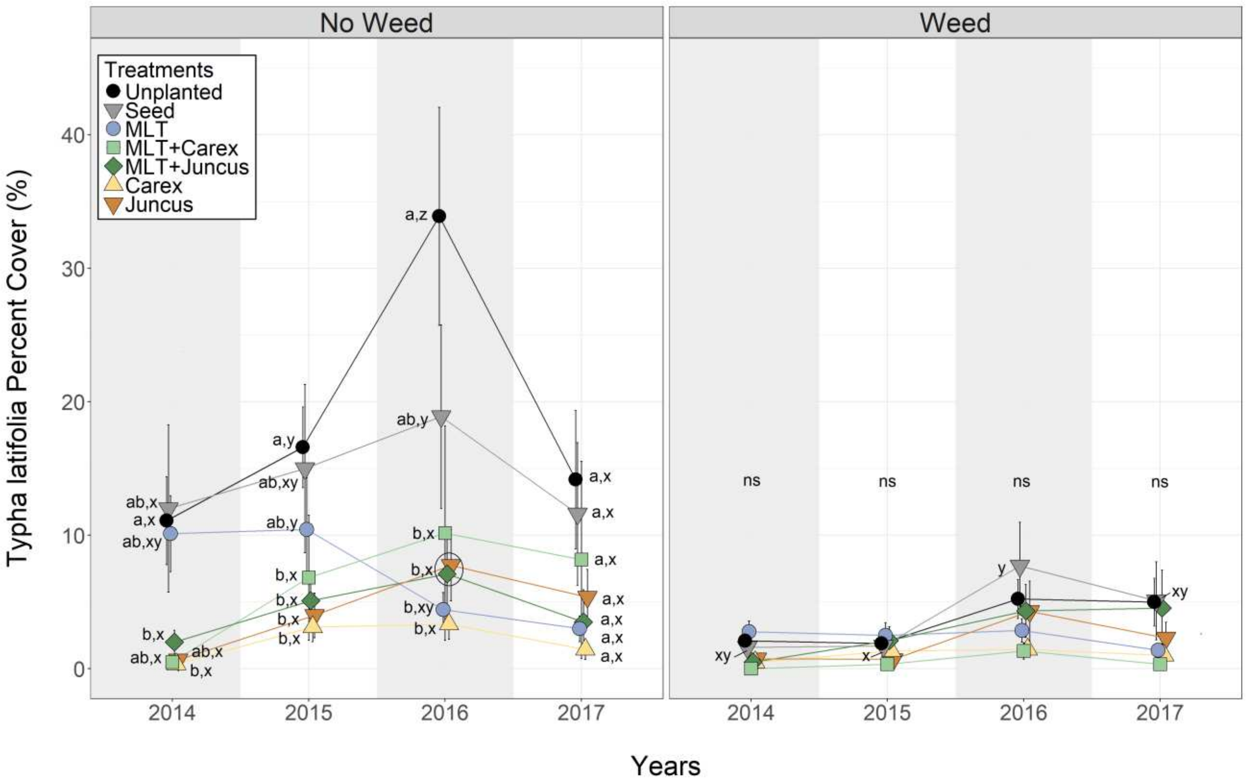

Figure 3.5 Effects of planting and weeding treatments on the percent cover of *Typha latifolia* from 2014 to 2017. Bars represent means ±1 standard error. Differences among treatments within a year indicated by letters a–c, differences over time within a treatment indicated by letters x–z. Means with the same letter (a–c or x–z) were not significantly different (Tukey-Kramer adjusted comparison of least squares means, α=0.05).

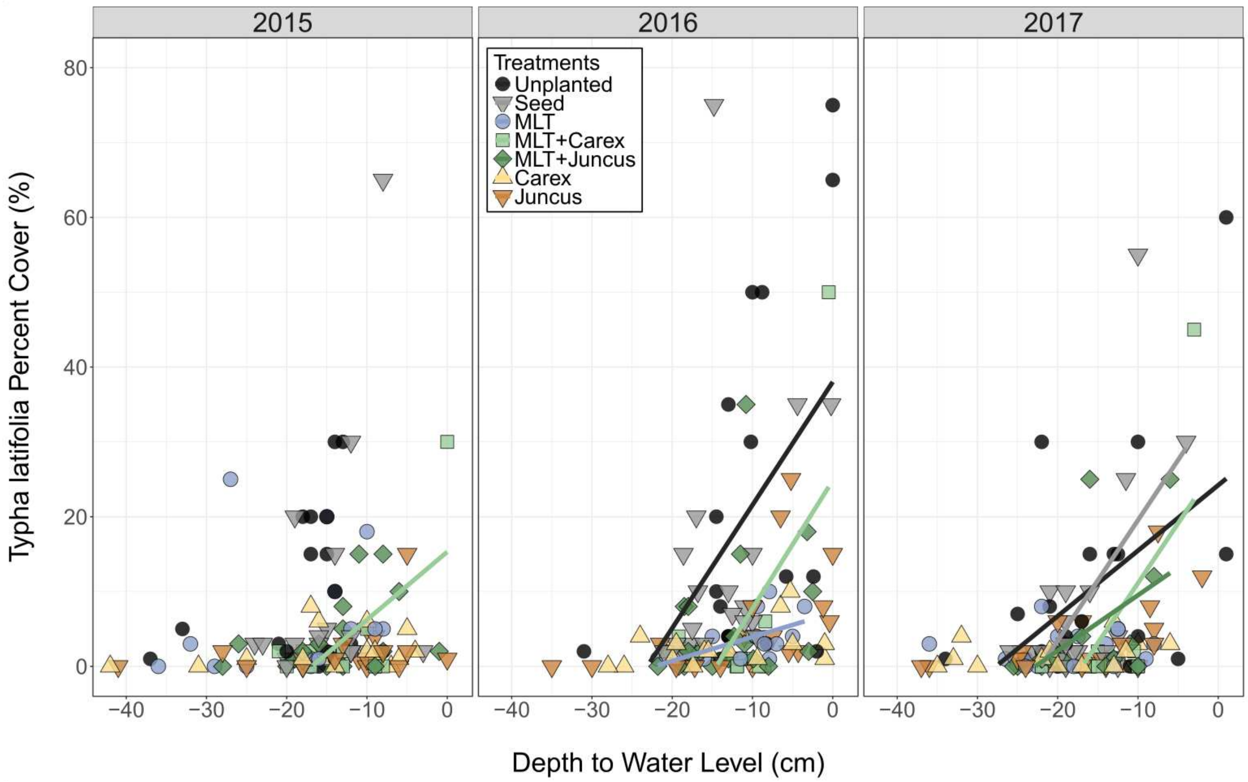

Figure 3.6. Effects of depth to water level and planting treatments on *Typha latifolia* percent cover from 2015 to 2017. Significant linear regressions relationships between percent cover and depth to water level are presented for each planting treatment by color.

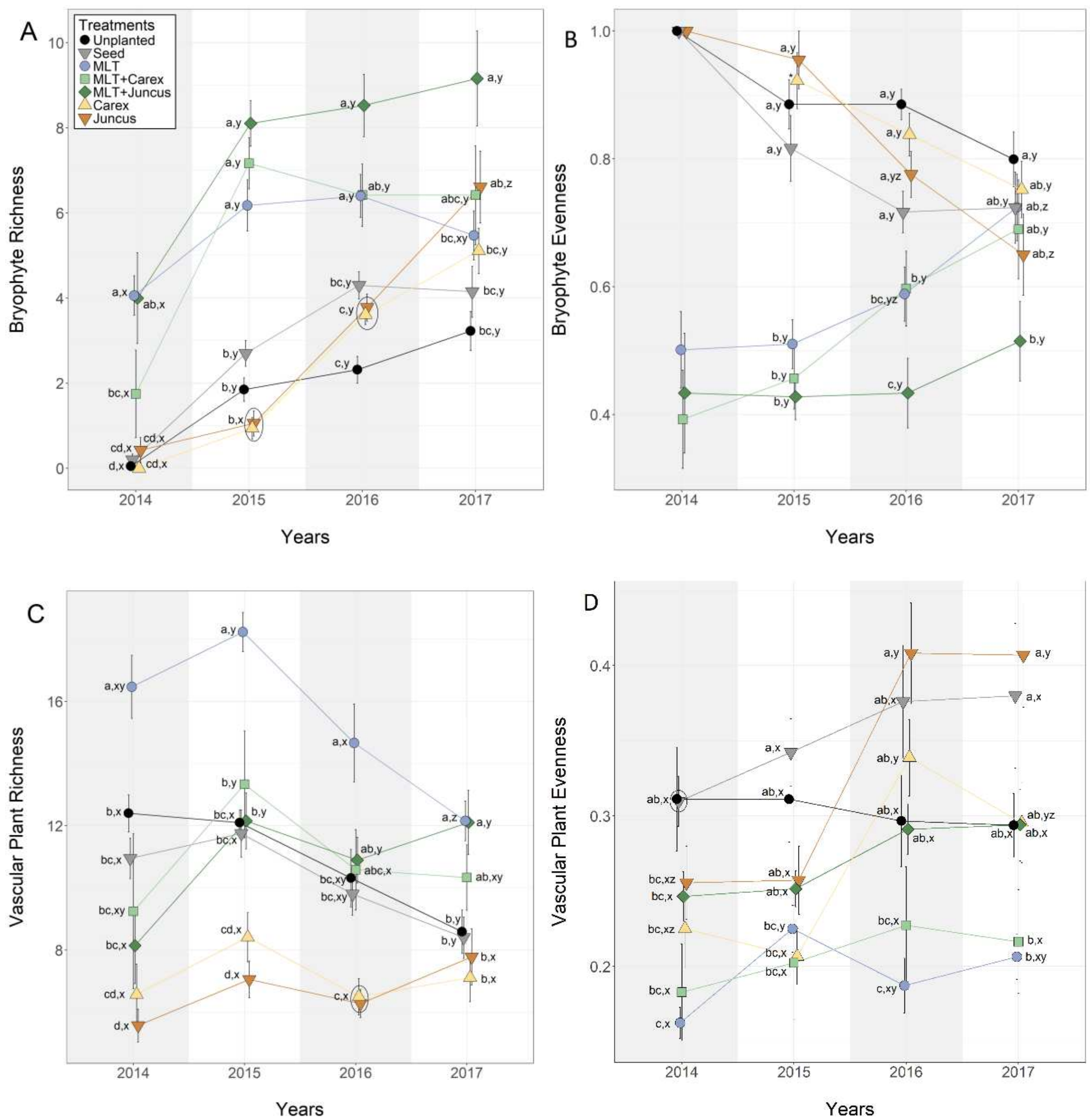

Figure 3.7 Effects of planting treatments on bryophyte (A, B) and vascular plant (C, D) species richness and evenness from 2014 to 2017. Bars represent mean values with ±1 standard error. Differences among treatments within a year indicated by letters a–c, differences over time within a treatment indicated by letters x–z. Means with the same letter (a–c or x–z) were not significantly different (Tukey-Kramer adjusted comparison of least squares means, α=0.05). Statistical differences between treatments in 2014 and *Carex* treatment in 2015 (*) for bryophyte evenness could not be determined because of small sample size.

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4 ESTABLISHMENT OF PEAT-FORMING VEGETATION AFFECTED BY WATER LEVEL AND SPECIES INTRODUCTION APPROACH: A COMPARISON ACROSS TWO CONSTRUCTED FENS IN MINED OIL SANDS LANDSCAPES OF BOREAL CANADA

4.1 INTRODUCTION

Species introduction approaches in restoration and reclamation are typically guided by traditional succession theory where species composition progresses towards a desirable reference ecosystem or defined success criteria (Walker et al. 2007). The target community can either occur spontaneously through passive approaches or be purposefully manipulated by active approaches to assist reassembly (Suding 2011). Passive approaches that depend on natural regeneration are less expensive and laborious, but active approaches can increase the likelihood of success and decrease time to reach the target (Holl and Aide 2011, Prach and Hobbs 2008). Active approaches vary in cost and effort of application from 'basic', such as broadcast seeding, to 'intensive', such as planting seedlings or vegetative propagules. Restoration is the process of assisting the recovery of degraded, damaged or destroyed ecosystems (SER 2004) and passive approaches can be used in resilient ecosystems with modest land-use histories that are connected to remnant reference areas (Holl and Aide 2011). Reclamation typically occurs in heavily disturbed areas where there has been a complete loss of an ecosystem and a replacement is constructed (EPEA 2000, Lima et al. 2016). Active approaches are more appropriate because soils are usually bare and lack propagules of desired species, and there is limited dispersal potential from nearby natural areas (Battaglia et al. 2008, Holl and Aide 2011). Evaluating outcomes after species introduction for reclamation can help inform decisionmakers about when basic approaches are suitable, where intensive actions are required, and whether selected species are appropriate or chosen targets are realistic given site constraints (Holl and Aide 2011, Matthews and Spyreas 2010, Prach and Hobbs 2008). This information is essential to efficiently allocate reclamation resources and maximize successful outcomes (Holl and Aide 2011).

Over time, communities in wetland restoration sites have been shown to follow successional patterns of convergence or divergence, resulting in outcomes that either attain or deviate from the proposed success criteria (Mathews and Spyreas 2010; Figure 4.1). Convergence is when community composition between sites become more similar over time and divergence occurs when community composition between sites becomes more dissimilar (Lepš and Rejmánek 1991). Convergence can occur when abiotic conditions are similar or a highly successful species dominates (Mathew and Spyres 2010). Divergence is commonly a result of environmental gradients or pioneer species that can permanently influence community composition (Prach et al. 2001, del Moral et al. 2007, Fukami et al. 2005, Young et al. 2005). Wetlands are particularly susceptible to invasive species that initially establish and disrupt restoration efforts by having long-term effects on habitat structure, biodiversity, and function (Zedler and Kercher 2004, Trowbridge 2007) Reclaimed wetlands provide a good opportunity to study community development as processes similar to primary succession occur because selected species are introduced to the bare site and biotic and abiotic filters strongly influence outcomes. Evaluating community convergence and divergence across species introduction approaches and whether sites attain or deviate from intended targets will help to identify drivers and refine success criteria.

Reclamation solutions are needed in the oil sands region of Alberta, Canada where large areas of the boreal landscape dominated by fen peatlands are removed by mining activities (Vitt et al. 1998). Reclamation is required to compensate for the loss of fen ecosystems, but projects over the years have focused on constructing marsh wetlands because they are easier to reclaim and often spontaneously form in disturbed basins (CEMA 2014, Daly et al. 2011). Fens are more specific in their hydrologic and geochemical requirements and are thought to require more precise construction designs (Price et al. 2010, Chimner et al. 2016). Regulators and operators are now aware that appropriate mitigation must include fen reclamation, although uncertainty exists on best practices for implementation and appropriate success criteria (Ketcheson et al. 2016). To test the process, two large-scale experimental fens have recently been constructed on oil sands mines north of Fort McMurray, Alberta. The projects were designed to recreate natural regional fen ecosystems within the constraints of a post-mining landscape (Ketcheson et al. 2016).

Although the designs differed, similarities have emerged, and the vegetation data from both sites was synthesized to evaluate responses to environment conditions and species introduction approaches.

The goal of each project was to design a fen with species composition and peat-accumulating processes like regional reference ecosystems. Reference conditions were used to set reclamation targets and evaluate success, but goals are increasingly being broadened to include ecosystem services and resilience to change because of constraints that prevent the return to historical conditions. Because there is high variation in regional reference fens, a range of acceptable targets may exist (Vitt et al. 1998, Halsey 2008). An obvious target includes the most common regional fen type, moderate to rich fens dominated by the mosses *Tomentypnum nitens, Aulacomnium palustre*, and *Sphagnum* species (Vitt et al. 1998, Halsey 2008). Sedge dominated fens are uncommon in the region but may be viable reclamation targets due to the rapid establishment of herbaceous plants and the wide environmental tolerance of many species (Vitt et al. 1998, Halsey 2008). However, because these reclaimed systems are only a few years old, comparing their processes or species composition to ancient reference fens may be misguided. Another practical target could be communities that represent a peatland initiation stage that have been identified at the base of regional peatlands and typically consist of sedges, shrubs and some rich-fen mosses (Bloise 2007, Vitt et al. 2011, Koropchak et al. 2012, Borkenhagen and Cooper 2016, Berube et al. 2017). Alternatively, a species-level approach that targets canopy cover and desirable fen species richness focusses on composition and function without a direct comparison to reference communities that tend to be more complex than reclaimed sites (Environment and Parks 2017).

The Sandhill Fen and Nikanotee Fen are the first two landscape-scale reclamation projects of fens and associated watersheds in the Alberta oil sands region. Designs and species introduction approaches differed yet clear patterns of vegetation establishment have emerged four and five years since project implementation. My goal was to extract the commonalities and determine the most effective approaches to establish peat-forming plants that are representative of regional fen types or stages. In the chapter, I address the following five questions:

1) Which plants are abundant in the reclamation fens and how do these species respond to the water level gradient?
2) Do vegetation communities at the sites converge or diverge and how are they influenced by the species introduction approach and water level gradient?
3) Which communities support bryophytes and desirable fen species cover and species richness?
4) How does water level affect bryophyte cover and desirable fen species richness in each community?
5) How does the ratio of desirable to undesirable species of each community vary in response to the species introduction approach and water level gradient?

4.2 METHODS

STUDY REGION

Sandhill and Nikanotee Fens are located north of Fort McMurray, Alberta, Canada. Reclamation designs were developed based on modeling of groundwater interactions with adjacent landscapes (Pollard et al. 2012), vegetation moisture requirements, and long-term climate data (Price and Whitehead 2001, Price et al. 2010). Average total annual precipitation in this area is 419 mm, mostly occurring as rain from May to September. Daily mean winter temperatures in January are -17.4 º C with average annual snowfall of approximately 134 cm. During the growing season, daily mean temperatures peak in July to 17.1 ºC with a total average annual rainfall of 316 mm (1981-2010 climate normal from weather station at 56° 39' N, 111° 13' W; Environment Canada 2018).

SANDHILL FEN

Sandhill Fen (SF) on the Syncrude Canada Limited's Mildred Lake lease was constructed in an old-oil sands mine that had been filled with consolidated tailings (Syncrude 2008). Starting in 2007, additional tailing sand was used to contour a network of connected upland hummocks, ephemeral channels, and fen basins (main fen and two perched fens) that were capped with a 0.5 m layer of harvested bog and fen peat placed in the winter of 2011 (Wytrykush et al. 2012, Ketcheson et al. 2016, Vitt et al. 2016; Figure A5.1). The entire fen and watershed landscape is 52 hectares in size. Following the Guidelines for Reclamation of Terrestrial Vegetation in the Oil Sands Region, a basic species introduction approach was implemented that involved spreading a native seed mix in November 2011 (Pollard et al. 2012) . Seeds were hand collected during the summer of 2011 from sites dominated by *Carex aquatilis.* Other rich fen species were also locally collected and added to the seed mixture in smaller abundances (1-5 %). A seeding rate is unavailable for SF, but percent contribution of species in the seed mix include *Carex aquatilis* (80 %), *Carex diandra* (5 %), *Carex utriculata* (5 %), *Scirpus atrocinctus* (5 %), *Carex bebbii* (1 %), *Carex paupercula* (1 %), *Scirpus microcarpus* (1 %), *Carex lasiocarpa* (<1 %), *Carex rostrata* (<1 %), *Carex limosa* (<1 %), *Carex interior* (<1 %), and *Juncus tenuis* (<1 %). The main fen was dry until August 2012 when water from a natural lake was supplied from a storage pond through 2013. Since 2014, available water has been maintained by snow melt and precipitation without additional intentional inputs.

Nikanotee Fen

Nikanotee Fen (NF) on the Millennium mine lease at Suncor Energy Inc. oil sands mining operations site was designed to be a single watershed-fen system and constructed on an overburden dump beginning in 2008. The pit was graded to a 3 % slope and covered with a synthetic liner to recreate a groundwater flow towards the fen basin. The upland aquifer was constructed of tailings sand and caped with 20 cm of stockpiled boreal forest floor soil. The base of the fen was lined with a 0.5 m layer of petroleum coke (by-product of extraction) to enhance the connectivity between the upland and fen and evenly distribute the groundwater before it rises through the peat. Fen peat harvested from a new lease area being mined was placed 2 m deep over the coke layer in the winter of 2012 (Daly et al. 2012, Price et al. 2010, Ketcheson et al. 2016). The entire fen and watershed landscape is 35 hectares in size.

Passive and active approaches of species introduction were tested and applied in June and July, 2013 (Figure A5.2). Natural regeneration in unplanted areas was tested as a passive approach, and active efforts included basic approaches of broadcast seeding, and intensive approaches included planted seedlings and spreading moss layer transfer material (MLT). Species introduction approaches were tested in a two-factor randomized split-plot design with 5 replicate blocks. The blocks are divided into 7 whole-block factor species introduction treatment plots: (1) *Carex aquatilis* (*Carex*) seedlings, (2) *Juncus balticus* (*Juncus*) seedlings, (3) *C. aquatilis* seedlings + MLT (Moss *Carex*), (4) *J. balticus* seedlings + MLT (Moss *Juncus*), (5) MLT (Moss), (6) mixed seeds (Seeded), and (7) unplanted bare-peat control (Unplanted). The plots are further divided into split-plots testing a wood-strand mulch cover treatment (WoodStraw® ECM). After four years, vascular plant and bryophyte establishment was not significantly affected by mulch cover so analyses were conducted by averaging across mulch and no-mulch split-plots (Borkenhagen and Cooper unpublished).

Seeds of regionally common fen species were locally collected and stratified by Smoky Lake Forest Nursery in Smoky Lake, Alberta. The seeds were mixed together with sand and vermiculite and sown onto plots with a broadcast spreader. The number of pure live seeds per square meter (pls/m^2) varied by species due to availability and viability. The species mixture included *C. aquatilis* (710 pls/m^2), *Betula pumila* (620 pls/m^2), *Calamagrostis inexpansa* (390 pls/m^2), *Sarracenia purpurea* (290 pls/m^2), *Triglochin martima* (170 pls/m^2), *J. balticus* (110 pls/m^2), and *Vaccinium oxycoccos* (30 pls/m^2) (proportions in Table A5.1). Seedling were grown at the nursery from the same collected seed over the 2012-2013 winter and planted in plots at 3/m^2. In one block, *C. aquatilis* was incorrectly planted at 4/m^2. Analysis showed that there were no statistical differences between these densities and the response variables, so the plots were combined.

The NF design tests the establishment of species introduced by MLT method using donor material collected from a treed rich fen located within 12 km of NF (56° 56' 34" N, 111° 33' 9" W). The site contains regionally abundant vascular plant and bryophyte species (Chee and Vitt 1989) that have suitable tolerances to oil sands process water (Pouliot et al. 2012), including *Tomentypnum nitens, Aulacomnium palustre, Sphagnum warnstorfii, Sphagnum angustifolium, B. pumila*, and *C. aquatilis* (proportions in Table A5.2). In early June 2013, the top 5-10 cm of the donor site was harvested using a large rototiller mounted on an excavator. The MLT material was delivered by truck and spread by hand onto plots from late-June to mid-July, 2013 at a 1:10 ratio of harvested to donor site (Rochefort et al. 2003).

