The Role of Microtopography in Vegetation Colonization and Early Forest Development on Mine Reclamation Sites

by

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Abstract

Microsite heterogeneity is an important variable that drives biodiversity in forests. Current forest reclamation practices often do not incorporate site heterogeneity in their practices which might pose a challenge to the reclamation goals of restoring disturbed sites to resilient and sustainable forests. The goal of this thesis is to explore whether increasing variation of microtopographical features and substrates types have positive impacts on forest restoration efforts. Specifically, I am interested in the benefits of implementing microsite variability by increasing the microtopographic variation to the establishment and performance of tree seedlings (naturally regenerated and planted) and the colonization of the vegetation community.

In this operational scale study, I examined the effect of increased microsite heterogeneity on planted and naturally regenerated trees and the vegetation community by comparing a contoured (levelled (least heterogeneous and current practice)) treatment, a treatment that produced small parallel ridges (ridged), and a treatment (most heterogeneous) that used large loose piles of different material types pushed into alternating rows (hilled). I hypothesized that increased microsite heterogeneity would have a positive impact on tree establishment and growth and produce a more diverse plant community.

Overall, increased microtopographical along with coversoil heterogeneity improved tree establishment and growth as well as contributed to greater native species richness (33% increase in hilled and 43% increase in ridged treatment) compared to the more homogenous levelled treatment. Furthermore, the hilled treatment produced the greatest positive response in tree growth and native vegetation diversity. In the hilled treatment, the improved growth was especially observable for trembling aspen (43% height increase) and jack pine seedlings (29% height increase) when compared to the levelled treatment. Increased microtopography also encouraged natural regeneration of trembling aspen with a 60% and 31% increase in individuals in both the hilled and ridged treatments, respectively. Natural balsam poplar regeneration was 24 times higher in the hilled treatment than the levelled treatment. Interestingly, the hilled treatment had fewer non-native agronomic competitors and maintained more open mineral soil conditions, likely providing an advantage of reduced competition for planted and natural seedlings during establishment. The greatest positive responses of these microtopographical treatments appear to be gained when applied in areas that have stressful conditions (e.g. south facing slopes). The differences between the levelled and the hilled treatment of planted seedlings and natural regeneration were much greater when applied on a south-facing slope compared to an east-facing slope. The results of this study support the positive impact of heterogeneity on early forest establishment and growth and suggest that greater use of site reconstruction techniques that increase topographical and soil variability might be beneficial for forest reclamation sites.

Preface

This thesis is an original work by Sophie Aasberg (LeBlanc).

This study presented in this thesis is part of a multi-study project initiated by Dr. Simon Landhäusser of the University of Alberta, Rob Vassov of Canadian Natural Resources, Canada's Oilsands Innovation Alliance, and the Natural Sciences and Engineering Research Council. Dr. Landhäusser conceived the overall idea and developed the experiment. Several graduate students were part of this larger study (see below), who all actively participated in the refinement of the objectives of this large-scale operational project.

Chapter 2 - Surface Microtopographical Variation Improves Natural Tree Regeneration and Early Seedling Growth on Boreal Forest Reclamation Sites: Andrew Shaman collected the initial data in 2015, followed by Trevor de Zeeuw (M.Sc. 2019) for the 2016 and 2017 seasons, which Trevor analyzed and interpreted the short-term data for his thesis work: "The role of microtopography variation in land reclamation" (De Zeeuw 2019). To explore growth responses more explicitly and over a longer time horizon, I collected additional data, which I analyzed, compared, and interpreted in the context of the entire data set for this study.

Chapter 3 - Surface microtopographical variation enhances natural understory vegetation diversity on boreal forest reclamation sites: Kate Melnik (M.Sc. 2017) collected the initial vegetation data for this study to which I added a new data set in 2018. Parts of Kate Melnik's initial data including the seedbank data were published Melnik et al. (2017) The role of microtopography in the expression of the soil propagule banks on reclamation sites. Rest. Ecol.

(DOI: 10.1111/rec.12587). I compiled, analyzed, and interpreted the new expanded data set in the context of the 2015 results.

Dedication

I dedicate my research work to my husband Matt, who has been a constant source of support and encouragement during the challenges of graduate school. Matt, I will be forever thankful for having you as the strongest support in my life.

This work is also dedicated to my daughter Hazel:
You have made me stronger, better, and more fulfilled than I could have ever imagined.

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First and foremost, I would like to thank Dr. Simon Landhäusser for providing me with a life-changing opportunity to be a part of his research lab. During my graduate studies with Dr. Landhäusser, he provided me with guidance and support, challenged me to think critically, and pushed me out of my comfort zone, all of which was necessary for me to reach one of my biggest life achievements. I am thankful for all the valuable professional skills that he taught me along the way which will greatly benefit my career as a forester.

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Two important people that I would like to recognize are Trevor de Zeeuw & Kate Melnik, both of whom were heavily involved in this long-term study. These two individuals laid the groundwork and helped develop the experimental design of this study, and also collected a large portion of the data I used in my analysis. To the lab technicians, Fran Leishman and Caren Jones, you two were extremely helpful in making sure I was set up at the beginning of my program and prepared me for my data collection trip. I appreciate all of the hard work you both did to manage the lab and make sure all students were taken care of, at both academic and well-being levels. Caren, you have truly made a positive impact in my life; you are a great mentor and the help and mental support you provided me in the first couple of years of my program is something I’ll forever be thankful for. To the summer students (Brittany Hynes, Daniel Doyon, and Emi Reich) that helped me with my data collection in Fort McMurray, thanks for making our trip memorable and being a great team. To Coral Fermaniuk, we went through so much in these last almost five years together and we should both be extremely proud of ourselves for sticking through and being there for each other while managing family and other life pressures. To my co-workers from Silvacom, thanks for being understanding, giving me advice, and pushing me to complete this program. To my family, thank you for being there for me, especially in the final stages of my thesis. To my mother-in-law Ute, I will never be able to repay you for all of the babysitting hours and time spent with Hazel while I needed time away to finish writing.

Most importantly, to Matt, my biggest support; This has been a long ride for both of us and your support means the world to me. Navigating the masters, the pandemic, and becoming parents all at the same time has made us grow better together. I know you are as excited as me to wrap up this challenging chapter of our lives. I hope we can look back on this experience and have it serve as a reminder of how strong we were during this process, and how strong we are.

To Hazel, you have taught me most of all out of this experience. You have taught me how to find a balance between my career and life. You came along mid-way through my thesis and have reshaped my life focus and goals. In the last three years, you have been my true motivation and have given me the strength I needed to reach the finish line.

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1. Introduction

1.1 Boreal Forest and Industrial Disturbance

The Boreal Forest Natural Region covers over half (58%; 381,046km^2) of the province of Alberta (Natural Regions Committee 2006), and 