VEGETATION SAMPLING

Surveys of vegetation composition and depth to water level were conducted at SF on July 12-13, 2017 and at NF from July 20-25, 2017. The data represent conditions five years (SF) and four years (NF) after project initiation. Species canopy cover were assessed by visual estimation to the nearest 1 % cover in an 8 m^2 circular plot at SF and a 4 m^2 square plot at NF. Plot sizes differed due to original sampling protocols at each fen but are both within recommended plot sizes for low-growing (bryophyte) and herbaceous vegetation (4 – 16 m^2; Chytrý and Otýpková 2003). Species-area relationships assume that species richness increases with increased sampling area (Connor and McCoy 1979), however species richness was higher at NF in 4 m^2 plots (avg. = 15 species, se = 0.9) than at SF in 8 m^2 plots (avg. = 9 species, se = 0.7). In addition, Otýpková and Chytrý (2006) show that ordinations containing differently sized plots (of less than a factor of four) do not produce patterns associated with plot sizes.

Depth to water level was measured in an open pit dug adjacent to the plot at SF on July 12-13, 2017 and at NF on July 25, 2017. Total precipitation in Fort McMurray during the days between the survey dates was 9 mm (local weather station at 56°39′N, 111°13′W; Environment Canada, 2018).

ANALYSIS

Rank abundance curves were calculated to assess the relative proportion of each species within species introduction approaches (Kindt and Coe 2005, R-Development Core Team 2017). Best fit regressions for species distribution along the water level gradient at each fen were determined by selecting the most parsimonious model with the lowest Akaike information criterion (AIC) value (Akaike 1987, Burnham et al. 2011). Species abundance data were relativized and plots were compared using Bray-Curtis dissimilarity (Bray and Curtis 1957) and analyzed using a non-metric multidimensional scaling ordination (NMDS). A solution for the NMDS ordination was reached in 2 dimensions after 9 iterations with a stress value of 0.16. A permutational multivariate analysis of variance (PERMANOVA; Anderson et al. 2008) was conducted to assess multivariate differences between and within species introduction approaches and in response to water level. Communities were determined using a group average cluster analysis with a cut off level of 40 % similarity, resolved by similarity profile analysis (SIMPROF). A similarity percentages analysis (SIMPER) identified the species contributing most to the similarities and differences between groups. All multivariate analyses were conducted in Primer v6 (Primer-E, Plymouth, UK; Clarke and Gorley 2015).

Species were classified as desirable or undesirable based on their known occurrences in fens in Alberta (synthesized by D.H. Vitt from literature including Chee and Vitt 1989, Environment and Parks 2017, Halsey 2008, Slack et al. 1980, Vitt et al. 1975, Vitt and Chee 1990; Table A5.3). Desirable species have been found in regional fens and consist of both bryophyte and vascular plant species. Undesirable species are upland, marsh or non-natives vascular plants that do not typically occur in regional fens. Recently published reclamation criteria for peatlands in Alberta were used to assess vegetation responses at the fens (Environment and Parks 2017). For the reclamation of fens from bare-soil, the targets include bryophyte cover ≥ 50 %, desirable species cover ≥ 50 %, undesirable species cover ≤ 20 %, and desirable species richness of ≥ 4-9 species depending on fen type (e.g. saline vs. rich fen).

Diversity within communities were calculated as species richness, the total number of species in each plot, and average Bray-Curtis distance from group centroid, presented as a measure of beta diversity (Anderson et al. 2006). I calculated species richness for bryophyte, and desirables and undesirable species. Differences between variables were not determined statistically due to strongly unequal sample sizes within communities (e.g. 5 plots vs. 86 plots).

4.3 RESULTS

SPECIES ABUNDANCE AND RESPONSE TO WATER LEVEL

Water levels in all plots were measured during three days in July, 2017, to determine the water level gradient at each site. The water levels differed between fens, varying from +79 cm (standing water above soil surface) to -50 cm (water level below the soil surface) at SF (avg = 3.7 cm, se = 3.4, n = 79) and from +1 cm to -36 cm at NF (avg = -17.3 cm, se = 0.9, n = 64). Five years after seeding, 70 vascular plant and 24 bryophyte species were found at SF, of which 51 were desirable and 43 undesirable species (Table A5.3). Four years after planting, 47 vascular plant and 20 bryophyte species were found at NF across all treatments, of which 51 were desirable and 16 undesirable species (Table A5.3).

The most abundant species in both fens was *Carex aquatilis* (Figure 4.2, Tables A5.4 and A5.5). Initially introduced by seed and a few seedlings at SF, and by seed, seedling, and in the MLT material at NF (Seeded, Carex, Moss, and Moss Carex plots), it has spread and established throughout both fens. *Carex aquatilis* was observed in 75 % of SF plots, is dominant in all but the *Juncus* plots at NF and occurred along the entire water level range (-49 cm to +79 cm) with increasing cover in intermediate water levels (Figure 4.3A and 4.3B). Areas with standing water in both fens were also colonized by *Typha latifolia* whose abundance had a log or 2nd degree polynomial relationship to water level, increasing from about -10 cm to +79 cm (Figure 4.3A and 4.3B). Even though it was not introduced, the most abundant species in drier areas at SF was *Calamagrostis canadensis*. Its distribution was linearly related to water level with higher cover in sites where the water level was below the soil surface (Figure 4.3A). At NF, *J. balticus* abundance was not related to water level (Figure 4.3B). The other abundant species had 2nd degree polynomial relationships to water level, with either increasing cover in drier areas (*Sonchus arvensis* and *Ptychostomum pseudotriquetrum* at SF) or at intermediate water levels (*Carex utriculata* and *Ptychostomum pseudotriquetrum* at NF) (Figure 4.3A and 4.3B).

Species abundance was also related to introduction approach and/or fecundity. For example, *Juncus balticus* was introduced at NF by seed and seedling (Seeded, *Juncus*, and Moss *Juncus* plots) and remained dominant in the *Juncus* plots but was overgrown by *C. aquatilis* in the Moss *Juncus* plots over time (Figure 4.2). *Triglochin maritima* successfully established from seed in the Seeded NF plots, whereas *Triglochin palustre, Sonchus arvensis, Carex atherodes,* and *Carex utriculata* established in NF from unknown sources. *Ptychostomum pseudotriquetrum* was the most successful bryophyte to establish in both fens. It colonized 94 % of NF plots and 35 % of SF plots and had higher cover in areas with a water level below -10 cm (Figure 4.3A and 4.3B). Species that were introduced but did not perform well include *Carex diandra* and *Scirpus atrocinctus* at SF, *Betula pumila, Calamagrostis inexpansa, Sarracenia purpurea* in Seeded NF, *Betula pumila, Vaccinium oxycoccos,* and all *Sphagnum* species in MLT plots at NF (Moss, Moss *Carex*, and Moss *Juncus*) (Tables A5.1 and A5.2).

Bryophytes established throughout both fens by intensive species introduction approaches in the MLT plots and through spontaneous colonization in suitable areas from indigenous sources. The most frequent bryophytes to colonize new areas were peat-forming *P. pseudotriquetrum, Aulacomnium palustre,* and *Drepanocladus polycarpus,* and ruderals *Leptobryum pyriforme, Funaria hygrometrica, and Ceratodon purpureus*. Bryophyte species richness was higher where MLT was implemented with frequent establishment of peat-forming mosses *P. pseudotriquetrum, Tomentypnum nitens, D. polycarpus, Brachythecium acutum, Drepanocladus aduncus, Calliergon giganteum, Campylium stellatum,* and ruderal *Leptobryum pyriforme*. *Sphagnum angustifolium, S. fuscum, S. warnstorfii, and S. capillifolium* were abundant at the donor site and in the harvested MLT but did not survive at NF (Table A5.2), whereas species more tolerant of the high salinity, such as *Campylium stellatum* and *Drepanocladus polygamous,* have increased in frequency over time.

COMMUNITIES AT THE RECLAIMED FENS

Five communities were determined by cluster analysis to have developed in the two fens along with four outlier plots. The communities were characterized by the species that contributed the greatest proportion of within community similarity (Table 4.1), which include Careaqu = *Carex aquatilis*, Calacan = *Calamagrostis canadensis*, Juncbal = *Juncus balticus*, Typhlat = *Typha latifolia*, and Trigpal = *Triglochin palustris.* Plots ordinated along the two major axes corresponding to water level and species introduction approach (Figure 4.4, Table 4.2, Tables A6 and A7). Depth to water level more strongly influenced plots along NMDS axis 1 that diverged to the wet (Typhlat) and dry (Calacan) ends of the water level gradient. Species introduction approach more strongly influenced plots planted with *J. balticus* at NF, such as *Juncus* and Moss *Juncus* that diverged along NMDS axis 2 (Juncbal).

The number of plots and the species introduction approach used at each fen differed by community (Figure 4.5, Table 4.1). Convergence occurred across species introduction approaches and fens in the Careaqu and Typhlat communities. Divergence occurred within fens due to the water level gradient that influenced dry (Calacan and Trigpal) and wet (Typhlat) communities, and species introduction approaches where *J. balticus* was planted (Juncbal). The Careaqu community developed across all species introduction approaches, including half of the Moss *Juncus* NF plots, the majority of Seeded SF, Unplanted NF, Seeded NF, and *Carex* NF plots, and all Moss NF and Moss *Carex* NF plots. The *Juncus* NF and Moss *Juncus* NF plots diverged equally into Juncbal and Careaqu communities.

COMMUNITY CHARACTERISTICS

Careaqu was the most common community type and occurred across all species introduction approaches in both fens (Figure 4.5). The average cover of *C. aquatilis* was over 60 % (Table 4.1) and the community occurred along the widest water level range from +78 cm to -30 cm (avg = -3 cm, se = 2.1; Figure 4.6A). Average bryophyte cover and species richness were low but with wide variance (6 %, se = 1.1 and 3 species, se = 0.3; Figures 7A and 7B). Desirable species cover was high (avg = 96 %, se = 4.5) but species richness was low (7 species, se = 0.5), whereas undesirable species cover and species richness were both low (avg = 13 %, se = 1.3 and avg = 2 sp., se = 0.2; Figures 7C and 7D).

The Calacan community is the second most common group and occurred in the driest plots at SF (avg = -21 cm, se = 3.1; Figure 4.6A), but was absent at NF. The average abundance of *Calamagrostis canadensis* was 48 % and the community had high multivariate dispersion (Figure 4.6B) with the most structural diversity, containing shrub, tree and bryophyte species (Table 4.1). Even though bryophytes were not intentionally introduced at SF, bryophyte cover reached 63 % and species richness peaked at 11 species in some Calacan plots (avg = 14 %, se = 3.2 and avg = 5 species, se = 0.5; Figures 7A and 7B). Because many plots had typical of upland water levels deep below the surface, the community had the highest cover and number of undesirable species (avg = 75 %, se = 4.6 and avg = 9 sp., se = 0.6; Figures 7C and 7D).

The Juncbal community developed at NF where *J. balticus* was introduced either by seed or seedling (Seeded, *Juncus*, and Moss *Juncus*) and the average water level was -21 cm (se = 2.0, n =11; Figure 4.6A). This community had the highest bryophyte cover and species richness (avg cover = 68 %, se = 7.8 and avg = 11 species, se = 1; Figures 7A and 7B) and supported the most peatland bryophyte species (Table 4.1). This community also had the highest cover and species richness of desirable species and low cover and species richness of undesirable species (Figures 7C and 7D).

The Trigpal community occurred in dry plots at NF where the water level averaged 27 cm below the soil surface (se = 2.4; Figure 4.6A). *Triglochin palustre* was not intentionally introduced but may have been in the MLT material seedbank and dispersed to Unplanted and SlC plots where it established. The undesirable species *Sonchus arvensis* and *Salix exigua* also occurred in this community (Table 4.1), likely recruited by aerially dispersed seed from populations in the adjacent uplands. Bryophyte establishment was limited, averaging 19 % cover (se = 7.5) with 7 species (se =1.2; Figures 7A and 7B). Cover and species richness of desirables exceeded that of undesirables, although undesirable cover was moderate at an average of 36 % (se = 6.0; Figures 7C and 7D).

The Typhlat community was dominated by *Typha latifolia* (Table 4.1), had the lowest multivariate dispersion (Figure 4.6B) and occurred only in ponded areas where the water level averaged 52 cm above the soil surface (se = 8.2, max = 79 cm, min = 1 cm; Figure 4.6A). The community developed in Seeded SF and Unplanted NF plots, did not support bryophytes (Figures 7A and 7C), and averaged 11 % desirable cover (se = 3.0) with 2 desirable species (se = 0.4; Figures 7C and 7D).

BRYOPHYTE COVER AND DESIRABLE SPECIES RICHNESS RESPONSE TO WATER LEVEL

Bryophyte cover in each community varied along the water level gradient (Figure 4.8A), being highest in plots where the water level was between -10 cm and -40 cm deep. Bryophyte cover declined sharply where the water level was at or near the surface and in Careaqu community plots where the water level was from 0 to -25 cm because of the dense *C. aquatilis* canopy. Desirable species richness had a similar pattern with decreasing species richness at near surface water levels and highest species richness where the water level was between -10 cm and -30 cm (Figure 4.8B).

Desirable to undesirable species ratio response to introduction approach and water level

The desirable to undesirable species relative cover ratio (D/U), where higher values indicate higher relative cover of desirables, differed in each community along the water level gradient (Figure 4.9). The Calacan community had a low D/U ratio and developed in Seeded SF plots where the water level averaged -21 cm (se = 3.2). Although water level ranges differed, the Juncbal community at NF occurred at the same average water level as the Calacan community but had a high D/U ratio (avg = -21 cm, se = 2.0). The Trigpal community also had a high D/U ratio and developed in drier Unplanted areas of NF (avg = -27 cm, se = 2.4). The Careaqu community had a high D/U ratio and developed in all species introduction approaches, where the water level was between -30 cm and +78 cm. The Typhlat community had a low D/U ratio and formed at SF in plots with ponded water (avg = +52 cm, se = 8.2).

4.4 Discussion

The two fen reclamation projects clearly demonstrate that establishing peat-forming bryophyte and vascular plant dominated communities along varying water level gradients is possible using targeted species and introduction approaches. Four and five years after plant introductions, five community types developed. Two communities were found at both fens and three were specific to each fen based on water level gradients and species introduction approaches. Community convergence was driven by one dominant species, *Carex aquatilis*, that spread from areas of introduction, obscuring other introduced species, and developed near homogenous colonies at both fens. Community divergence was attributed to a strong water level gradient, intensive species introductions, and initial establishment of pioneer species in unplanted areas. Basic approaches were appropriate for rapid colonizers, but also had the greatest proportion of divergent trajectories driven by water level gradients and pioneer species. Certain intensive approaches for planted species seedlings and spreading MLT did result in diverse fen communities, but others were susceptible to community convergence where *C. aquatilis* proliferated. Intensive approaches effort influenced the development of a desirable fen community and were effective in areas of niche overlap with undesirable communities along the water level gradient.

Plant species that performed well across the sites included desirable and undesirable peatland species. The most common desirable species to establish from seeds, seedlings, and the MLT, was *Carex aquatilis*, a wide-spread sedge in boreal marshes and fens (Vitt and Chee 1990, Gignac et al. 2004, Bayley and Mewhort 2004). Proposed as a key candidate for wetland reclamation in the oil sands region (Koropchak et al. 2012), *C. aquatilis* has established in reclaimed marshes and well pads (Trites and Bayley 2009, Vitt et al. 2011, Caners and Lieffers 2014) and is tolerant of higher solute concentrations than most other plants species (Mollard et al. 2012). *Juncus balticus* successfully established from seed and seedlings at NF and is a viable candidate for reclamation sites that likely will become saline (Trites and Bayley 2009). It is common in boreal wet meadows and fens (Cooper et al. 2006) and tolerates a wide range of soil water solute concentrations, observed at sites with EC of 0.1 to 20.1 mS/cm (avg. = 3.3 mS/cm; Kantrud et al. 1989). The most common bryophyte to establish was *Ptychostomum pseudotriquetrum*, a slender ruderal moss that inhabits a variety of substrates (BFNA 2014). *Ptychostomum pseudotriquetrum* is not known to be a dominant peat former in fens but is a key species in the recovery of recently burned peatlands (Rowe et al. 2017). Its abundance could represent an early seral stage of the reclamation fens and may provide the foundation for other mosses to colonize in areas with optimal water level depth and light availability.

Undesirable species spontaneously established in the fens in higher abundances in the most dry and wet areas. *Calamagrostis canadensis* is a perennial grass typical of upland forests and tolerant of temporarily flooded areas (Kantrud et al. 1989) and commonly associated with non-saline to slightly saline reclaimed oil sands wet meadows (Purdy et al. 2005). *Typha latifolia* is a rhizomatous wetland plant, occurring from tidal estuaries to temporary or permanently ponded marshes (Cooper et al. 2006). *Typha latifolia* has a preferred water level depth between 0 to +32 cm (Golder Associates 2005) and its growth increases with increasing standing water depth (Grace 1989). It is a major concern for fen reclamation because it readily colonized disturbed habitats, is tolerant of moderate salinity levels, and grows in dense monocultures that exclude other species (Koropchak and Vitt 2013, Shih and Finkelstein 2008).

Significant community convergence occurred between species introduction approaches and across fens due to the proliferation and dominance of *C. aquatilis*. Community convergence has been linked to dominant species in competition experiments where high relative growth rates suppress other planted species over time (Suter et al. 2007). Convergence has also been attributed to similarity in environmental conditions that support colonization and spread of adapted species (Inouye and Tilman 1995). *Carex aquatilis* has an advantage over many species because it is adapted to a wide range of environmental conditions. The habitat niche of *C. aquatilis* is extremely wide, having been recorded in sites with pH ranging from 3.1-9.2, electrical conductivities from 36 to 8820 uS/cm, calcium concentration from 0.2 to 146.6 mg/L, and water level depths ranging from -80 to +80 cm (Koropchak et al. 2012). It also produces long rhizomes that allow each genet to spread up to several meters per year. This tolerance of environmental heterogeneity and stressful abiotic conditions allows it to occupy habitats that restrict other species in reclamation sites. Dominance by *C. aquatilis* could also be a combination of initial propagule numbers and its increased performance relative to the decreased performance of other species in response to environment.

Community divergence of areas seeded with the same species mixture occurred along the water level gradient that facilitated the formation of a drier upland community characterized by *Calamagrostis canadensis* and a wetter marsh community dominated by *Typha latifolia*. Environmental heterogeneity in younger sites can exert a greater influence on community development and increase divergence of successional pathways compared to older areas (Inouye and Tilman 1995). Similar divergent pathways following passive reclamation of oil well pads in Alberta occurred where higher water levels and homogeneous topography restricted natural bryophyte recolonization and supported higher *C. aquatilis* cover (Caners and Lieffers 2014). Priority effects of pioneer species are also stronger following passive reclamation approaches and initial species identity and abundance can permanently alter ecosystem function resulting in undesirable alternate states (Young et al. 2001, Suding et al. 2004, Weidlich et al. 2017).

Basic and intensive species introduction approaches are appropriate in different situations. At water levels between -25 to +30 cm, the Careaqu community dominated following both basic and intensive approaches. At NF this included convergence of seeded and MLT plots where species richness was high in the first two years but then declined as *C. aquatilis* spread, attained high cover and taller stature that suppressed other species (Borkenhagen and Cooper unpublished). This indicated that intensive approaches of planted *C. aquatilis* seedlings or spreading MLT containing *C. aquatilis* are not required in certain areas. Because of the reduction of species richness overtime in plots containing *C. aquatilis,* its inclusion should be considered with caution in projects that are prioritizing biodiversity. Intensive approaches may be more important in shallow ponded areas where planting *C. aquatilis* could be a viable strategy to prevent the establishment of *T. latifolia*. The overlap between these two species along the water level gradient suggest that targeted introductions of *C. aquatilis* in areas predicted to become ponded could limit *T. latifolia* invasion to ensure that the area is colonized by desirable fen species. Experiments support the potential for this approach as *C. aquatilis* has higher performance (growth, fruiting, and tillering) in wetter environments compared to drier environments (Vitt et al. 2011) and *T. latifolia* cover was reduced where *C. aquatilis* seedlings were planted at the NF (Borkenhagen and Cooper unpublished). Overlaps in community distributions occurred in drier areas where intensive approaches could introduce Juncbal or Trigal communities to deter development of the Calacan community that resulted from a basic approach.

An intensive species introduction approach was responsible for creating the Juncbal community, the only community to achieve fen reclamation targets for Alberta of bryophyte and desirable species cover and species richness (Environment and Parks 2017). All plots were planted with *J. balticus* seedlings and half the plots received the MLT. After four years, bryophytes had established in all *J. balticus* seedlings plots, a trend also documented at SF in the Calacan community where no intensive introductions occurred (Vitt and House 2015). This suggests that bryophyte cover and species richness may increase over time in suitable areas from indigenous sources (Vitt and House 2015). Recruitment of peat-forming species could lag if reclaimed sites are far from source populations, however actively inoculating areas within the optimal hydrologic range and under co-dominant vascular plant species accelerated establishment and increases species richness. This also allowed the intentional selection of species tolerant of current or expected environmental conditions. Another potential approach could be to inoculate the site with adapted bryophyte species a year or two after planting once the post-construction environmental conditions have been evaluated.

Bryophyte cover and desirable species richness was strongly influenced by depth to water level, with the highest cover and species richness occurring where the water level was between -10 cm to -40 cm in July. This water level range is deeper than recommended for *Sphagnum* spp. establishment in bog restoration (Rochefort et al. 2003). However, experiments with fen mosses have achieved high cover in mesocosms with deeper water levels as species composition shifts to support more drought tolerant species such as *Tomentypnum nitens* and *Aulacomnium palustre* (Borkenhagen and Cooper 2016). Desirable species richness also declined when water levels were near the soil surface. This trend has been observed in other wetland types where species richness was negatively correlated with water level, flood duration and frequency (Casanova and Brock 2000, Dwire et al. 2006). Although rich-fens have near surface water levels, their high biodiversity is influenced by the numerous microhabitats and district hummock-hollow topography (Vitt and Chee 1990, Vitt et al. 1995). Future projects should aim to increase microsite topography and increase depth to water level to support bryophytes and desirable vascular plant cover and species richness.

Three of the communities that developed have vegetation composition typical of natural regional fens based on their desirables species cover and species richness. All three communities were dominated by herbaceous plants, and although not as common as bryophyte-dominated fens, herbaceous rich and saline fens that contain *C. aquatilis*, *J. balticus*, and *T. palustre* exist in the region (Vitt et al. 1998, Halsey 2008, Environment and Parks 2017). This suggests that we need to reevaluate our short-term targets based on reference fens that are thousands of years old as vegetation composition common during fen initiation may be more appropriate (Koropchak et al. 2012, Borkenhagen and Cooper 2016). Boreal peatlands can develop from pond/marsh to rich fen to poor fen to bog and peat formed by *Carex* species are commonly found at the base of profiles, but basal layers very rarely contain *T. latifolia* (Kuhry et al. 1993, Zoltai and Vitt 1995, Koropchak et al. 2012). This emphasizes the importance of excluding *T. latifolia* as its presence indicates different environmental and ecological conditions that likely will not develop into a peat-accumulating system.

Although function was not measured, structure-function relationships can help predict the peat-accumulation potential of these communities (Cortina et al. 2006). Herbaceous fens and marshes accumulate less peat than moss-dominated wooded fens due to higher herbaceous litter quality and fluctuating water levels that increase decomposition rates (Thormann et al. 1999). Herbaceous plant productivity has been correlated with site wetness, and many fen mosses do not tolerate repeated inundation or wide variations in water level (Szumigalski and Bayley 1997, Thormann and Bayley 1997, Borkenhagen and Cooper 2018). Diversity-productivity goals for reclaimed fens may also be in opposition to each other as the relationship can be negative and confounded by environmental variables (Gough et al. 1994, Doherty and Zedler 2014). Even though species richness decreased at shallower water levels in communities dominated by *C. aquatilis,* a loss of species does not always result in a loss of function (Zedler 2000, Smith and Knapp 2003, Sasaki and Lauenroth 2011). *Carex aquatilis* at NF produced more biomass per square meter than any other species (Messner et al. in prep), so if stable water levels that promote production and reduce decomposition are maintained, high peat- accumulation rates driven by *C. aquatilis* primary production is possible.

Predicting community development is challenging in restoration and reclamation projects as successional trajectories are highly variable in rate and direction and in response to environmental gradients and biotic interactions (Tilman 1987, Zedler and Callaway 1999, Suding and Hobbs 2009). Water level position and fluctuations plays a critical role in species composition within a wetland (e.g. Jeglum 1971, Spence 1982, Casanova and Brock 2000, Andersen et al. 2011), however additional analyses could include water and peat chemistry, pH, and conductivity as they have been shown to affect growth and distribution of bryophytes and vascular plants across fens (Vitt and Chee 1990, Pouliot et al. 2012, Trites and Bayley 2009, Vitt et al. 2016). Previous evaluations have shown that sodium concentrations increase in drier areas at SF (Vitt et al. 2016) and bryophyte cover is reduced where water electrical conductivity values exceed 3,250 uS/cm at NF (Borkenhagen and Cooper unpublished). Furthermore, the niche range of certain species in early seral communities may be wider than occupied in reference communities (Lepš and Rejmánek 1991). This may explain the unpredicted proliferation of *C. aquatilis*, typically a co-dominant species in regional fens (Cooper et al. 2006), and *P. pseudotriquetrum*, a slender erect moss that occurs in low abundances in rich fens (BFNA 2014).

Despite the challenges of oil sands reclamation, peat-accumulating bryophyte and vascular plant dominated communities have developed in both constructed fens. Community convergence occurred between intensive species introduction approaches across fens due to the dominance of *C. aquatilis*, whereas community divergence occurred where a basic species introduction approach was used in response to a broad water level gradient. Future projects should aim to increase microtopography to enable summer depth to water level ranges between -10 and -40 cm that support desirable bryophyte and vascular plant cover and species richness. In addition, co-dominant vascular plants, such as *J. balticus*, should be planted to encourage bryophyte establishment. Bryophytes spontaneously colonized suitable areas, but intensive introduction using the MLT introduced peat-forming species with high-production rates and tolerances to water level ranges and elevated salinity. Intensive introduction of desirable vascular plant species along the full water level gradient could exclude the establishment of undesirable upland or marsh plants that do not have peat-accumulation potential. The convergence and divergence of communities across constructed fens in response to abiotic and biotic conditions highlights the challenges of selecting species for novel environments and the importance of experimentation, comparative assessments, and long-term monitoring.

4.5 TABLES

Table 4.1 Results from the Similarity Percentage analysis (SIMPER) showing the highest contributory species in abundance to the similarity among communities and the number of plots in each community (n). Includes average abundance (Av.Abund), average similarity (Av.Sim), similarity over standard deviation (Sim/SD), percent contribution (Contrib%), and cumulative contribution (Cum.%).

Community	Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Careaqu (n=86)	*Carex aquatilis*	61.58	52.04	3.83	88.24	88.24
	Typha latifolia	8.09	2.06	0.39	3.5	91.74
Juncbal (n=11)	*Juncus balticus*	28.1	22.45	3.79	38.87	38.87
	Carex aquatilis	16.7	12.26	1.81	21.23	60.09
	Ptychostomum pseudotriquetrum	14.17	10.46	2.67	18.11	78.2
	Sonchus arvensis	4.76	2.08	0.89	3.6	81.81
	Leptobryum pyriforme	3.27	1.7	1.24	2.95	84.75
	Drepanocladus polycarpus	3.14	1.45	1.09	2.51	87.26
	Tomentypnum nitens	4.15	1.29	0.89	2.23	89.49
	Triglochin maritima	4.21	1.15	0.46	1.99	91.47
Trigpal (n=5)	*Triglochin palustris*	23.6	17.08	1.97	32.64	32.64
	Sonchus arvensis	16.75	11.89	1.28	22.73	55.36
	Carex aquatilis	11.7	7.48	1.39	14.29	69.65
	Salix exigua	10.5	3.07	0.48	5.87	75.52
	Ptychostomum pseudotriquetrum	5.63	2.61	0.96	4.98	80.5
	Leptobryum pyriforme	3.68	1.9	1.69	3.63	84.13
	Juncus balticus	3.7	1.8	1.07	3.44	87.57
	Salix species	3.82	1.77	1.02	3.39	90.95
Typhlat (n=9)	*Typha latifolia*	82.51	74.35	6.85	91.84	91.84
Careath (n=2)	*Carex atherodes*	38.31	31.58	NA	49.66	49.66
	Calamagrostis canadensis	24.04	21.05	NA	33.1	82.76
	Equisetum arvense	15.03	9.01	NA	14.17	96.93
Calacan (n=28)	*Calamagrostis canadensis*	47.62	39.1	3.49	76.96	76.96
	Carex aquatilis	6.04	1.49	0.35	2.93	79.88
	Rubus idaeus	4.62	1.47	0.5	2.9	82.78
	Populus balsamifera	4.77	1.47	0.51	2.88	85.67
	Ptychostomum pseudotriquetrum	4.6	1.17	0.46	2.3	87.96
	Salix species	3.61	0.91	0.48	1.78	89.75
	Leptobryum pyriforme	1.8	0.71	0.76	1.39	91.14
Poapal (n=1)	*Poa palustris*	NA	NA	NA	NA	NA
Popubal (n=1)	*Populus balsamifera*	NA	NA	NA	NA	NA

Table 4.2 Fit of continuous environmental variable vectors that are significantly correlated at $\alpha = 0.05$ to the NMDS ordination. Coordinates on NMDS1 and NMDS2 axes are for the heads of unit length vectors that are scaled by their squared correlation coefficient (r^2).

Environmental Vectors	NMDS1	NMDS2	r^2	*P*-value
Depth to water level	-0.9265	0.3764	0.3453	0.001
Sp. Introduction method			0.1404	0.028
Seeded SF	0.2126	0.1229		
Unplanted N	-0.3151	-0.2369		
Seeded NF	-0.3340	-0.1991		
Carex NF	-0.2132	0.2246		
Juncus NF	-0.2266	-0.6631		
Moss NF	-0.2094	0.0355		
Moss Carex NF	-0.3098	0.2911		
Moss Juncus NF	-0.2196	-0.3662		

4.6 FIGURES

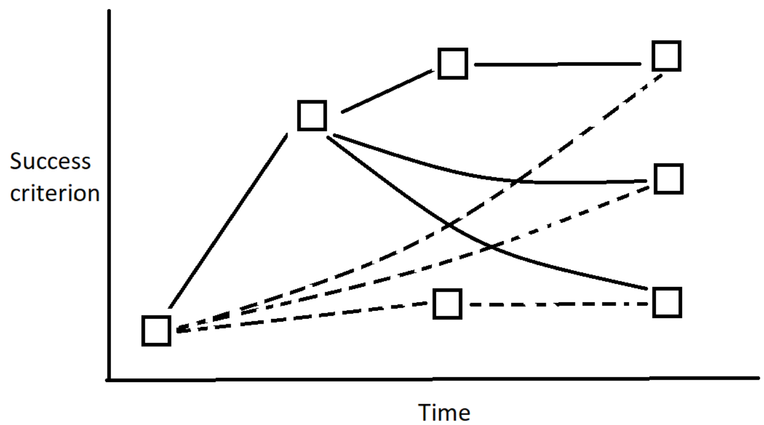

Figure 4.1 Conceptual figure of patterns of convergence and divergence along a success criterion over time following intensive (solid line) or basic (dashed line) species introduction approaches of a site (circle). Adapted from Mathews and Spyreas (2010) and Suding (2011).

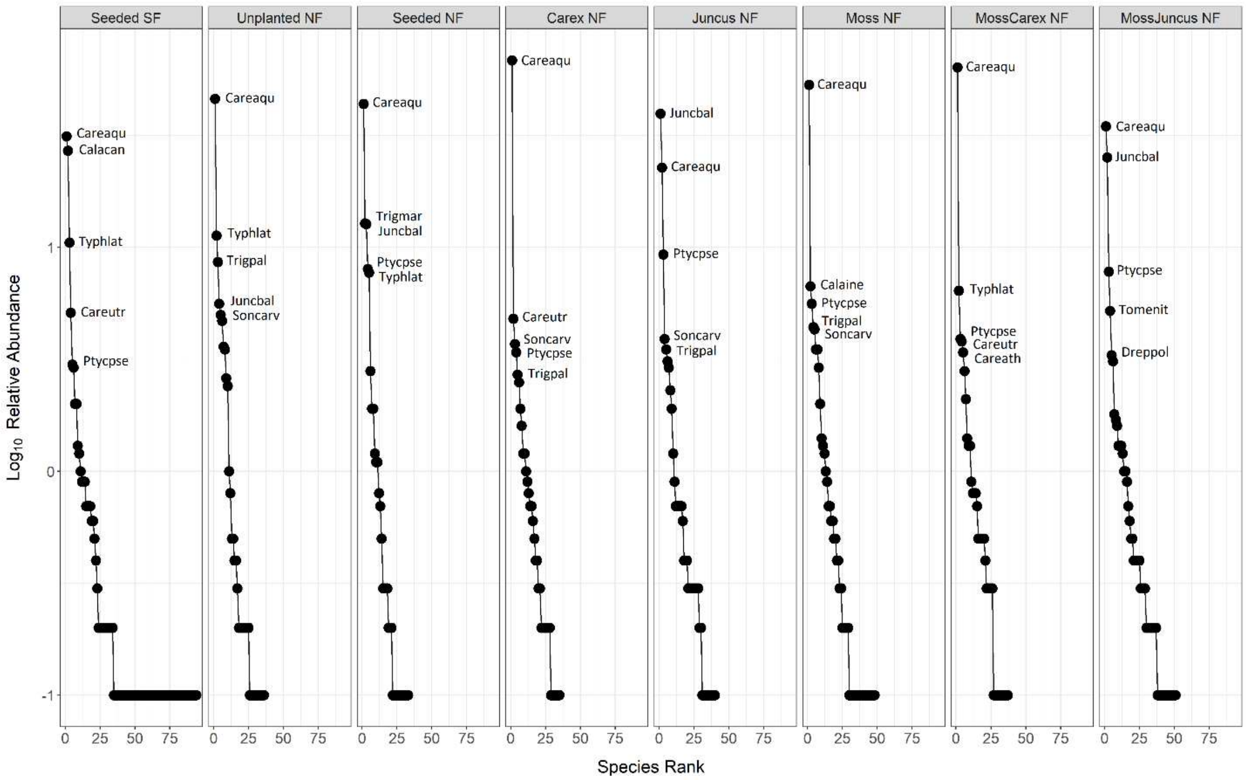

Figure 4.2 Rank abundance diagram of $\log_{10}$ relative abundance plotted against species rank, where 1 = highest abundance. The top 5 ranked species are labeled for each species introduction method in the Sandhill Fen (SF) and Nikanotee Fen (NF). The number of species recorded at SF was 94 and at NF was 67. Calacan=*Calamagrostis canadensis*, Calaine=*Calamagrostis inexpansa*, Careath=*Carex atherodes*, Careaqu=*Carex aquatilis*, Careutr= *Carex utriculata*, Dreppol=*Drepanocladus polycarpus*, Juncbal=*Juncus balticus*, Ptycpse=*Ptychostomum pseudotriquetrum*, Soncarv=*Sonchus arvensis*, Tomenit=*Tomentypnum nitens*, Trigmar=*Triglochin maritima*, Trigpal=*Triglochin palustris*, Typhlat=*Typha latifolia.*

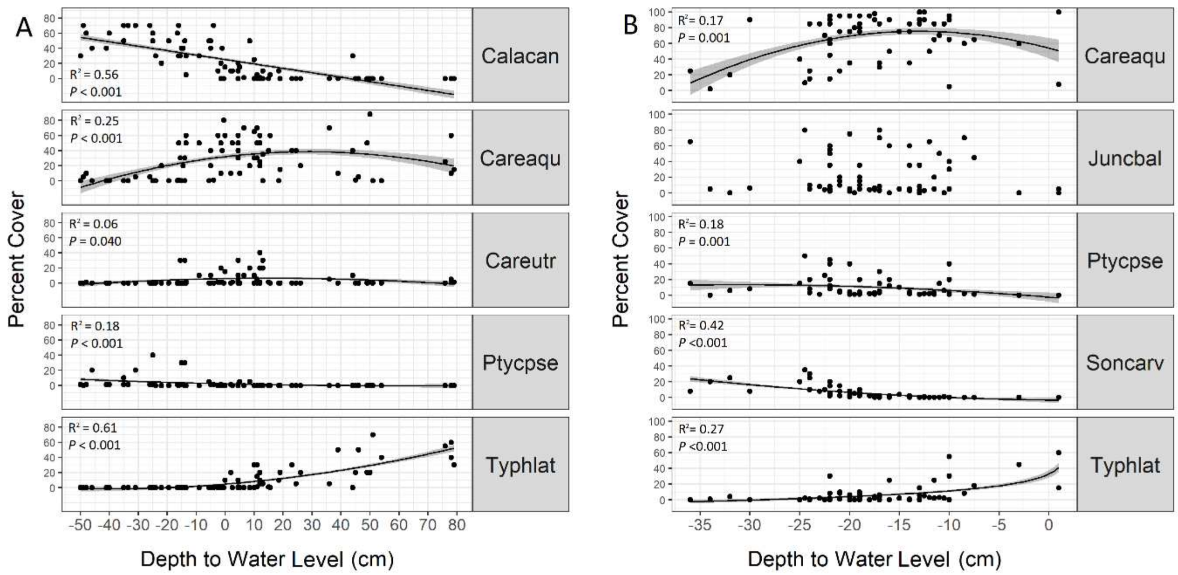

Figure 4.3 Relative abundance distribution of species growing along the depth to water level gradient in the Sandhill Fen (A) and Nikanotee Fen (B). Water level sampled on July 13-14, 2017 at SF and July 25, 2017 at NF. Selected species are the top 5 ranked at each fen respectively. Best fit regressions determined by AIC and adjusted R^2 shows goodness of fit. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Careutr = *Carex utriculata*, Juncbal = *Juncus balticus*, Ptycpse = *Ptychostomum pseudotriquetrum,* Soncarv = *Sonchus arvensis*, and Typhalat = *Typha latifolia*.

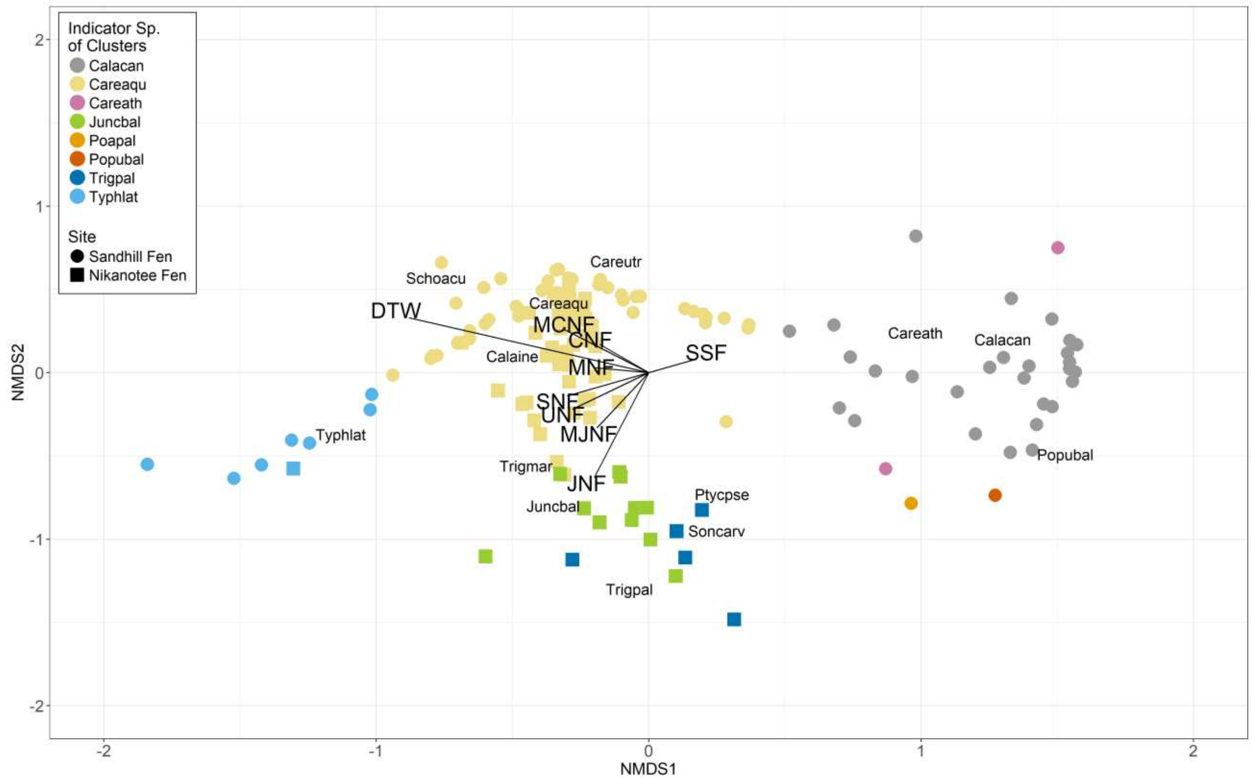

Figure 4.4 Non-metric multidimensional scaling (NMDS) ordination of plots from Sandhill (n=79) and Nikanotee (n=64) Fens. Symbols represent plots. Colors indicate communities determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Vectors are projected for depth to water level (DTW) and species introduction method (SSF=Seeded Sandhill Fen, UNF=Unplanted Nikanotee Fen(NF), SNF=Seeded NF, CNF=Carex NF, JNF=Juncus NF, MNF=Moss NF, MCNF=Moss Carex NF, MJNF=Moss Juncus NF; $P < 0.05$). Presented species are dominant and show their distribution relative to communities (Calacan = *Calamagrostis canadensis*, Calaine = *Calamagrostis inexpansa*, Careaqu = *Carex aquatilis*, Careath = *Carex atherodes*, Careutr = *Carex utriculata*, Juncbal = *Juncus balticus*, Popubal = *Populus balsamifera*, Schacu = *Schoenoplectus acutus*, Soncarv = *Sonchus arvensis*, Trigmar = *Triglochin maritima*, Trigpal = *Triglochin palustris*, Typhalat = *Typha latifolia*, Ptycpse = *Ptychostomum pseudotriquetrum*)

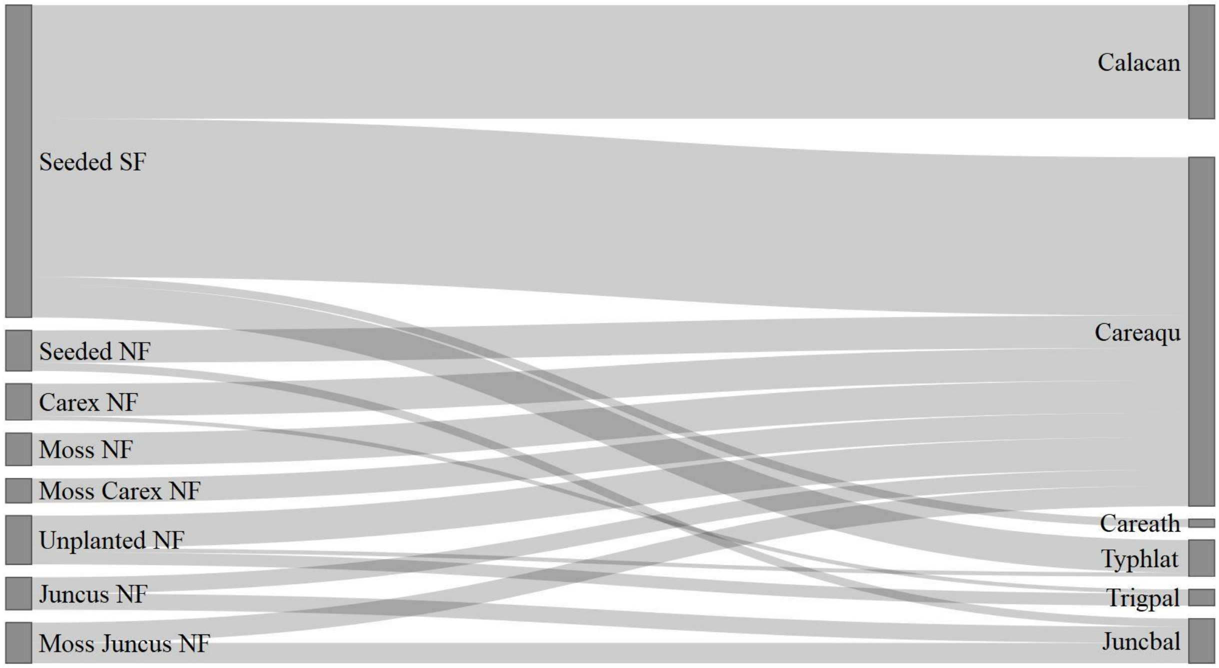

Figure 4.5 Flow diagram showing the proportion of plots in each species introduction approach that converged or diverged into the communities four (NF) and five (SF) years after project implementation at the Sandhill Fen (SF) and Nikanotee Fen (NF). Communities were determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Omitted are Poapal, n=1; and Popubal, n=1, which were outlier SF plots. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Careath = *Carex atherodes*, Juncbal = *Juncus balticus*, Trigpal = *Triglochin palustris*, and Typhalat = *Typha latifolia*.

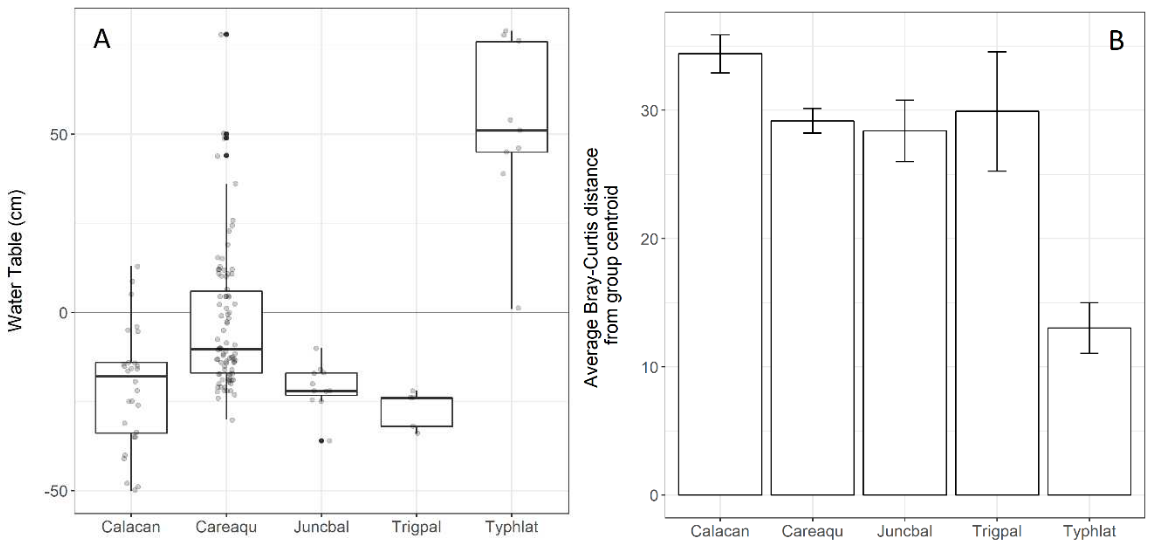

Figure 4.6 Plots showing the water level (A) and average Bray-Curtis distance from group (community) centroid (B) for each community. Water level depth was measured at SF on July 12-13, 2017 and at NF on July 25, 2017. Average Bray-Curtis distance from group centroid is presented as a measure of beta diversity in the communities. Boxplots show median values with the 25th and 75th percentiles, the whiskers show the range of values falling within 1.5 interquartile ranges of either quartile, black points represent outliers, and gray points show the distribution of values. Bars represent mean values with +1 standard error. The number of plots per community vary; Calacan, n=28; Careaqu, n=86; Juncbal, n=11; Trigpal, n=5; Typhlat, n=9. Omitted here are Careath, n=2; Poapal, n=1; and Popubal, n=1. Communities were determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Juncbal = *Juncus balticus*, Trigpal = *Triglochin palustris*, and Typhalat = *Typha latifolia*.

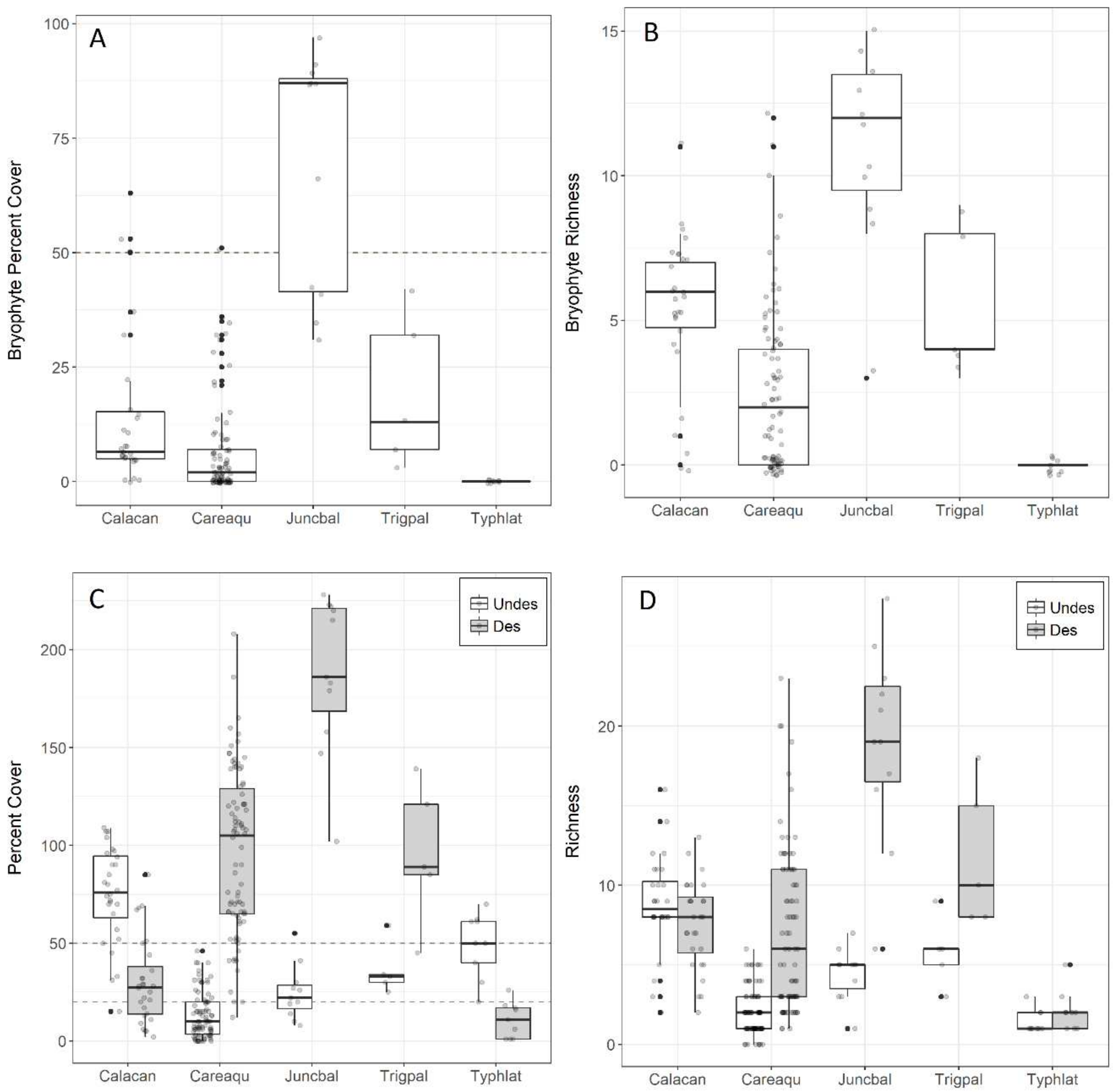

Figure 4.7 Plots showing total bryophyte percent canopy cover (A) and species richness (B), and desirable (Des) and undesirable (Undes) peatland species percent cover (C) and species richness (D) for each community. Dashed lines show targets based on Environment and Parks Canada peatland reclamation criteria (2017). Boxplots show median values with the 25th and 75th percentiles, the whiskers show the range of values falling within 1.5 interquartile ranges of either quartile, black points represent outliers, and gray points show the distribution of values. The number of plots per community vary; Calacan, n=28; Careaqu, n=86; Juncbal, n=11; Trigpal, n=5; Typhlat, n=9. Omitted here are Careath, n=2; Poapal, n=1; and Popubal, n=1. Communities were determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Juncbal = *Juncus balticus*, Trigpal = *Triglochin palustris*, and Typhalat = *Typha latifolia*.

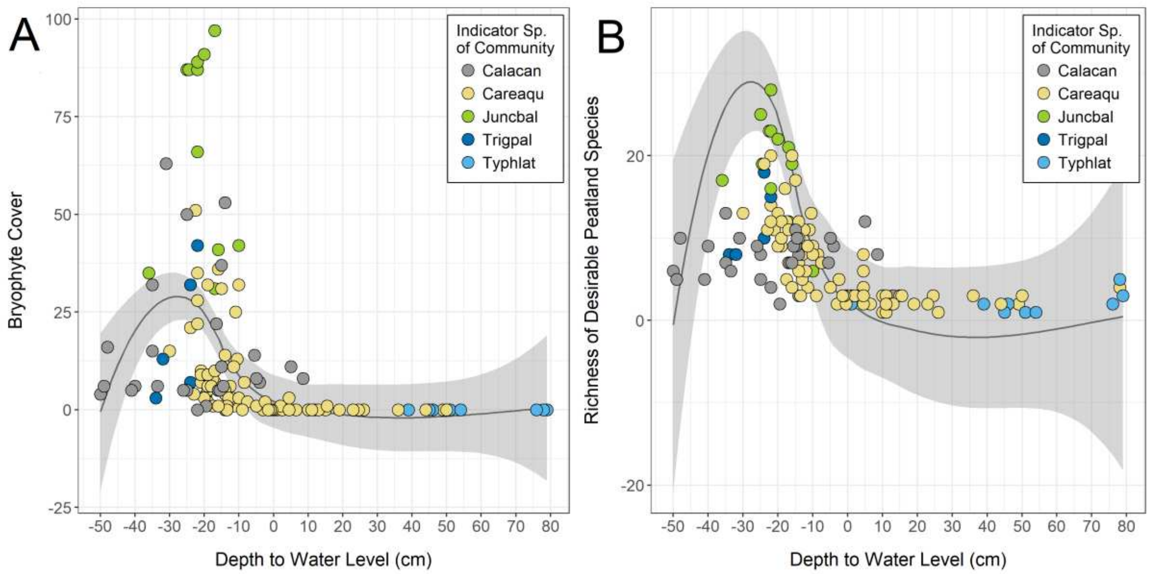

Figure 4.8 The effect of water level on total bryophyte cover (A) and desirable species richness (B) in communities across Sandhill (n=75) and Nikanotee Fens (n=64). Local polynomial regression fitting curve across communities with standard error shading. Communities were determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Juncbal = *Juncus balticus*, Trigpal = *Triglochin palustris*, and Typhalat = *Typha latifolia*. Omitted here are Careath, n=2; Poapal, n=1; and Popubal, n=1.

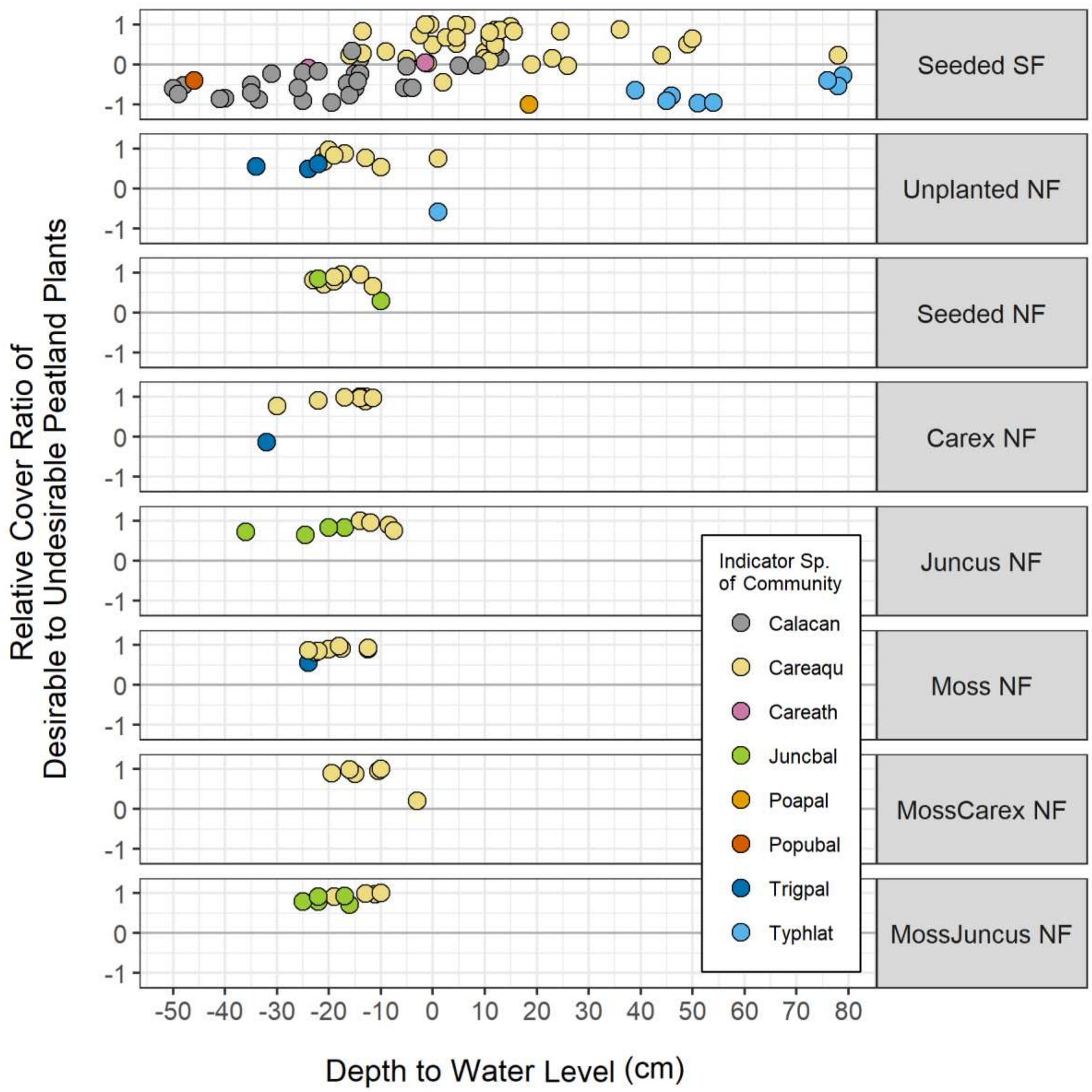

Figure 4.9 Relative cover ratio of desirable and undesirable peatland plants in each species introduction approach and the divergence of communities in response to water level. Values > 0 have higher relative cover of desirable to undesirable species. Species introduction methods include Seeded Sandhill Fen (SF), n=79; Unplanted Nikanotee Fen (NF), n=12; Seeded NF, n=10; Carex NF, n=9; Juncus NF, n=8; Moss NF, n=9; Moss Carex NF, n=6; and Moss Juncus NF, n=10. Communities were determined by group average cluster analysis with a cut off level of 40% similarity and classified by the species that contributed the greatest proportion of similarity within groups, as determined by SIMPER analysis. Calacan = *Calamagrostis canadensis*, Careaqu = *Carex aquatilis*, Juncbal = *Juncus balticus*, Trigpal = *Triglochin palustris*, and Typhalat = *Typha latifolia*. Omitted here are Careath, n=2; Poapal, n=1; and Popubal, n=1.

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5 SYNTHESIS

The preceding chapters address key concepts that contribute to the development of reclamation solutions required after oil sands mining in Canada. I evaluated the response of four common moss species to submergence in a natural boreal fen, tested the establishment of bryophyte and vascular plants in a constructed fen using a large-scale multi-factorial experiment, and conducted a comparative study of the drivers on plant community development in two reclaimed fens in the oil sands region. Results from these studies highlight the critical role moss communities have in maintaining the functional stability of boreal ecosystems, that it is possible to establish desirable peatland species in a constructed fen, and that abiotic and biotic drivers influence fen community composition.

In chapter 2, I evaluated the tolerance of four fen moss species and moss communities to submergence duration. Flood disturbance in peatlands can create temporary or permanently submerged areas that can dramatically affects ecosystem function (Roulet et al. 1992, Roulet et al. 1997, Kelly et al. 1997, Kim et al. 2014, St. Louis et al. 2000). Fen moss species and communities occur along a depth to water level gradient (Gignac et al. 1991; Rydin & Jeglum 2013; Vitt 2014), but their resistance and resilience to different durations of submergence had not been tested. I found that moss species response to submergence duration was not strictly related to their preferred niche along a water level gradient and that more tolerant species replaced less tolerant species following flooding disturbance. Overall, moss communities without *S. warnstorfii* were resilient to 4 weeks of submergence because *T. nitens* and *H. vernicosus* maintained dominance or established where the cover of less resilient species was limited. I showed that moss species varied in their responses to submergence duration, resulting in shifts in community composition that increased diversity and moss community resilience. This empirical work demonstrates how moss communities create stability to peatlands in response to disturbance through shifts in community composition that support tolerant dominant species.

In chapter 3, I tested a variety of planting methods to determine the most effective approach to establish fen bryophyte and vascular plants in a constructed boreal fen. Oil sands mining in Alberta has disturbed large areas of fen peatlands and no formal strategies exist to reconstruct the landscape. Peatland reclamation is difficult, but industry and regulators are now supporting innovative research to address the knowledge gap and refine reclamation approaches (Price et al. 2010, Daly et al. 2012). In a large-scale multifactorial field experiment, I designed and implemented various species introduction approaches that tested introducing bryophyte and vascular plants by moss layer transfer (MLT), seeds, and seedlings under wood-strand mulch and with a *Typha latifolia* weeding treatment. Four years after planting, the MLT and *Juncus balticus* seedling treatment supported the highest fen bryophyte and vascular plant cover and species richness. Weeding did reduce *T. latifolia* cover but was not necessary in areas where seedlings or the MLT was introduced likely due to competitive exclusion. The most successful fen species to establish was *C. aquatilis*, which readily colonized but also reduced cover and richness of bryophytes and other vascular plants. Depth to water table influenced species distribution, with shallow water tables supporting more *T. latifolia* and lower bryophyte and vascular plant species richness and cover. My research is the first of its kind to evaluate a range of vegetation introduction methods for fen reclamation and shows that it is possible to restore fen peatlands in the post-mining landscape of Alberta.

In chapter 4, I examined vegetation establishment in two regional reclaimed fens under different species introduction approaches and water level gradients. Two pilot fen projects on oil sands mines in northern Alberta were constructed to mimic natural regional fen ecosystems and consider the constraints of a post-mining landscape. Though the two designs differed, similarities emerged and the vegetation data was synthesized to extract commonalities and inform future fen reclamation efforts. Despite the challenges of oil sands reclamation, peat-accumulating bryophyte and vascular plant dominated communities developed in both constructed fens. Community convergence occurred due to the dominance of *C. aquatilis*, and community divergence occurred in response to water level gradient. Basic and intensive species introduction approaches are appropriate in different situations. My research showed that intensive approaches that introduce a dominant species adapted to site conditions are not required.

Dominant species with wide habitat niches and high production can outcompete other species and their introduction should be considered with caution in projects that are prioritizing biodiversity. Intensive approaches should prioritize areas of overlap in desirable and undesirable community distributions to deter establishment of non-peat forming species. Bryophyte cover and desirable species richness was highest following intensive approaches and where the summer water level was between -10 cm to -40 cm from the soil surface. Of the fen communities that developed, all were herbaceous dominated and comparable to an uncommon regional fen type. This highlights the need to shift our short-term targets away from moss-dominated reference fens that are thousands of years old and towards communities that are typically found during the initiation stage of peatlands (Koropchak et al. 2012, Borkenhagen and Cooper 2016).

To prioritize plant diversity and bryophyte cover, future projects should increase microtopography, design for summer water tables between -10 and -40 cm from the soil surface, select species tolerant of expected abiotic conditions, and introduce co-dominant plants. My dissertation work is an integral part of the increasing body of knowledge addressing peatland reclamation issues in Alberta's oil sands. My experimental design testing various species introduction approaches at Nikanotee Fen provided the foundation for research in greenhouse gas flux dynamics, above and below-ground nutrient cycling and biomass accumulation, biogeochemical processes, and the development of a functional-based approaches for evaluating constructed peatlands (Murray et al. 2017, Scarlett et al. 2017, Nwaishi et al. 2015, Nwaishi et al. 2016a, 2016b, Messner et al. in prep). Future research at Nikanotee Fen includes evaluating the controls on water use efficiency and biogeochemical cycling of dominant vegetation and how this varies through the range of wetland succession trajectories, assessing the relationship between fen plant diversity and plant production, and altering depth to water table to maximize desirable fen bryophyte and vascular plants.

This research has broad implications for peatland reclamation in the oil sands region of Alberta and other highly-disturbed sites worldwide. Previous assertions that peatlands cannot be reclaimed after mining activities are antiquated (Rooney et al. 2012) as large-scale construction designs and species introduction approaches for fens are actively underway and the results are proven. Functioning watersheds have been constructed (Ketcheson et al. 2016), fen plant communities established (Vitt and House 2015, Vitt et al. 2016), biomass production and below-ground nutrient cycling processes are occurring (Nwaishi et al. 2016a), and carbon is being sequestered (Nwaishi et al. 2016b). The gap in our understanding and ability to reclaim peatlands in a post-mining landscape is narrowing. My research shows that it is possible establish fen bryophyte and vascular plant communities in the post-mining landscape of Alberta. This research also highlights the need for long-term monitoring of reclamation sites and the benefits of multi-factorial experiments, and that despite the constraints, a range of successful outcomes are achievable

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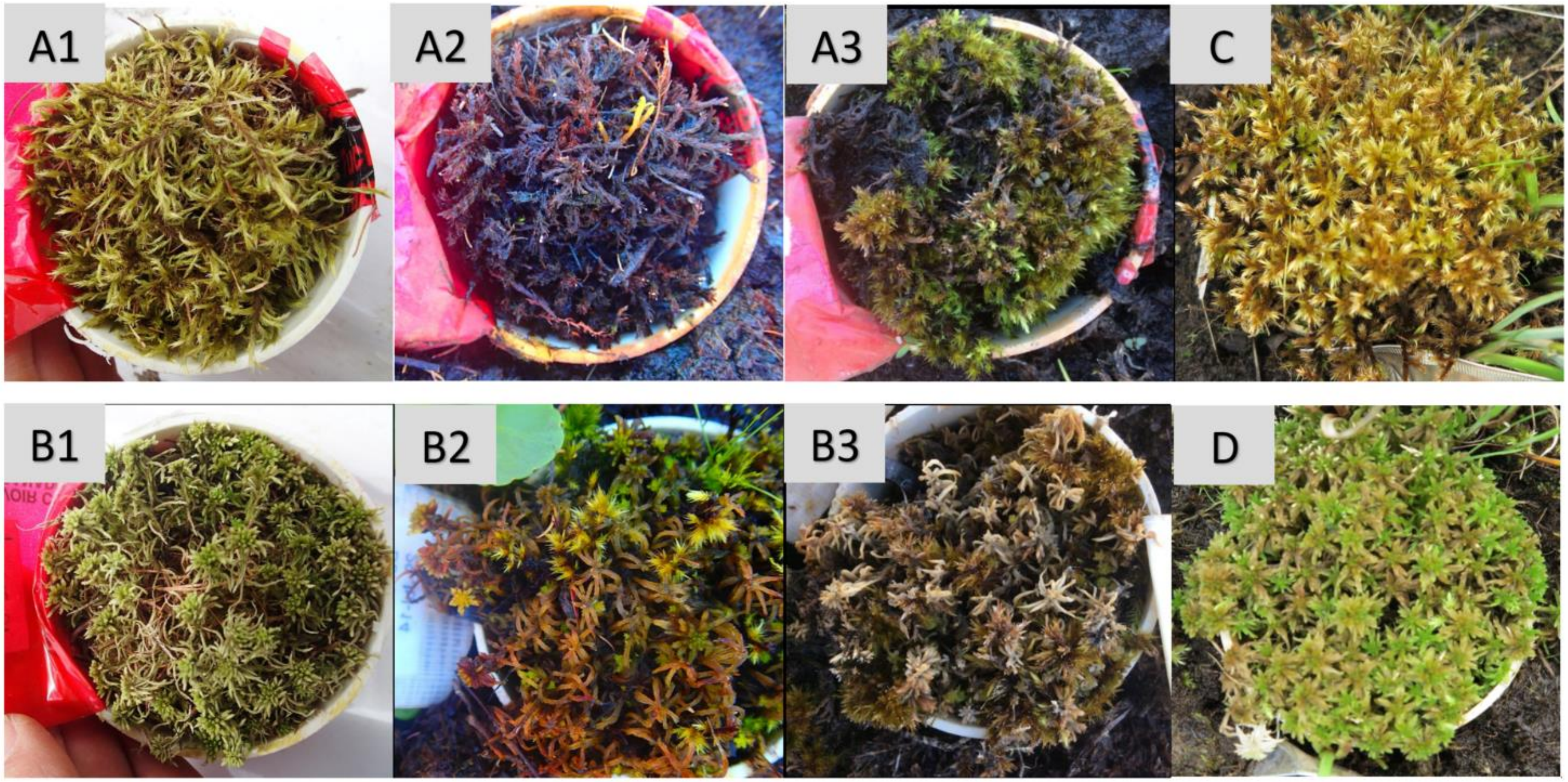
APPENDIX I

Table A1.1 Water chemistry of rich fen, sampled from dugout pits on August 8, 2014 and July 11, 2015. The value presented is the mean from three samples with standard error in parenthesis.

Date	pH	EC (uS/cm)	F^- (µg/ml)	Cl^- (µg/ml)	Br^- (µg/ml)	NO_3^- (µg/ml)	PO_4^{3-} (µg/ml)	SO_4^{-2} (µg/ml)	Na^+ (µg/ml)	NH_4^+ (µg/ml)	K^+ (µg/ml)	Mg^{2+} (µg/ml)	Ca^{2+} (µg/ml)
08/14	6.88	340.8	0.09	4.17	< 0.05	0.16	< 0.05	4.24	8.96	0.30	5.16	10.6	26.7
	(0.09)	(72.0)	(0.01)	(0.62)		(0.02)		(1.84)	(1.03)	(0.02)	(3.74)	(0.64)	(0.54)
07/15	6.94	254.3	0.08	2.41	0.16	0.21	< 0.05	1.97	6.77	1.31	1.45	9.11	39.2
	(0.06)	(37.8)	(0.01)	(0.93)	(0.01)	(0.13)		(0.53)	(1.68)	(0.50)	(0.48)	(3.20)	(11.4)

APPENDIX II

Figure A2.1 Photos of TnOM (A) and SwOM (B) plugs prior to submergence on August 6, 2014 (1), six weeks after submergence (Short-term response) on September 23, 2014 (2), and 11-months after submergence (Long-term response) on June 12, 2015 (3). The TnOM plug (A) was submerged for 8 weeks and the SwOM plug (B) that was submerged for 2 weeks. Photos of a control TnOM plug (C) and SwOM plug (D) six weeks after planting (Short-term response) on August 19, 2017 to effect of planting without submergence.



APPENDIX III

Table A3.1 Kruskal-Wallis chi-squared model results of species and total moss percent cover short and long-term responses to submergence duration. The plugs were named to represent the original dominant moss; TnOM = *Tomentypnum nitens* Original Moss, HvOM = *Hamatocaulis vernicosus* Original Moss, ApOM = *Aulacomnium palustre* Original Moss, and, SwOM = *Sphagnum warnstorfii* Original Moss. Bolded values indicate significant differences between treatment means at α = 0.05.

Plugs	Dep. variable	Effect	Term	$\chi 2$	df	P
HvOM	*Hamatocaulis vernicosus*	Weeks of Submergence	Short-term	4.84	4	0.304
			Long-term	7.01	4	0.136
	Total moss	Weeks of Submergence	Short-term	2.88	4	0.578
			Long-term	3.98	4	0.408
TnOM	*Tomentypnum nitens*	Weeks of Submergence	Short-term	13.66	4	**0.009**
			Long-term	10.81	4	**0.029**
	Total moss	Weeks of Submergence	Short-term	13.94	4	**0.008**
			Long-term	12.46	4	**0.014**
ApOM	*Aulacomnium palustre*	Weeks of Submergence	Short-term	15.37	4	**0.004**
			Long-term	14.15	4	**0.007**
	Total moss	Weeks of Submergence	Short-term	16.51	4	**0.002**
			Long-term	13.35	4	**0.01**
SwOM	*Sphagnum warnstorfii*	Weeks of Submergence	Short-term	17.36	4	**0.002**
			Long-term	14.40	4	**0.006**
	Total moss	Weeks of Submergence	Short-term	16.96	4	**0.002**
			Long-term	14.54	4	**0.006**

Table A3.2 Pairwise Conover multiple comparison test results of species and total moss percent cover short (St) and long-term (Lt) responses to submergence duration (Weeks Sub). Bolded values indicate significant differences between treatment means at α = 0.05. Means with different letters represent total moss covers that are significantly different within short and long-term response periods (Kruskal-Wallis Conover's test for multiple comparison, $P < 0.05$; $n = 4$ for each species, except *T. nitens* in Week 4 has $n = 3$). Tomenit = *Tomentypnum nitens*, Aulapal = *Aulacomnium palustre,* Sphawar = *Sphagnum warnstorfii*.

		TnOM				ApOM				SwOM			
Effect	Treatment comparison	Tomenit		Total moss		Aulapal		Total moss		Sphawar		Total moss	
		St	Lt	St	Lt	St	Lt	St	Lt	St	Lt	St	Lt
Weeks Sub	1 : 2	0.437	**0.074**	0.310	**0.010**	1.000	0.836	0.885	0.848	**0.011**	0.057	0.125	0.787
	1 : 4	**0.003**	0.661	**0.002**	0.202	**<0.001**	0.357	**<0.001**	0.567	**<0.001**	0.654	**<0.001**	**0.039**
	1 : 6	0.053	**0.023**	**0.031**	**0.003**	**0.001**	**0.005**	**<0.001**	**0.010**	**<0.001**	**<0.001**	**<0.001**	**0.002**
	1 : 8	**<0.001**	0.001	**<0.001**	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
	2 : 4	**0.014**	0.201	**0.012**	0.179	**0.001**	0.264	**<0.001**	0.702	**0.006**	0.130	**<0.001**	**0.023**
	2 : 6	0.210	0.548	0.199	0.492	**0.001**	**0.003**	**<0.001**	**0.015**	**<0.001**	**0.009**	**0.001**	**0.001**
	2 : 8	**<0.001**	**0.045**	**<0.001**	**0.044**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**0.009**	**<0.001**	**<0.001**
	4 : 6	0.135	0.076	0.122	0.057	0.720	**0.035**	0.885	**0.033**	**0.004**	**<0.001**	0.633	0.144
	4 : 8	**0.078**	**0.005**	0.081	**0.004**	0.063	**0.001**	**0.004**	**0.001**	**<0.001**	**<0.001**	**0.002**	**0.005**
	6 : 8	**0.002**	0.136	**0.002**	0.153	**0.031**	0.070	**0.003**	0.076	0.254	1.000	**0.001**	0.109
Weeks Sub	1	a	A	a	A	a	A	a	A	a	A	a	A
	2	a	Ab	ab	B	a	A	a	A	b	A	a	A
	4	bc	Ab	cd	AB	bc	A	b	A	c	A	b	B
	6	ab	Bc	bc	BC	b	B	b	B	d	B	b	BC
	8	c	C	d	C	c	B	c	B	d	B	c	C

Table A3.3 Table for species and total moss percent cover responses to submergence duration (Weeks Sub) and between short and long-term evaluation periods (Term). ANOVA analyses were generated for species and total moss in HvOM, TnOM, and ApOM plugs. Kruskal-Wallis chi-squared analysis was generated for species and total moss in SwOM because variances were non-constant and data were not normally distributed. Bolded values indicate significant differences between treatment means at α = 0.05.

Plugs	Dep. variable	Effect	df	Sum Sq	Mean Sq	F value	P
HvOM	*Hamatocaulis vernicosus*	Weeks Sub	4	851.4	212.9	2.040	0.1140
		Term	1	36.1	36.10	0.346	0.5608
		Weeks Sub*Term	4	794.4	198.6	1.903	0.1357
		Total	30	3130.5	104.4		
	Total moss	Weeks Sub	4	192.1	48.02	0.491	0.7424
		Term	1	422.5	422.5	4.318	**0.0464**
		Weeks Sub*Term	4	489.5	122.4	1.251	0.3110
		Total	30	2935.5	97.9		
TnOM	*Tomentypnum nitens*	Weeks Sub	4	26480.6	6620.1	30.304	**<0.0001**
		Term	1	2213.2	2213.2	10.131	**0.0036**
		Weeks Sub*Term	4	1837.0	459.2	2.102	0.10715
		Total	28	6116.7	218.5		
	Total moss	Weeks Sub	4	26688.8	6672.2	29.274	**<0.0001**
		Term	1	2464.1	2464.1	10.811	**0.0027**
		Weeks Sub*Term	4	1954.4	488.6	2.144	0.1017
		Total	28	6381.9	227.9		
ApOM	i	Weeks Sub	4	31567.8	7892.0	37.898	**<0.0001**
		Term	1	469.2	469.2	2.253	0.1438
		Weeks Sub*Term	4	3146.6	786.7	3.778	**0.0133**
		Total	30	6247.2	208.2		
	Total moss	Weeks Sub	4	31203.9	7801.0	42.810	**<0.0001**
		Term	1	2975.6	2975.6	16.329	**0.0003**
		Weeks Sub*Term	4	5950.5	1487.6	8.164	**0.0001**
		Total	30	5466.7	182.2		
Plugs	**Dep. variable**	**Effect**	**$\chi 2$**	**df**	**P**		
SwOM	*Sphagnum warnstorfii*	Term	11.45	1	**0.0007**		
	Total moss	Term	0.36	1	0.5502		

Table A3.4 Comparison of a Tukey-adjusted least squares means test for species and total moss percent cover responses to submergence duration (Weeks Sub) between short (St) and long-term (Lt) evaluation periods. Comparisons shown for ANOVA model results that where significant at the α = 0.05 level. Bolded values indicate significant differences between treatment means at α = 0.05. Tomenit = *Tomentypnum nitens*, Aulapal = *Aulacomnium palustre.*

			HvOM	TnOM		ApOM	
Effect	Weeks Sub	Comparison	Total moss	Tomenit	Total moss	Aulapal	Total moss
Term	1	St : Lt	**0.0196**	0.2417	0.2004	0.7840	0.2003
	2	St : Lt	0.9717	0.6361	0.6598	0.2301	0.2690
	4	St : Lt	0.1182	**0.0107**	**0.0108**	**0.0005**	**<0.0001**
	6	St : Lt	0.4378	0.8682	0.9076	0.4535	**0.0447**
	8	St : Lt	0.8594	**0.0048**	**0.0038**	0.8459	0.7555

APPENDIX IV

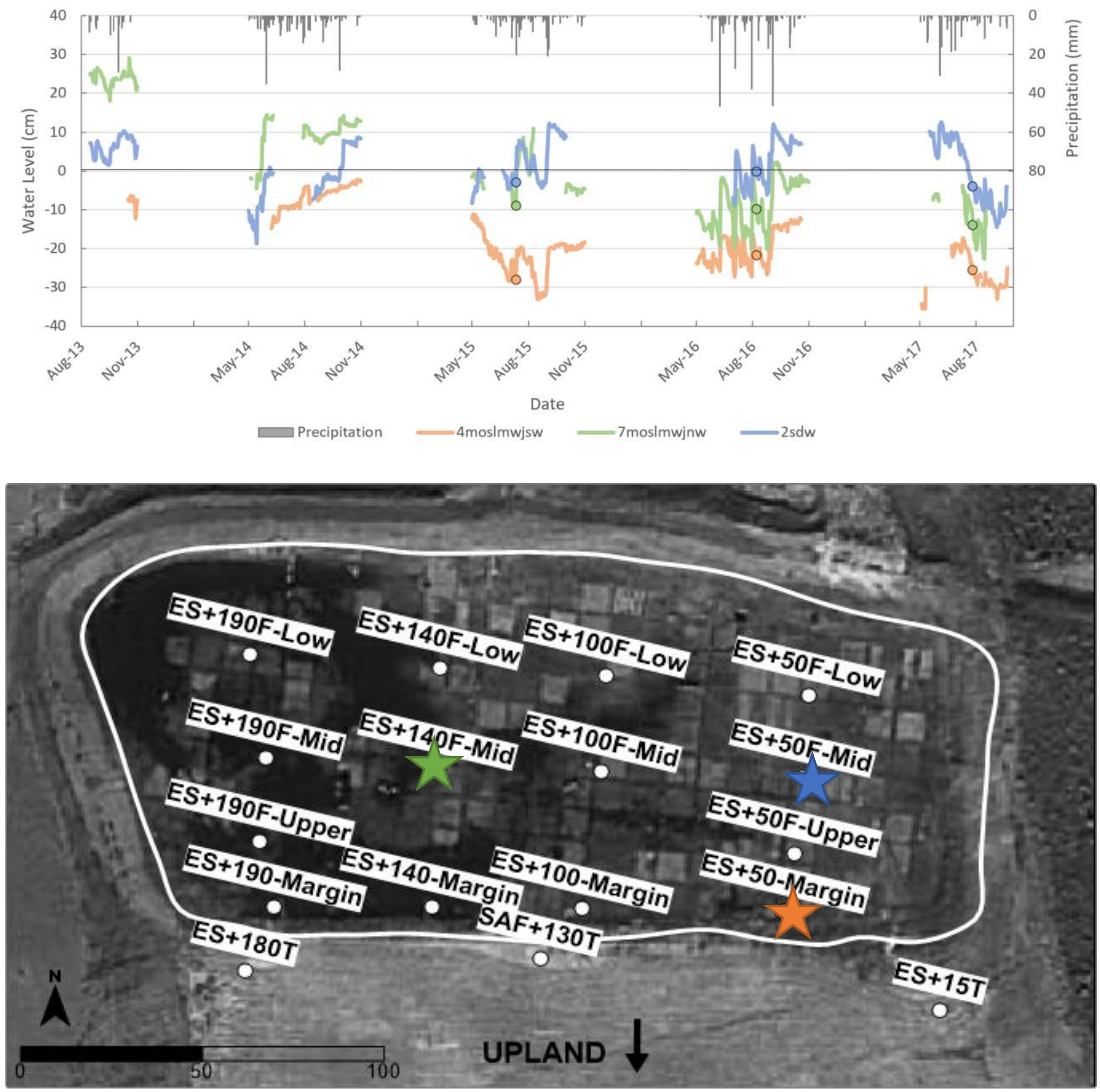


Figure A4.1. Season and annual depth to water table variation from 2013 to 2017 for three plots selected as examples that represent a range in water table depths (Top). Continuous measurements of water level relative to the soil surface (0 cm) were recorded with a waterlogger in three wells across the site (Bottom). Plot level depth to water table relative to soil surface was corrected using elevation and hand-collected measurements by year. Daily precipitation data was collected onsite using a rain gauge.

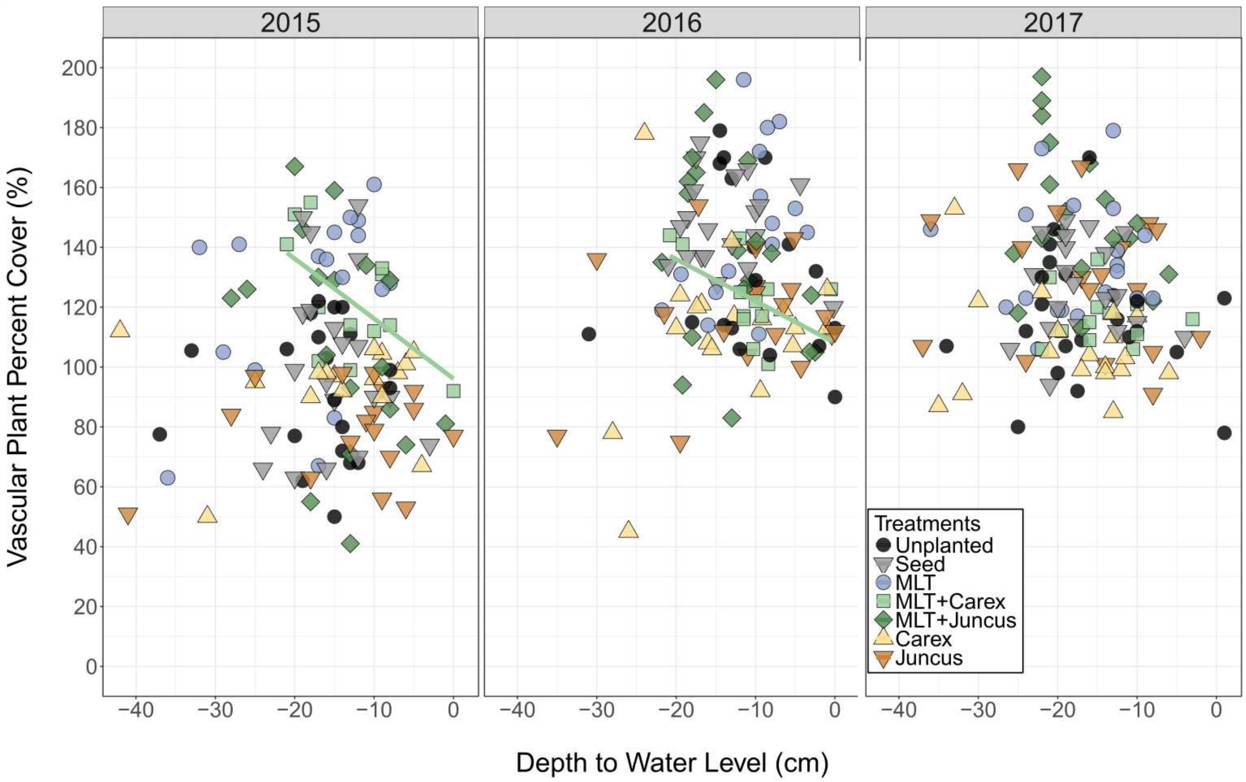


Figure A4.2. Effects of depth to water level and planting treatments on the vascular plant percent cover from 2015 to 2017. Significant linear regressions relationships between percent cover and depth to water level are presented for each planting treatment by color.

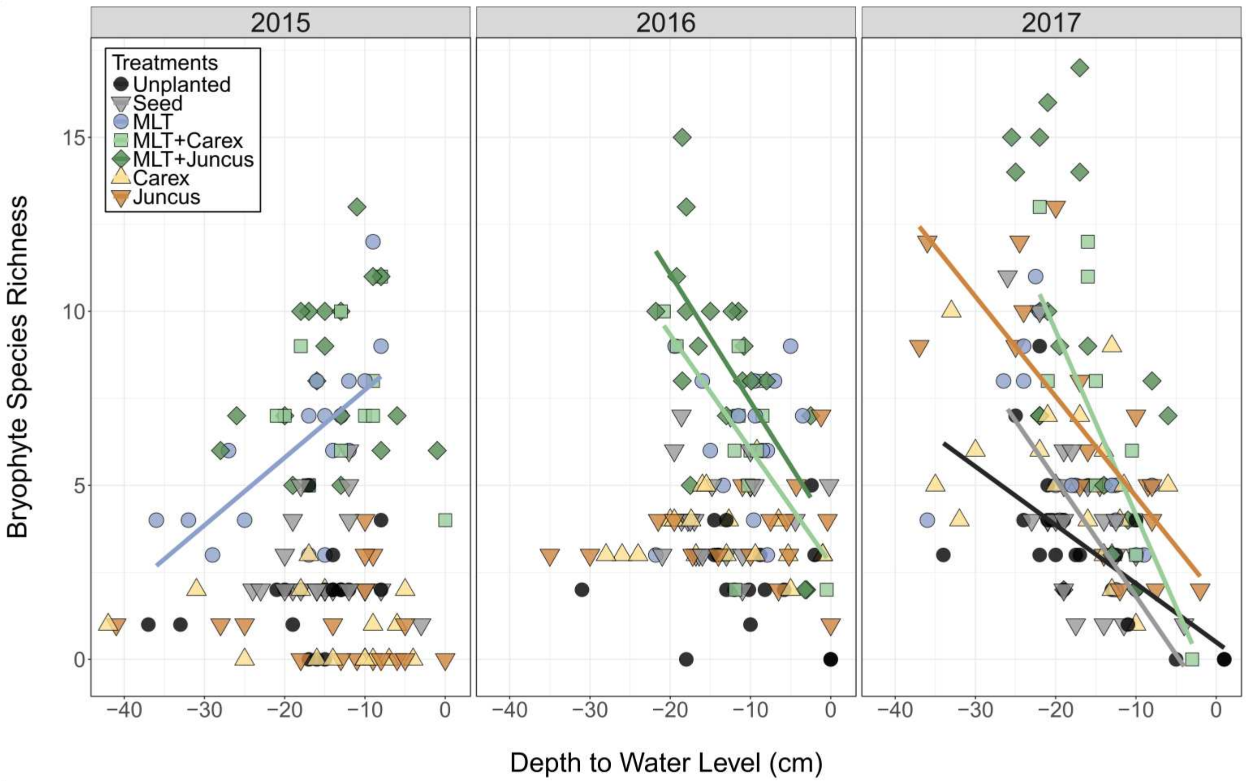


Figure A4.3 Effects of depth to water level and planting treatments on bryophyte species richness from 2015 to 2017. Significant linear regressions relationships between species richness and depth to water level are presented for each planting treatment by color.

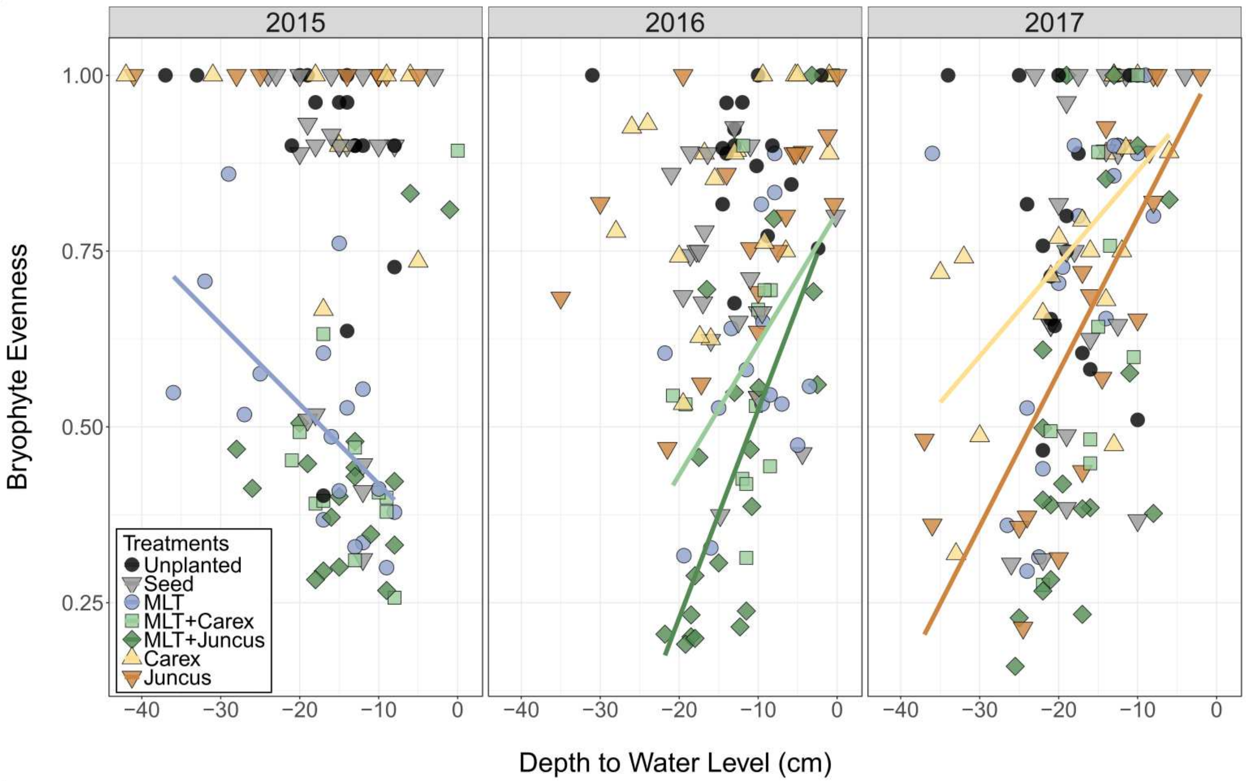


Figure A4.4. Effects of depth to water level and planting treatments on bryophyte evenness from 2015 to 2017. Significant linear regressions relationships between evenness and depth to water level are presented for each planting treatment by color.

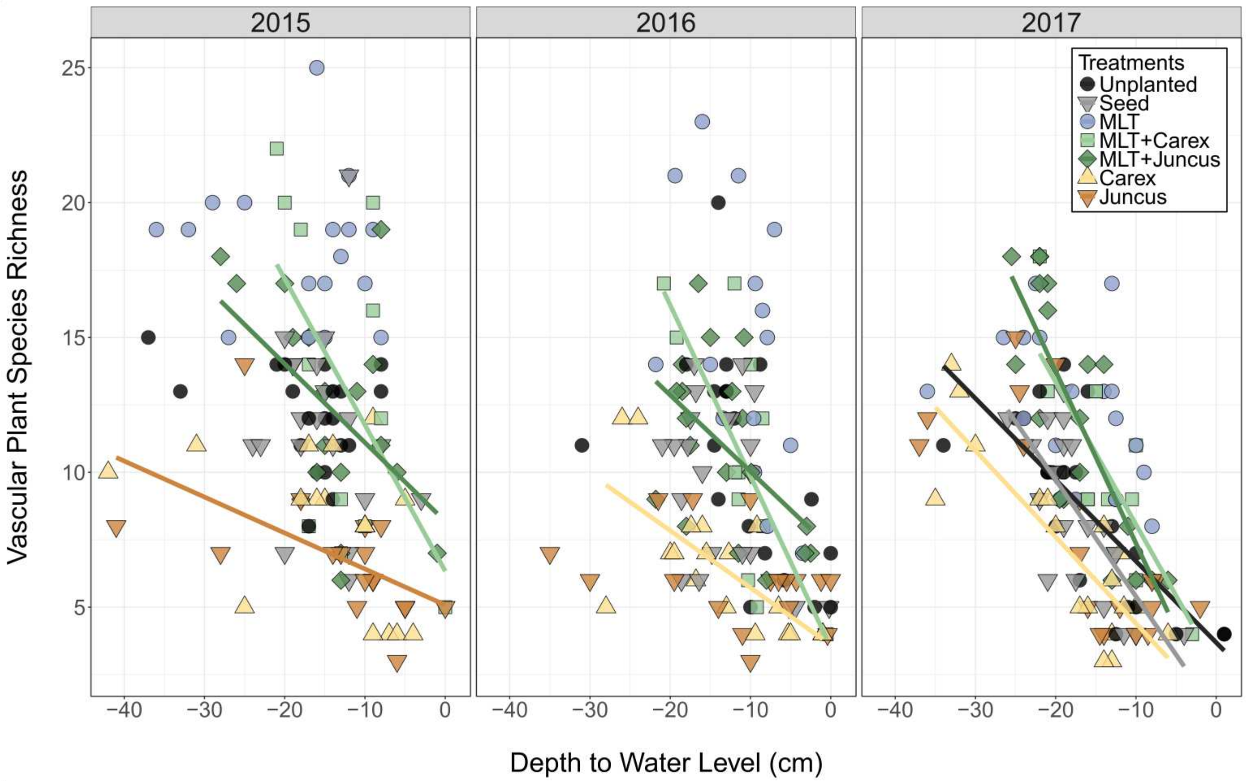


Figure A4.5. Effects of depth to water level and planting treatments on the vascular plant species richness from 2015 to 2017. Significant linear regressions relationships between species richness and depth to water level are presented for each planting treatment by color.

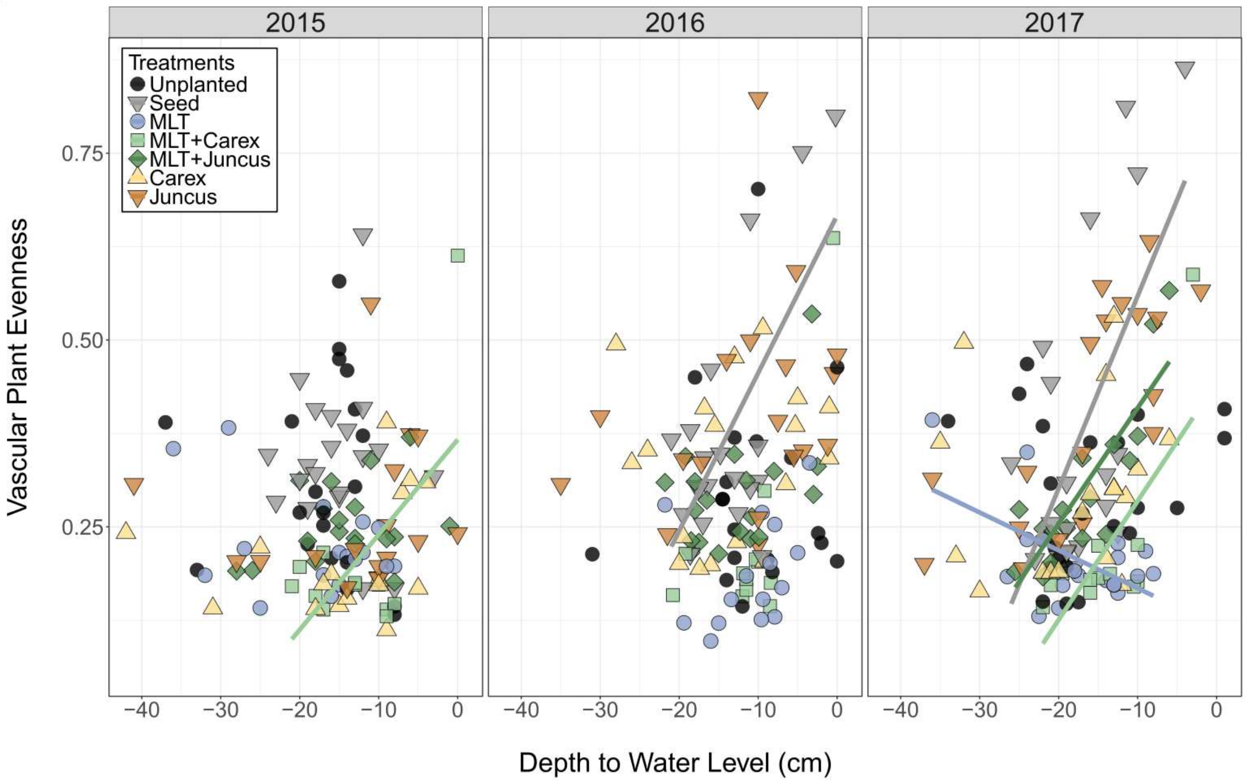


Figure A4.6. Effects of depth to water level and planting treatments on the vascular plant evenness from 2015 to 2017. Significant linear regressions relationships between evenness and depth to water level are presented for each planting treatment by color.

Table A4.1 Monthly average air temperatures and cumulative precipitation amounts from 2013 to 2017 on the Nikanotee Fen on the Suncor Millennium Mine north of Fort MacMurray, Alberta. Standard deviations in parenthesis.

Month	2013	2014	2015	2016	2017
Average Air Temperature (Celsius)					
January		-14.9 (6.1)	-13.4 (7.4)	-13.0 (6.4)	-12.0 (7.5)
February		-16.4 (3.7)	-14.7 (5.2)	-10.2 (5.0)	-9.4 (8.0)
March	1.4 (4.5)	-8.8 (6.9)	-2.4 (6.2)	-2.2 (3.6)	-6.9 (8.5)
April	0.6 (4.0)	2.5 (5.9)	6.1 (5.3)	5.5 (6.2)	3.8 (4.2)
May	15.6 (5.3)	10.2 (5.8)	13.8 (6.0)	15.0 (4.1)	14.2 (4.1)
June	18.8 (4.2)	18.2 (4.1)	19.5 (3.6)	19.6 (3.6)	18.6 (3.4)
July	19.5 (4.1)	22.1 (2.6)	20.4 (2.6)	20.9 (2.6)	21.4 (2.5)
August	19.6 (2.2)	19.6 (4.6)	19.1 (3.3)	18.7 (3.3)	18.7 (2.9)
September	14.9 (4.5)	10.6 (3.9)	10.3 (2.6)	12.3 (2.2)	13.6 (3.8)
October	4.2 (3.5)	5.3 (3.7)	5.9 (4.3)	1.4 (2.0)	NA
November	-8.8 (6.5)	-10.9 (7.1)	-5.3 (5.9)	-1.9 (4.3)	NA
December	-17.9 (3.8)	-13.2 (5.3)	-11.2 (5.6)	-15.3 (5.8)	NA
Precipitation (mm)					
January	13.3	8.6	8.8	13.2	14.6
February	2.7	5.5	2.1	7.4	2.8
March	4.6	8.5	13.4	4.1	1.9
April	23.8	10.6	2.6	15	6
May	5.9	73.8	17.8	19.1	36.7
June	139	47.1	29	82	102.1
July	91.3	65.8	68.9	102.8	25.8
August	25.7	41	37.9	53.3	29.6
September	47.8	53.6	49.4	73.5	16.2
October	19.2	9.2	13.4	19.7	13.9
November	7.6	0.3	6.1	2.5	NA
December	9.5	5.6	1.6	2	NA
Totals	**390.4**	**329.6**	**251**	**394.6**	**249.6**

Table A4.2 Water and soil chemistry of the Nikanotee Fen (NF) and donor rich-fen (Donor) where the moss layer transfer material was harvested. Water samples were collected from dugout pits and soil samples from the surface on August 8, 2014 and July 11, 2015. The value presented is the mean from *n* samples with standard deviations in italics. Stared values are of total oxidized nitrogen ($NO_2^- + NO_3^-$). ND = Not detected.

Sample	**Site**	**Date**	**EC (uS/cm)**	**pH**	**Cl^- (mg/L)**	**SO_4^{-2} (mg/L)**	**NO_3^- (mg/L)**	**Na^+ (mg/L)**	**K^+ (mg/L)**	**Mg^{2+} (mg/L)**	**Ca^{2+} (mg/L)**
Water	Donor	14-Aug	341	6.88	4.17	4.24	0.16	8.96	5.16	10.55	26.72
	(n=3)		*125*	*0.15*	*1.07*	*3.19*	*0.02*	*1.78*	*6.49*	*1.10*	*0.94*
	Donor	15-Jul	254	6.94	2.41	1.97	0.20*	6.77	1.45	9.11	39.15
	(n=3)		*38*	*0.06*	*1.60*	*0.92*	*0.12*	*2.91*	*0.83*	*5.55*	*19.76*
Water	NF	14-Aug	2602	8.11	37.91	589.00	ND	115.78	8.07	46.02	113.19
	(n=3)		*178*	*0.07*	*7.88*	*50.69*		*29.04*	*1.96*	*13.90*	*10.94*
	NF	15-Jul	2711	6.66	24.24	829.07	0.09*	121.63	1.74	82.09	285.44
	(n=45)		*833*	*0.17*	*20.28*	*531.34*	*0.04*	*65.37*	*1.25*	*32.42*	*148.95*
Soil	NF	14-Aug	3160	7.48	43.60	1893.00	0.77	178.60	22.23	162.10	467.00
	(n=10)		*1061*	*0.12*	*24.82*	*725.60*	*0.71*	*113.32*	*10.83*	*79.99*	*127.98*
	NF	15-Jul	4242	NA	63.09	2450.55	20.90	398.89	9.06	229.29	673.81
	(n=45)		*1350*	*NA*	*53.74*	*617.35*	*22.91*	*291.57*	*7.27*	*75.50*	*135.90*

Table A4.3 Hydrologic parameters from 2013 to 2017 for three plots selected as examples that represent a range in water table depths. Standard deviations in parenthesis.

	Plots	2013	2014	2015	2016	2017
Cumulative Precipitation from May-Sept (cm)		309.7	281.3	203	330.7	210.4
Average seasonal depth to water table (cm)	4moslmwjsw	-7.99 (0.39)	-6.80 (0.66)	-21.74 (0.36)	-19.64 (0.33)	-26.00 (0.49)
	7moslmwjnw	23.86 (0.22)	9.49 (0.39)	-1.61 (0.50)	-8.50 (0.45)	-11.90 (0.76)
	2sdw	6.18 (0.29)	-1.77 (0.62)	2.28 (0.51)	2.57 (0.53)	-0.21 (0.73)
Maximum seasonal depth to water table (cm)	4moslmwjsw	-6.41	-2.27	-11.06	-12.17	-17.21
	7moslmwjnw	29.03	14.47	10.85	2.32	-3.74
	2sdw	10.32	8.64	12.19	12.14	12.52
Minimum seasonal depth to water table (cm)	4moslmwjsw	-12.34	-15.13	-33.08	-27.29	-35.49
	7moslmwjnw	18.15	-5.10	-9.42	-20.31	-22.68
	2sdw	1.54	-18.76	-8.36	-8.98	-14.43
Total number of days water table was over 0 cm	4moslmwjsw	0	0	0	0	0
	7moslmwjnw	73	116	23	13	0
	2sdw	78	38	63	73	63
Total number of days water table was under 0 cm	4moslmwjsw	16	148	184	172	95
	7moslmwjnw	0	11	65	171	50
	2sdw	0	81	61	39	64

Table A4.4 Effects of planting treatment (Trt), mulchweed (MW) or mulch (M), and their interactions on bryophyte percent cover across years and in each year from 2014 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

	Bryophyte Cover								
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Effect**	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Date	3	128	24.53	**<.0001**	Date	3	153	24.93	**<.0001**
Trt	6	32.9	18.51	**<.0001**	Trt	6	30.5	14.45	**<.0001**
Date*Trt	18	137	4.88	**<.0001**	Date*Trt	18	162	5.25	**<.0001**
MW	3	99	1.65	0.1822	M	1	116	0.55	0.4617
Date*MW	9	129	4.79	**<.0001**	Date*M	3	152	11.31	**<.0001**
Trt*MW	18	96.9	1.77	**0.0396**	Trt*M	6	115	3.06	**0.0081**
Date*Trt*MW	54	140	1.16	0.2407	Date*Trt*M	18	161	1.74	0.0369
2014					**2014**				
Trt	6	14	20.63	**<.0001**	Trt	6	15.2	12.58	**<.0001**
MW	3	44.2	32.62	**<.0001**	M	1	50.8	78.4	**<.0001**
Trt*MW	18	39.6	8.55	**<.0001**	Trt*M	6	49.2	22.69	**<.0001**
2015					**2015**				
Trt	6	92.1	31.94	**<.0001**	Trt	6	106	32.32	**<.0001**
MW	3	89.3	0.26	0.8543	M	1	103	0.56	0.4543
Trt*MW	18	89.1	0.9	0.583	Trt*M	6	103	0.47	0.8256
2016					**2016**				
Trt	6	22.9	6.58	**0.0004**	Trt	6	23.1	6.76	**0.0003**
MW	3	72.1	1.15	0.3331	M	1	86.9	0.77	0.3817
Trt*MW	18	70.7	1.52	0.1104	Trt*M	6	85.6	2.87	0.0136
2017					**2017**				
Trt	6	26.3	5.35	**0.001**	Trt	6	26.4	5.29	**0.0011**
MW	3	77.6	1.3	0.282	M	1	91.2	0.03	0.8574
Trt*MW	18	76.2	0.91	0.5638	Trt*M	6	89.8	1.2	0.3129

Table A4.5 Effects of depth to water level (DWL) in each planting treatment on bryophyte, vascular plant, *Carex aquatilis*, and *Juncus balticus* percent cover in each year from 2015 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

Effect DWL	**Bryophyte Cover**				**Vascular Plant Cover**				***Carex aquatilis* Cover**				***Juncus balticus* Cover**			
Trt	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**
2015																
Unplanted	1	16	0.09	0.767	1	15.4	0.03	0.8597	1	17.4	1.78	0.1992	1	17.6	1.21	0.286
MLT	1	15	2.94	0.1071	1	12.1	1.16	0.3032	1	15	1.4	0.2544	1	13.9	0.18	0.6776
MLT + *Carex*	1	10	0.47	0.5101	1	9.1	11.42	**0.008**	1	9.8	1.42	0.2617	1	8.95	3.21	0.1068
MLT + *Juncus*	1	16	10.17	**0.0057**	1	15.7	2.88	0.1094	1	13.3	2.08	0.1728	1	14	0.42	0.5271
Seed	1	17.6	0	0.9587	1	14.6	0.52	0.4818	1	15.2	2.69	0.1215	1	18	0.02	0.8835
Carex	1	7.01	11.26	**0.0121**	1	14	0.13	0.7272	1	14	2.9	0.1108	1	11.2	4.91	**0.0484**
Juncus	1	15	5.02	**0.0407**	1	15	0.64	0.4362	1	5.28	0.75	0.4231	1	15	2.61	0.1273
2016																
Unplanted	1	16.4	0.1	0.7608	1	16.7	0.43	0.521	1	12.9	1.48	0.2451	1	16.2	0.44	0.5181
MLT	1	13	3.27	0.0939	1	12.7	0.11	0.7455	1	13	13.09	**0.0031**	1	4.58	6.3	0.0582
MLT + *Carex*	1	9.66	2.71	0.1316	1	9.94	5.2	**0.0459**	1	10	0.67	0.4332	1	10	0.65	0.4398
MLT + *Juncus*	1	11.4	8.31	**0.0144**	1	12.8	2.31	0.1528	1	17	3.94	0.0637	1	11.5	0.14	0.7167
Seed	1	18	0.1	0.7511	1	15.2	0.11	0.7427	1	15.8	4.59	**0.0481**	1	18	0.56	0.4654
Carex	1	7.83	3.11	0.1166	1	16	0.57	0.4599	1	11.7	7.29	**0.0197**	1	16	1.19	0.2919
Juncus	1	13.5	21.05	**0.0005**	1	16	0.78	0.3888	1	9.12	2.05	0.186	1	15.8	3.01	0.1022
2017																
Unplanted	1	19.6	0	0.9743	1	18.5	0.28	0.6016	1	12.4	1.5	0.2442	1	15.3	0.94	0.3486
MLT	1	16.7	2.63	0.1236	1	14.5	0.64	0.4366	1	14.1	22.52	**0.0003**	1	16.6	0.27	0.607
MLT + *Carex*	1	10	5.65	**0.0389**	1	10	0.04	0.8392	1	10	0.09	0.7761	1	9.96	0.47	0.5083
MLT + *Juncus*	1	16.5	0.88	0.362	1	17	2.33	0.1453	1	6.73	0.27	0.6177	1	17	0.65	0.4314
Seed	1	18	1.97	0.1771	1	17.5	0.38	0.5444	1	18	1.47	0.2409	1	18	1.24	0.2795
Carex	1	16	5.94	**0.0269**	1	16	1.1	0.3108	1	16	5.29	**0.0353**	1	16	5.05	**0.0391**
Juncus	1	16	6.12	**0.025**	1	16	0.7	0.4162	1	5.14	3.65	0.1127	1	1.82	2.89	0.2435

Table A4.6 Effects of planting treatment (Trt), mulchweed (MW), and their interactions on vascular plant, *Carex aquatilis*, and *Juncus balticus* percent cover across years and in each year from 2014 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

	Vascular Plant Cover					***Carex aquatilis* Cover**					***Juncus balticus* Cover**			
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Effect**	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Effect**	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Date	3	182	124.6	**<.0001**	Date	3	150	80.08	**<.0001**	Date	3	161	14.6	**<.0001**
Trt	6	28	8	**<.0001**	Trt	6	33	13.17	**<.0001**	Trt	6	25.6	277.99	**<.0001**
Date*Trt	18	174	5.62	**<.0001**	Date*Trt	18	142	5.99	**<.0001**	Date*Trt	18	166	2.29	**0.0033**
MW	3	75	0.98	0.4082	MW	3	81.3	0.45	0.7209	MW	3	76.8	4.65	**0.0049**
Date*MW	9	175	0.47	0.8965	Date*MW	9	142	0.77	0.6455	Date*MW	9	148	1.26	0.2615
Trt*MW	18	71.9	0.65	0.8475	Trt*MW	18	80	1.24	0.2548	Trt*MW	18	74	2.49	0.0032
Date*Trt*MW	54	179	0.76	0.8762	Date*Trt*MW	54	142	0.74	0.9012	Date*Trt*MW	54	161	0.98	0.5282
2014					**2014**					**2014**				
Trt	6	14.6	4.5	**0.0089**	Trt	6	14.8	12.8	**<.0001**	Trt	6	17.2	30.42	**<.0001**
MW	3	45.7	0.3	0.8272	MW	3	46.3	0.31	0.8213	MW	3	40	5.94	0.0019
Trt*MW	18	42	1.21	0.3003	Trt*MW	18	42.5	1.12	0.365	Trt*MW	18	36.6	8.32	**<.0001**
2015					**2015**					**2015**				
Trt	6	25.3	4.16	**0.0048**	Trt	6	24.8	15.53	**<.0001**	Trt	6	29	111.05	**<.0001**
MW	3	72.3	1.22	0.3102	MW	3	70.1	0.2	0.8945	MW	3	77	1.96	0.127
Trt*MW	18	71.4	1.13	0.3422	Trt*MW	18	69.1	1.41	0.1539	Trt*MW	18	75.6	1.47	0.124
2016					**2016**					**2016**				
Trt	6	20.2	3.18	**0.023**	Trt	6	23.1	8.41	**<.0001**	Trt	6	23.1	74.07	**<.0001**
MW	3	70.7	0.74	0.5341	MW	3	72.9	1.39	0.2532	MW	3	71	4.45	**0.0063**
Trt*MW	18	69.4	1.19	0.2968	Trt*MW	18	71.5	1.47	0.1271	Trt*MW	18	69.8	1.94	**0.0259**
2017					**2017**					**2017**				
Trt	6	21.5	8.31	**<.0001**	Trt	6	22.4	5.9	**0.0008**	Trt	6	23.6	148.21	**<.0001**
MW	3	77	2.14	0.1024	MW	3	78.5	0.99	0.4011	MW	3	77.1	5.42	**0.0019**
Trt*MW	18	75.1	1.11	0.3634	Trt*MW	18	76.6	0.94	0.5331	Trt*MW	18	75.3	1.94	**0.0248**

Table A4.7 Effects of planting treatment (Trt), mulchweed (MW) or weed (W), and their interactions on *Typha latifolia* percent cover across years and in each year from 2014 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

	***Typha latifolia* Cover**								
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Effect**	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Date	3	170	11.32	**<.0001**	Date	3	202	12.27	**<.0001**
Trt	6	58.3	3.41	**0.006**	Trt	6	60.5	3.4	**0.0059**
Date*Trt	18	169	2.22	**0.0046**	Date*Trt	18	200	2.4	**0.0017**
MW	3	92	8.74	**<.0001**	W	1	109	15.78	**0.0001**
Date*MW	9	169	2.49	**0.0106**	Date*W	3	200	7.46	**<.0001**
Trt*MW	18	91	1.54	0.0942	Trt*W	6	108	2.26	**0.0432**
Date*Trt*MW	54	175	0.81	0.8207	Date*Trt*W	18	199	1.8	**0.0278**
2014					**2014**				
Trt	6	22.1	1.18	0.3506	Trt	6	24	1.3	0.2944
MW	3	48.4	1.49	0.2283	W	1	60.6	3.11	0.0827
Trt*MW	18	46	0.56	0.9102	Trt*W	6	59.1	0.75	0.6157
2015					**2015**				
Trt	6	31.5	2.09	0.0819	Trt	6	28.7	2.15	0.0773
MW	3	77.2	13.87	**<.0001**	W	1	93.1	27.63	**<.0001**
Trt*MW	18	75.7	1.75	**0.0482**	Trt*W	6	92	2.25	**0.0452**
2016					**2016**				
Trt	6	22.3	2.86	**0.0321**	Trt	6	22.4	2.91	**0.0299**
MW	3	70.7	8.27	**<.0001**	W	1	84.5	16.9	**<.0001**
Trt*MW	18	69.1	1.99	**0.0222**	Trt*W	6	83	3.85	**0.0019**
2017					**2017**				
Trt	6	26.1	1.88	0.1221	Trt	6	26.1	1.78	0.1425
MW	3	79.2	4.46	**0.006**	W	1	95.1	5.44	**0.0218**
Trt*MW	18	77.4	1.08	0.3892	Trt*W	6	93.6	0.86	0.5245

Table A4.8 Effects of depth to water level (DWL) in each planting treatment on *Typha latifolia* percent cover in each year from 2015 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

Effect DWL	***Typha latifolia* Cover**			
Trt	**N DF**	**D DF**	**FValue**	**Pr > F**
2015				
Unplanted	1	18	0.55	0.469
MLT	1	14.8	0.57	0.4609
MLT + *Carex*	1	9.86	8.3	**0.0166**
MLT + *Juncus*	1	7.56	2.18	0.1804
Seed	1	18	2.37	0.1411
Carex	1	5.13	0.45	0.5335
Juncus	1	15	1.43	0.2511
2016				
Unplanted	1	17	5.79	**0.0278**
MLT	1	11.4	5.48	**0.0382**
MLT + *Carex*	1	7.53	27.43	**0.001**
MLT + *Juncus*	1	12.8	2.82	0.1175
Seed	1	17.5	2.66	0.1208
Carex	1	7.09	3.41	0.1069
Juncus	1	16	1.68	0.2132
2017				
Unplanted	1	20	7.04	**0.0153**
MLT	1	16.8	0.62	0.4427
MLT + *Carex*	1	9.61	8.22	**0.0174**
MLT + *Juncus*	1	17	6.9	**0.0177**
Seed	1	16	10.58	**0.005**
Carex	1	7.42	0.51	0.4961
Juncus	1	5.48	5.33	0.0644

Table A4.9 Effects of planting treatment (Trt), mulchweed (MW), and their interactions on species richness and evenness of bryophytes and vascular plants across years and in each year from 2014 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

	Bryophyte Species Richness				Bryophyte Evenness				Vascular Plant Species Richness				Vascular Plant Evenness			
Effect	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Num DF**	**Den DF**	**F Value**	**Pr > F**	**Num DF**	**Den DF**	**F Value**	**Pr > F**
Date	3	110	106.19	**<.0001**	3	49.3	1.27	0.2934	3	164	20.25	**<.0001**	3	168	8.23	**<.0001**
Trt	6	29.1	30.44	**<.0001**	6	31.9	14.1	**<.0001**	6	23.5	13.78	**<.0001**	6	26.6	5.8	**0.0006**
Date*Trt	18	123	4.63	**<.0001**	17	60	3.37	**0.0003**	18	155	4.36	**<.0001**	18	160	3.18	**<.0001**
MW	3	105	4.36	**0.0062**	3	40.1	1.27	0.2972	3	90.2	6.96	**0.0003**	3	90.2	1.34	0.2654
Date*MW	9	113	0.49	0.8791	9	54	0.81	0.6103	9	158	1.56	0.1328	9	176	0.46	0.8987
Trt*MW	18	101	1.01	0.4583	18	92.8	0.98	0.4927	18	85.7	1.23	0.2532	18	87	1.78	**0.0411**
Date*Trt*MW	54	126	0.79	0.8373	40	100	0.82	0.7545	54	157	0.57	0.9902	54	163	0.62	0.979
2014																
Trt	6	12.1	22.51	**<.0001**	5	12.7	5.02	**0.0093**	6	15.6	13.12	**<.0001**	6	17.2	3.22	**0.0261**
MW	3	44	4.32	**0.0094**	3	12.8	2.14	0.1445	3	45.8	4.13	**0.0112**	3	47.6	0.6	0.6166
Trt*MW	18	38.9	1.96	**0.0394**	5	12.2	0.18	0.9661	18	42	1.77	0.0638	18	44	0.67	0.8225
2015																
Trt	6	23.3	28.39	**<.0001**	6	24.5	14.81	**<.0001**	6	20.3	15.7	**<.0001**	6	19.4	4.24	**0.0069**
MW	3	70.6	1.95	0.1296	3	58.9	0.77	0.5159	3	69.3	10.01	**<.0001**	3	71.9	1.2	0.3172
Trt*MW	18	69.6	1.34	0.1916	17	57.6	0.85	0.6296	18	67.8	1	0.4719	18	70.1	1.13	0.342
2016																
Trt	6	22	14.15	**<.0001**	6	22.6	10.7	**<.0001**	6	23	4.95	**0.0022**	6	28	4.41	**0.0029**
MW	3	69.4	3.09	**0.0325**	3	66.7	2.33	0.082	3	68.9	3.05	**0.0343**	3	73.4	1.27	0.2924
Trt*MW	18	67.9	0.64	0.8518	18	65.8	3.59	**<.0001**	18	67.9	0.99	0.4827	18	71.9	1.45	0.1351
2017																
Trt	6	24.7	7.39	**0.0001**	6	24.8	3.07	**0.0219**	6	24.7	5.2	**0.0014**	6	25.4	4.16	**0.0048**
MW	3	75.1	2.25	0.089	3	73.3	1.02	0.3903	3	75.2	5.07	**0.003**	3	78.2	1.78	0.1586
Trt*MW	18	73.8	1.07	0.3946	18	70.9	0.73	0.7675	18	74	1.4	0.1579	18	76.6	1.67	0.0638

Table A4.10 Effects of depth to water level (DWL) in each planting treatment on bryophyte and vascular plant species richness and evenness in each year from 2015 to 2017. Results from mixed-model type 3 ANOVAs with Satterthwaite's approximation. Bolded values indicate significant differences between treatment means at α = 0.05.

Effect DWL	**Bryophyte Species Richness**				**Bryophyte Evenness**				**Vascular Plant Species Richness**				**Vascular Plant Evenness**			
Trt	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**	**N DF**	**D DF**	**FValue**	**Pr > F**
2015																
Unplanted	1	16.7	0.14	0.7151	1	12.5	0.06	0.8058	1	18	1.9	0.1847	1	16.9	0.47	0.5027
MLT	1	13.7	4.74	**0.0474**	1	15	8.64	**0.0102**	1	15	0.23	0.637	1	15	3	0.1038
MLT + *Carex*	1	9.06	0.04	0.8384	1	8.84	1.23	0.2965	1	8.28	10.96	**0.0102**	1	9.08	5.41	**0.0448**
MLT + *Juncus*	1	14.8	4.52	0.0507	1	16	2.79	0.1143	1	15.8	8.37	**0.0107**	1	16	1.72	0.2086
Seed	1	15.1	0.28	0.603	1	15.1	0.02	0.9019	1	17.3	0.23	0.6377	1	17.8	0.5	0.4887
Carex	1	6.91	1.74	0.2291	1	3.21	0.5	0.5261	1	5.24	3.24	0.1291	1	6.32	0.82	0.3973
Juncus	1	5.59	0.05	0.8362	NA	NA	NA	NA	1	15	6.97	**0.0185**	1	5.72	0	0.9611
2016																
Unplanted	1	11.6	0.14	0.7152	1	12.2	0.57	0.4643	1	13.1	2.3	0.1529	1	17	0.08	0.7805
MLT	1	5.42	0.3	0.6067	1	6.06	1.44	0.2754	1	11.5	0.96	0.3465	1	11.9	0	0.9464
MLT + *Carex*	1	10	8.09	**0.0174**	1	9.97	7.91	**0.0185**	1	8.21	14.53	**0.0049**	1	2.26	7.34	0.0996
MLT + *Juncus*	1	17	14.1	**0.0016**	1	14.4	14.4	**0.0019**	1	9.74	5.16	**0.047**	1	12.8	3.93	0.0694
Seed	1	15.1	0.2	0.6644	1	17.7	1.37	0.2568	1	14.9	0.71	0.413	1	15.4	17.64	**0.0007**
Carex	1	16	0.11	0.7447	1	13.1	2.16	0.1656	1	16	14.47	**0.0016**	1	7.54	0.21	0.6632
Juncus	1	14.1	0	0.993	1	8.78	0.34	0.5721	1	16	1.44	0.248	1	4.28	1.38	0.3014
2017																
Unplanted	1	9.59	9.05	**0.0138**	1	16.9	0	0.9656	1	10.9	11.71	**0.0058**	1	8.72	0.14	0.7132
MLT	1	16.3	2.81	0.1127	1	14.7	1.28	0.2755	1	16.6	2.01	0.1746	1	16	9.16	**0.008**
MLT + *Carex*	1	8.55	5.73	**0.0417**	1	5.82	0.53	0.4937	1	10	15.28	**0.0029**	1	9.62	10.59	**0.0091**
MLT + *Juncus*	1	16.8	1.43	0.2483	1	16.9	0.91	0.3537	1	17	31.91	**<.0001**	1	7.37	44.25	**0.0002**
Seed	1	17.3	6.05	**0.0247**	1	18	1.11	0.3063	1	17.4	14.77	**0.0013**	1	17.9	11.17	**0.0037**
Carex	1	7.08	2.21	0.1803	1	16	9.41	**0.0074**	1	3.22	32.83	**0.0086**	1	4.77	0.13	0.7325
Juncus	1	16	23.61	**0.0002**	1	16	26.04	**0.0001**	1	5.47	5.16	0.0677	1	6.33	5.73	0.0515

APPENDIX V

Table A5.1 Proportions of species introduced (Prop Initial) in 2012 at the Sandhill Fen (SF) and in 2013 at the Nikanotee Fen (NF) compared to the species proportion within total plots surveyed in 2017 (Prop Final) within each species introduction method. Three seedlings per square meter of *Carex aquatilis* were planted in the NF Carex plots and of *Juncus balticus* in the NF Juncus plots. Proportions in 2017 determined by rank abundance analysis.

Species	Prop Initial	Prop Final	Species	Prop Initial	Prop Final
SF Seeded			**NF Seeded**		
Carex aquatilis	80	31.3	*Betula pumila*	27	NA
Carex diandra	5	NA	*Calamagrostis inexpansa*	17	1.2
Carex utriculata	5	5.1	*Carex aquatilis*	31	43.7
Scirpus atrocinctus	5	<1	*Juncus balticus*	5	12.7
Carex bebbii	1	<1	*Vaccinium oxycoccos*	1	NA
Carex paupercula	1	NA	*Sarracenia purpurea*	12	NA
Scirpus microcarpus	1	<1	*Triglochin maritima*	7	12.8
Carex lasiocarpa	<1	NA	**NF Carex**		
Carex rostrata	<1	NA	*Carex aquatilis*	100	68.4
Carex limosa	<1	NA	**NF Juncus**		
Carex interior	<1	NA	*Juncus balticus*	100	39.5
Juncus tenuis	<1	NA			

Table A5.2 Proportions of species introduced (Prop Initial) in 2013 at the Nikanotee Fen (NF) in plots treated with the moss layer transfer material (MLT) compared to the species proportion within total plots surveyed in 2017 (Prop Final). The initial proportions of species in the MLT material were determined prior to harvest during a vegetation survey conducted in 1 m^2 plots along 3 transects on May 27-28, 2013. The MLT material was spread at a 1:10 ratio. Three seedlings per square meter of *Carex aquatilis* were planted in the Moss Carex plots and of *Juncus balticus* in the Moss Juncus plots. Proportions in 2017 determined by rank abundance analysis.

Species	**Prop Initial**	**Prop Final**		
	Harvested MLT	**NF Moss**	**NF Moss Carex**	**NF Moss Juncus**
Juncus balticus	NA	2.9	1.3	25.2
Tomentypnum nitens	34.6	<1	<1	5.2
Betula pumila	12.1	<1	NA	<1
Sphagnum angustifolium	11.4	NA	NA	NA
Aulacomnium palustre	5.9	<1	NA	<1
Vaccinium.oxycoccos	5.6	<1	NA	<1
Carex aquatilis	3.9	53.2	63.7	34.8
Sphagnum fuscum	2.8	NA	NA	NA
Sphagnum warnstorfii	2.6	NA	NA	NA
Sphagnum capillifolium	2.4	NA	NA	NA
Larix laricina	2.4	NA	NA	NA
Calliergon giganteum	1.8	<1	<1	<1
Hamatocaulis vernicosus	1.6	<1	<1	<1
Salix pedicellaris	1.5	<1	NA	<1
Carex gynocrates	1.3	NA	NA	NA
Ptychostomum pseudotriquetrum	1.2	5.6	3.9	7.8
Carex prairea	1.2	3.5	1.4	1.7
Ledum groenlandicum	1	NA	NA	NA
Salix planifolia	1	NA	NA	<1
Plagiomnium.ellipticum	<1	<1	<1	<1
Carex limosa	<1	NA	NA	NA
Salix glauca	<1	NA	NA	NA
Drepanocladus aduncus	<1	<1	<1	1
Smilacina trifolia	<1	NA	NA	NA
Pohlia nutans	<1	NA	NA	NA
Pyrola asarifolia	<1	NA	NA	NA
Sphagnum magellanicum	<1	NA	NA	NA
Menyanthes trifolia	<1	<1	<1	NA
Potentilla palustrus	<1	<1	<1	<1
Stellaria longifolia	<1	<1	NA	NA
Galium trifidum	<1	<1	<1	<1
Oxycoccos quadripetalus	<1	NA	NA	NA
Carex leptalea	<1	NA	NA	NA
Polytricum strictum	<1	NA	NA	NA

Andromeda polifolia	<1	NA	NA	NA
Picea mariana	<1	NA	NA	NA
Carex disperma	<1	NA	NA	NA
Salix candina	<1	<1	NA	NA
Sphagnum teres	<1	NA	NA	NA
Viola sp.	<1	NA	NA	NA
Drosera rotundifolia	<1	NA	NA	NA
Campilium stellatum	<1	<1	<1	<1

Table A5.3 Species list of Desirable (D) and Undesirable (U) peatland plants and their count occurrences in plots at the Sandhill (n=79) and Nikanotee (n=64) Fens. Surveys were conducted at the Sandhill Fen on July 12-13, 2017 and at the Nikanotee Fen from July 20-25, 2017. Desirable and Undesirable peatland plant classifications derived by M. House and D.H. Vitt.

Species	Desirable/ Undesirable	Number of occurrences	
		SF	NF
Achillea millefolium	U	11	0
Agropyron trachycaulum	U	2	0
Agrostis scabra	U	4	7
Alopecurus arundinaceus	U	0	1
Aster conspicuus	U	10	0
Aster puniceus	U	7	1
Betula pumila	D	2	3
Bromus ciliatus	U	4	0
Calamagrostis canadensis	U	51	0
Calamagrostis inexpansa	D	0	21
Caltha palustris	D	0	2
Carex aquatilis	D	59	64
Carex atherodes	D	2	8
Carex aurea	D	1	0
Carex bebbii	D	7	0
Carex canescens	D	1	0
Carex diandra	D	0	17
Carex hystericina	D	1	0
Carex interior	D	0	4
Carex prairea	D	0	16
Carex pseudocyperus	D	1	0
Carex utriculata	D	32	22
Chenopodium alba	U	0	1
Cicuta bulbifera	D	1	0
Cicuta maculata	D	2	3
Crepis tectorum	U	0	15
Circium arvense	U	5	0
Cornus stolonifera	U	1	0
Dasiphora fruticosa	U	3	0
Deschampsia caespitosa	U	6	6
Eleocharis palustris	D	1	0
Epilobium angustifolium	U	20	2
Epilobium ciliatum	D	0	24
Equisetum arvense	U	18	20
Equisetum fluviatile	D	1	0
Equisetum pratense	U	1	0
Equisetum sylvaticum	D	1	0
Fragaria vesca	U	19	0
Galeopsis tetrahit	U	1	0
Galium trifidum	D	0	7
Geum aleppicum	U	2	0
Geum rivale	D	2	0

Glyceria borealis	U	3	0
Hieracium umbellatum	U	4	0
Hippuris vulgaris	U	1	3
Hordeum jubatum	U	3	23
Juncus alpinoarticulatus	D	1	0
Juncus balticus	D	0	57
Lemna minor	U	1	0
Lotus corniculatus	U	1	0
Melilotus alba	U	2	0
Melilotus officinalis	U	2	0
Mentha arvensis	U	2	0
Menyanthes trifoliata	D	0	2
Myrica gale	D	2	0
Parnassia palustris	D	6	1
Petasites palmatus	U	2	0
Petasites sagittatus	U	4	0
Poa palustris	U	13	9
Poa pratensis	U	4	0
Polygonum amphibium	U	1	0
Populus balsamifera	U	21	5
Populus tremuloides	U	13	0
Potentilla norvegica	U	0	7
Potentilla palustris	D	0	7
Puccinellia nuttalliana	D	0	6
Ranunculus sceleratus	D	0	1
Ribes lacustre	U	3	0
Rosa acicularis	U	1	0
Rubus idaeus	U	19	0
Rubus pubescens	U	15	0
Rumex occendentalis	U	0	1
Salix candida	D	0	2
Salix exigua	D	4	14
Salix pedicellaris	D	0	5
Salix planifolia	D	0	1
Salix species	D	29	39
Schoenoplectus acutus	D	11	1
Scirpus atrocinctus	D	1	0
Scirpus microcarpus	D	2	9
Scutellaria galericulata	D	3	0
Sium suave	D	0	4
Solidago canadensis	U	1	0
Sonchus arvensis	U	17	40
Stellaria longifolia	D	3	8
Stellaria longipes	D	0	5
Taraxacum officinale	U	10	0
Trientalis borealis	U	15	0
Triglochin maritima	D	4	31
Triglochin palustris	D	0	20
Typha latifolia	U	33	46

Utricularia minor	D	11	0
Vaccinium oxycoccos	D	0	4
Vicia americana	U	3	0
Aneura pinguis	D	9	13
Aulacomnium palustre	D	25	4
Barbula unguiculata	D	1	2
Brachythecium acutum	D	8	23
Bryum argenteum	D	2	1
Calliergon giganteum	D	0	5
Campylium stellatum	D	0	18
Ceratodon purpureus	D	22	19
Drepanocladus aduncus	D	3	17
Drepanocladus polycarpus	D	13	53
Funaria hygrometrica	D	5	14
Hamatocaulis vernicosus	D	0	8
Helodium blandowii	D	1	6
Hypnum pratense	D	3	6
Leptobryum pyriforme	D	26	47
Marchantia polymorpha	D	1	13
Plagiomnium ellipticum	D	2	9
Pohlia nutans	D	6	0
Pohlia wahlenbergii	D	0	14
Polytrichum juniperinum	D	4	0
Polytrichum strictum	D	1	0
Ptychostomum pseudotriquetrum	D	28	60
Tomentypnum nitens	D	5	22
Cephalozia connivens	D	3	0
Lophozia ventricosa	D	1	0
Riccardia multifida	D	1	0
Calypogeia phagnicola	D	1	0
Mylia anomala	D	1	0

Table A5.4 Rank abundance results from species abundance data collected from vegetation plots in 2017 at the Sandhill Fen. Results include species rank (Rank), abundance, proportional abundance (Proportion), confidence interval limits for the proportion of each species (plower and pupper), accumulated proportional abundance (Accumfreq), and logarithmic abundance (Logabun).

Species	Rank	Abundance	Proportion	PLower	PUpper	Accumfreq	Logabun
Carex aquatilis	1	2176	31.3	24.2	38.5	31.3	3.3
Calamagrostis canadensis	2	1877	27	21.9	32.1	58.4	3.3
Typha latifolia	3	729	10.5	6	15	68.9	2.9
Carex utriculata	4	352	5.1	2.7	7.4	74	2.5
Ptychostomum pseudotriquetrum	5	209	3	1.2	4.8	77	2.3
Populus balsamifera	6	203	2.9	1.3	4.6	79.9	2.3
Rubus idaeus	7	142	2	0.9	3.2	81.9	2.2
Salix species	8	138	2	0.7	3.3	83.9	2.1
Equisetum arvense	9	90	1.3	0.4	2.2	85.2	2
Carex atherodes	10	80	1.2	-0.5	2.8	86.4	1.9
Aulacomnium palustre	11	66	1	0.3	1.6	87.3	1.8
Drepanocladus polycarpus	12	65	0.9	0.1	1.7	88.3	1.8
Leptobryum pyriforme	13	61	0.9	0.4	1.3	89.1	1.8
Schoenoplectus acutus	14	61	0.9	-0.1	1.8	90	1.8
Ceratodon purpureus	15	50	0.7	0.1	1.3	90.7	1.7
Poa palustris	16	48	0.7	0.2	1.2	91.4	1.7
Populus tremuloides	17	47	0.7	0.2	1.1	92.1	1.7
Rubus pubescens	18	47	0.7	0.1	1.3	92.8	1.7
Epilobium angustifolium	19	45	0.6	0.3	1	93.4	1.7
Fragaria vesca	20	42	0.6	0.2	1	94	1.6
Sonchus arvensis	21	35	0.5	0.2	0.8	94.5	1.5
Bromus ciliatus	22	27	0.4	-0.2	1	94.9	1.4
Trientalis borealis	23	24	0.3	0.1	0.6	95.3	1.4
Myrica gale	24	16	0.2	-0.1	0.6	95.5	1.2
Scirpus microcarpus	25	16	0.2	-0.2	0.7	95.7	1.2
Aster conspicuus	26	15	0.2	0.1	0.4	96	1.2
Taraxacum officinale	27	14	0.2	0	0.4	96.2	1.1
Aneura pinguis	28	14	0.2	0	0.4	96.4	1.1
Circium arvense	29	13	0.2	0	0.4	96.5	1.1
Achillea millefolium	30	12	0.2	0.1	0.3	96.7	1.1
Aster puniceus	31	11	0.2	0	0.3	96.9	1
Petasites palmatus	32	11	0.2	-0.1	0.4	97	1
Utricularia minor	33	11	0.2	0.1	0.3	97.2	1
Parnassia palustris	34	11	0.2	0	0.3	97.3	1
Carex pseudocyperus	35	10	0.1	-0.1	0.4	97.5	1
Brachythecium acutum	36	9	0.1	0.1	0.2	97.6	1
Petasites sagittatus	37	8	0.1	0	0.3	97.7	0.9

Poa pratensis	38	8	0.1	0	0.3	97.9	0.9
Carex bebbii	39	7	0.1	0	0.2	98	0.8
Hypnum pratense	40	7	0.1	0	0.2	98.1	0.8
Polytrichum juniperinum	41	7	0.1	0	0.2	98.2	0.8
Cicuta maculata	42	6	0.1	-0.1	0.2	98.2	0.8
Deschampsia caespitosa	43	6	0.1	0	0.2	98.3	0.8
Pohlia nutans	44	6	0.1	0	0.2	98.4	0.8
Tomentypnum nitens	45	6	0.1	0	0.2	98.5	0.8
Solidago canadensis	46	5	0.1	-0.1	0.2	98.6	0.7
Funaria hygrometrica	47	5	0.1	0	0.1	98.6	0.7
Polytrichum strictum	48	5	0.1	-0.1	0.2	98.7	0.7
Equisetum sylvaticum	49	5	0.1	-0.1	0.2	98.8	0.7
Agrostis scabra	50	4	0.1	0	0.1	98.8	0.6
Hieracium umbellatum	51	4	0.1	0	0.1	98.9	0.6
Triglochin maritima	52	4	0.1	0	0.1	99	0.6
Salix exigua	53	4	0.1	0	0.1	99	0.6
Dasiphora fruticosa	54	3	0	0	0.1	99.1	0.5
Glyceria borealis	55	3	0	0	0.1	99.1	0.5
Hordeum jubatum	56	3	0	0	0.1	99.2	0.5
Ribes lacustre	57	3	0	0	0.1	99.2	0.5
Vicia americana	58	3	0	0	0.1	99.2	0.5
Drepanocladus aduncus	59	3	0	0	0.1	99.3	0.5
Cephalozia connivens	60	3	0	0	0.1	99.3	0.5
Scutellaria galericulata	61	3	0	0	0.1	99.4	0.5
Stellaria longifolia	62	3	0	0	0.1	99.4	0.5
Agropyron trachycaulum	63	2	0	0	0.1	99.4	0.3
Geum rivale	64	2	0	0	0.1	99.5	0.3
Melilotus alba	65	2	0	0	0.1	99.5	0.3
Melilotus officinalis	66	2	0	0	0.1	99.5	0.3
Mentha arvensis	67	2	0	0	0.1	99.6	0.3
Bryum argenteum	68	2	0	0	0.1	99.6	0.3
Plagiomnium ellipticum	69	2	0	0	0.1	99.6	0.3
Betula pumila	70	2	0	0	0.1	99.6	0.3
Geum aleppicum	71	2	0	0	0.1	99.7	0.3
Carex canescens	72	1	0	0	0	99.7	0
Cicuta bulbifera	73	1	0	0	0	99.7	0
Cornus stolonifera	74	1	0	0	0	99.7	0
Eleocharis palustris	75	1	0	0	0	99.7	0
Equisetum fluviatile	76	1	0	0	0	99.7	0
Equisetum pratense	77	1	0	0	0	99.8	0
Galeopsis tetrahit	78	1	0	0	0	99.8	0
Hippuris vulgaris	79	1	0	0	0	99.8	0
Juncus alpinoarticulatus	80	1	0	0	0	99.8	0

Lemna minor	81	1	0	0	0	99.8	0
Lotus corniculatus	82	1	0	0	0	99.8	0
Rosa acicularis	83	1	0	0	0	99.8	0
Scirpus atrocinctus	84	1	0	0	0	99.9	0
Barbula unguiculata	85	1	0	0	0	99.9	0
Helodium blandowii	86	1	0	0	0	99.9	0
Lophozia ventricosa	87	1	0	0	0	99.9	0
Riccardia multifida	88	1	0	0	0	99.9	0
Calypogeia phagnicola	89	1	0	0	0	99.9	0
Carex aurea	90	1	0	0	0	99.9	0
Polygonum amphibium	91	1	0	0	0	100	0
Marchantia polymorpha	92	1	0	0	0	100	0
Mylia anomala	93	1	0	0	0	100	0
Carex hystericina	94	1	0	0	0	100	0

Table A5.5 Rank abundance results from species abundance data collected from vegetation plots in 2017 at the Nikanotee Fen. Results include species rank (Rank), abundance, proportional abundance (Proportion), confidence interval limits for the proportion of each species (plower and pupper), accumulated proportional abundance (Accumfreq), and logarithmic abundance (Logabun).

Species	rank	abundance	proportion	plower	pupper	accumfreq	logabun
Carex aquatilis	1	4293	44.9	38.8	51	44.9	3.6
Juncus balticus	2	1389	14.5	10.9	18.1	59.5	3.1
Ptychostomum pseudotriquetrum	3	600	6.3	4.5	8	65.8	2.8
Typha latifolia	4	453	4.7	2.6	6.9	70.5	2.7
Sonchus arvensis	5	340	3.6	2.3	4.8	74.1	2.5
Triglochin maritima	6	300	3.1	1.8	4.5	77.2	2.5
Triglochin palustris	7	299	3.1	1.3	4.9	80.3	2.5
Leptobryum pyriforme	8	173	1.8	1.2	2.5	82.1	2.2
Drepanocladus polycarpus	9	169	1.8	1.2	2.4	83.9	2.2
Carex utriculata	10	155	1.6	0.8	2.5	85.5	2.2
Calamagrostis inexpansa	11	145	1.5	0.6	2.4	87.1	2.2
Tomentypnum nitens	12	138	1.4	0.4	2.5	88.5	2.1
Salix sp	13	129	1.4	0.9	1.8	89.8	2.1
Carex prairea	14	99	1	0.5	1.5	90.9	2
Carex diandra	15	93	1	0.4	1.5	91.9	2
Salix exigua	16	92	1	0.1	1.8	92.8	2
Brachythecium acutum	17	56	0.6	0.3	0.8	93.4	1.7
Carex atherodes	18	50	0.5	0	1	93.9	1.7
Ceratodon purpureus	19	44	0.5	0.2	0.7	94.4	1.6
Scirpus microcarpus	20	40	0.4	0.1	0.7	94.8	1.6
Drepanocladus aduncus	21	38	0.4	0.2	0.6	95.2	1.6
Hordeum jubatum	22	35	0.4	0.2	0.5	95.6	1.5
Equisetum arvense	23	32	0.3	0.1	0.5	95.9	1.5
Epilobium ciliatum	24	29	0.3	0.2	0.4	96.2	1.5
Campylium stellatum	25	26	0.3	0.2	0.4	96.5	1.4
Poa palustris	26	22	0.2	0	0.4	96.7	1.3
Funaria hygrometrica	27	21	0.2	0.1	0.3	96.9	1.3
Marchantia polymorpha	28	20	0.2	0.1	0.3	97.1	1.3
Crepis tectorum	29	18	0.2	0.1	0.3	97.3	1.3
Aneura pinguis	30	17	0.2	0.1	0.3	97.5	1.2
Pohlia wahlenbergii	31	16	0.2	0.1	0.2	97.7	1.2
Puccinellia nuttalliana	32	16	0.2	0	0.3	97.8	1.2
Potentilla norvegica	33	15	0.2	0	0.3	98	1.2
Galium trifidum	34	15	0.2	0	0.3	98.2	1.2
Calliergon giganteum	35	13	0.1	0	0.3	98.3	1.1
Stellaria longifolia	36	13	0.1	0	0.3	98.4	1.1
Deschampsia caespitosa	37	11	0.1	0	0.2	98.5	1

Potentilla palustris	38	11	0.1	0	0.2	98.7	1
Plagiomnium ellipticum	39	10	0.1	0	0.2	98.8	1
Carex interior	40	10	0.1	0	0.2	98.9	1
Agrostis scabra	41	9	0.1	0	0.2	99	1
Hamatocaulis vernicosus	42	9	0.1	0	0.2	99.1	1
Hippuris vulgaris	43	8	0.1	0	0.2	99.1	0.9
Cicuta maculata	44	7	0.1	0	0.2	99.2	0.8
Vaccinium oxycoccos	45	7	0.1	0	0.2	99.3	0.8
Hypnum pratense	46	7	0.1	0	0.1	99.4	0.8
Helodium blandowii	47	6	0.1	0	0.1	99.4	0.8
Populus balsamifera	48	5	0.1	0	0.1	99.5	0.7
Salix pedicellaris	49	5	0.1	0	0.1	99.5	0.7
Sium suave	50	5	0.1	0	0.1	99.6	0.7
Stellaria longipes	51	5	0.1	0	0.1	99.6	0.7
Epilobium angustifolium	52	5	0.1	0	0.1	99.7	0.7
Aulacomnium palustre	53	5	0.1	0	0.1	99.7	0.7
Salix candida	54	4	0	0	0.1	99.8	0.6
Betula pumila	55	4	0	0	0.1	99.8	0.6
Alopecurus arundinaceus	56	2	0	0	0.1	99.8	0.3
Menyanthes trifoliata	57	2	0	0	0.1	99.9	0.3
Salix planifolia	58	2	0	0	0.1	99.9	0.3
Barbula unguiculata	59	2	0	0	0.1	99.9	0.3
Caltha palustris	60	2	0	0	0.1	99.9	0.3
Aster puniceus	61	1	0	0	0	99.9	0
Chenopodium alba	62	1	0	0	0	99.9	0
Ranunculus sceleratus	63	1	0	0	0	100	0
Bryum argenteum	64	1	0	0	0	100	0
Parnassia palustris	65	1	0	0	0	100	0
Schoenoplectus acutus	66	1	0	0	0	100	0
Rumex occendentalis	67	1	0	0	0	100	0

Table A5.6 Plots surveyed at the Nikanotee (NF) and Sandhill (SF) Fens in each species introduction method type, their classification into communities, and coordinates within the NMDS ordination (Figure 4.3). Communities defined using group cluster methods and a 40% cut-off of similarity. Species introduction approach (SIA) include SSF=Seeded Sandhill Fen; UNF=Unplanted Nikanotee Fen (NF); SNF=Seeded NF; CNF=Carex NF; JNF=Juncus NF; MNF=Moss NF; MCNF=Moss Carex NF; MJNF=Moss Juncus NF.

Plot	Fen	SIA	Community	NMDS1	NMDS2
10sljnw	NF	J	Juncbal	-0.23752	-0.81435
10slmjsw	NF	J	Careaqu	-0.33695	-0.53619
2sljsw	NF	J	Careaqu	-0.39868	-0.36969
2slmjse	NF	J	Careaqu	-0.42044	-0.28671
4sljse	NF	J	Juncbal	0.099535	-1.22184
4slmjse	NF	J	Juncbal	0.006627	-1.00267
5sljnw	NF	J	Juncbal	-0.06202	-0.88477
7slmjsw	NF	J	Careaqu	-0.46302	-0.18838
1mo	NF	M	Trigpal	0.102346	-0.95222
1mom	NF	M	Careaqu	-0.10978	-0.17559
2mo	NF	M	Careaqu	-0.29012	0.278402
2mom	NF	M	Careaqu	-0.27963	0.338333
3mo	NF	M	Careaqu	-0.33018	0.347842
3mom	NF	M	Careaqu	-0.30526	0.355178
5mo	NF	M	Careaqu	-0.23445	-0.16728
5mom	NF	M	Careaqu	-0.19461	-0.0235
7mo	NF	M	Careaqu	-0.2427	0.318015
10mosljnw	NF	MJ	Careaqu	-0.21475	-0.27008
10moslmjse	NF	MJ	Careaqu	-0.30118	0.125356
2mosljne	NF	MJ	Careaqu	-0.27396	-0.24239
2moslmjnw	NF	MJ	Juncbal	-0.32513	-0.60854
4mosljne	NF	MJ	Juncbal	-0.10278	-0.62445
4moslmjne	NF	MJ	Juncbal	-0.00562	-0.81052
5mosljse	NF	MJ	Juncbal	-0.18017	-0.8991
5moslmjnw	NF	MJ	Juncbal	-0.10775	-0.59666
7mosljse	NF	MJ	Careaqu	-0.35444	0.151107
7moslmjsw	NF	MJ	Careaqu	-0.33019	0.113539
10moslcse	NF	MU	Careaqu	-0.68698	0.178957
10moslmcsw	NF	MU	Careaqu	-0.23323	0.443918
2moslcsw	NF	MU	Careaqu	-0.25457	0.376678
2moslmcse	NF	MU	Careaqu	-0.16211	-0.01002
5moslmcse	NF	MU	Careaqu	-0.20803	0.283247
7moslmcnw	NF	MU	Careaqu	-0.314	0.473563
1sd	NF	S	Juncbal	-0.59904	-1.10355
1sdm	NF	S	Careaqu	-0.30854	-0.61368

2sd	NF	S	Careaqu	-0.55343	-0.10706
2sdm	NF	S	Careaqu	-0.33514	0.438094
3sd	NF	S	Careaqu	-0.3747	0.10088
3sdm	NF	S	Careaqu	-0.21904	-0.1582
4sd	NF	S	Careaqu	-0.32823	0.050679
4sdm	NF	S	Careaqu	-0.27591	0.167209
5sd	NF	S	Juncbal	-0.05103	-0.81244
5sdm	NF	S	Careaqu	-0.29444	0.046676
10slcse	NF	U	Careaqu	-0.3481	0.477247
1c	NF	U	Trigpal	0.13447	-1.10994
1cm	NF	U	Trigpal	0.314142	-1.48229
2c	NF	U	Careaqu	-0.3237	0.265709
2cm	NF	U	Careaqu	-0.41579	0.239212
2slcse	NF	U	Careaqu	-0.29254	0.508136
2slmcnw	NF	U	Careaqu	-0.29014	0.43994
3c	NF	U	Careaqu	-0.21917	0.192532
3cm	NF	U	Careaqu	-0.29264	-0.05533
4c	NF	U	Careaqu	-0.27995	0.405236
4cm	NF	U	Careaqu	-0.22745	0.220651
4slcsw	NF	U	Careaqu	-0.20449	0.213949
4slmcsw	NF	U	Trigpal	0.195652	-0.82495
5c	NF	U	Careaqu	-0.44825	-0.17892
5cm	NF	U	Trigpal	-0.2798	-1.12275
5slcse	NF	U	Careaqu	-0.19636	0.159733
5slmcsw	NF	U	Careaqu	-0.23042	0.323836
7c	NF	U	Typhlat	-1.30448	-0.57577
7cm	NF	U	Careaqu	-0.43899	0.359175
7slcse	NF	U	Careaqu	-0.31356	0.330822
7slmcne	NF	U	Careaqu	-0.23864	0.393026
s1	SF	S	Careath	1.502566	0.749517
s10	SF	S	Calacan	0.700258	-0.21119
s11	SF	S	Careaqu	0.16569	0.367217
s12	SF	S	Careaqu	0.1333	0.384581
s13	SF	S	Calacan	1.54646	0.194743
s14	SF	S	Calacan	1.409093	-0.4643
s15	SF	S	Poapal	0.963649	-0.78445
s16	SF	S	Calacan	1.327112	-0.47959
s18	SF	S	Calacan	1.449852	-0.18902
s19	SF	S	Calacan	1.555418	-0.05317
s2	SF	S	Careaqu	-0.05721	0.361721
s20	SF	S	Careaqu	-0.70683	0.417369
s21	SF	S	Calacan	1.54536	0.063531
s22	SF	S	Calacan	1.545294	0.024125

s23	SF	S	Calacan	0.981004	0.81971
s24	SF	S	Calacan	1.565926	0.003437
s26	SF	S	Careaqu	0.364844	0.268264
s27	SF	S	Careaqu	-0.03008	0.457957
s28	SF	S	Careaqu	-0.79448	0.099678
s3	SF	S	Careaqu	-0.29505	0.568465
s30	SF	S	Calacan	1.133008	-0.11519
s31	SF	S	Careaqu	-0.65629	0.252477
s32	SF	S	Careaqu	-0.7606	0.661496
s33	SF	S	Careaqu	-0.93895	-0.01514
s34	SF	S	Careaqu	0.284625	-0.29399
s37	SF	S	Calacan	1.423836	-0.3111
s38	SF	S	Calacan	1.331067	0.444761
s39	SF	S	Careaqu	-0.5861	0.318643
s4	SF	S	Careaqu	-0.79941	0.084076
s40	SF	S	Careaqu	-0.65741	0.203875
s41	SF	S	Typhlat	-1.01675	-0.13082
s42	SF	S	Careaqu	-0.33148	0.620758
s44	SF	S	Calacan	1.48057	0.322658
s45	SF	S	Calacan	1.537829	0.118078
s46	SF	S	Typhlat	-1.31061	-0.40551
s48	SF	S	Careaqu	-0.60047	0.293072
s49	SF	S	Careaqu	-0.77724	0.104205
s5	SF	S	Careaqu	-0.09325	0.436168
s50	SF	S	Careaqu	0.367999	0.287434
s51	SF	S	Calacan	1.302668	0.091393
s53	SF	S	Typhlat	-1.8404	-0.55111
s54	SF	S	Typhlat	-1.84007	-0.55026
s55	SF	S	Careaqu	-0.48287	0.390935
s56	SF	S	Careaqu	-0.48583	0.398875
s57	SF	S	Careaqu	0.211445	0.333757
s59	SF	S	Calacan	1.378018	-0.03269
s6	SF	S	Careaqu	-0.60536	0.511302
s61	SF	S	Typhlat	-1.24528	-0.42343
s62	SF	S	Typhlat	-1.42168	-0.55306
s63	SF	S	Careaqu	-0.39136	0.494072
s64	SF	S	Calacan	1.199635	-0.36839
s65	SF	S	Popubal	1.271978	-0.73687
s66	SF	S	Calacan	0.74071	0.094046
s67	SF	S	Careaqu	-0.15043	0.511017
s69	SF	S	Typhlat	-1.52303	-0.63518
s7	SF	S	Calacan	1.572158	0.166903
s70	SF	S	Careaqu	-0.34021	0.615055

s71	SF	S	Calacan	1.481673	-0.20344
s72	SF	S	Calacan	1.396223	0.039603
s73	SF	S	Calacan	0.96722	-0.02252
s74	SF	S	Calacan	0.680034	0.286867
s75	SF	S	Careaqu	-0.47676	0.339848
s76	SF	S	Careaqu	-0.36968	0.549463
s77	SF	S	Careaqu	-0.54257	0.56423
s79	SF	S	Calacan	1.25206	0.031327
s8	SF	S	Careaqu	-0.17708	0.559231
s80	SF	S	Calacan	0.756369	-0.28889
s81	SF	S	Careaqu	0.201078	0.351363
s82	SF	S	Careaqu	0.277208	0.326433
s83	SF	S	Careaqu	-0.70133	0.177602
s84	SF	S	Typhlat	-1.02267	-0.22295
s85	SF	S	Careaqu	-0.04677	0.456309
s86	SF	S	Careaqu	0.208893	0.299015
s87	SF	S	Calacan	0.517151	0.248221
s88	SF	S	Careaqu	-0.18309	0.529
s9	SF	S	Careaqu	-0.09995	0.466619
s91	SF	S	Calacan	0.831884	0.010381
s92	SF	S	Careaqu	-0.28141	0.561571
s93	SF	S	Careath	0.870356	-0.57724

Table A5.7 Model results from the multivariate permutational analysis (PERMANOVA) of differences in species abundance between species introduction methods and depth to water level. Bolded values indicate significant differences between treatment means at α = 0.05.

Source Effect	**df**	**SS**	**MS**	**Pseudo-F**	***P(MC)***
Species introduction method (Trt)	7	48463	6923.3	6.6735	**0.001**
Depth to water level (DTW)	82	207610	2531.8	2.4405	**0.001**
Trt x DTW	23	20084	873.21	0.8417	0.749
Residuals	30	31123	1037.4		
Total	142	354840			

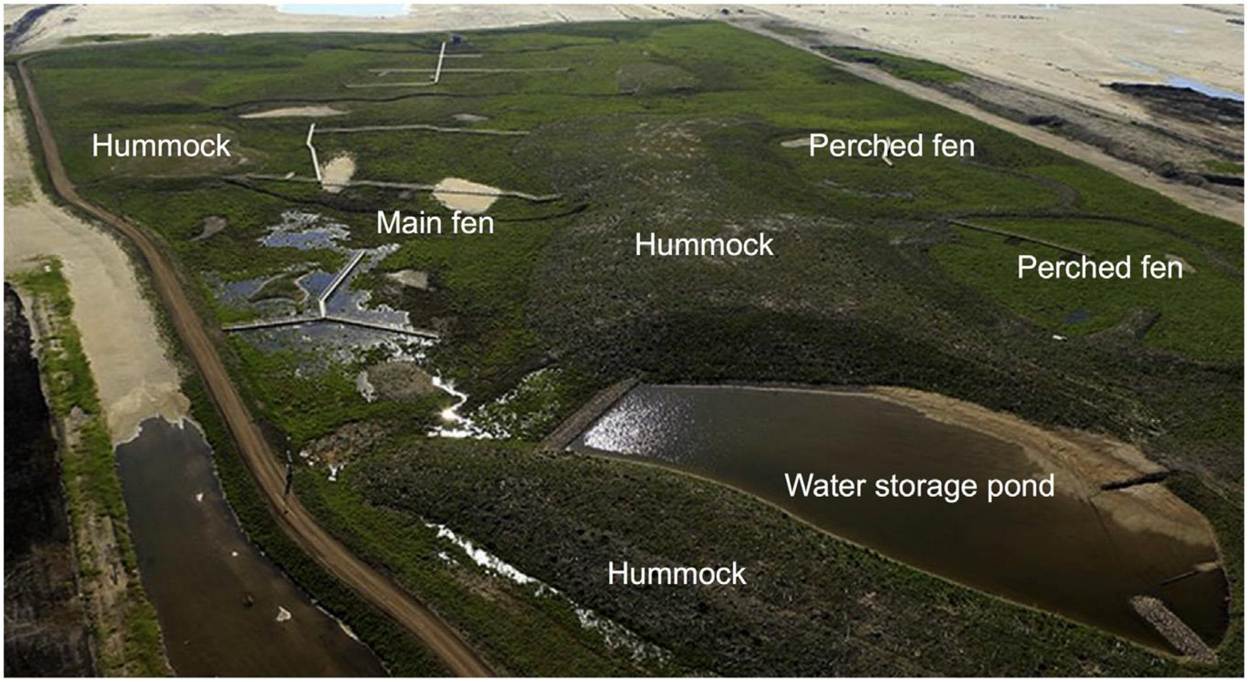


Figure A5.1 Aerial view of Sandhill Watershed in 2013. Data for this study was collected from the Main fen. Image provided by Syncrude Canada Ltd and Ketcheson et al. (2016).

Figure A5.2 Overview of Nikanotee Fen in 2014. Image provided by Andrea Borkenhagen.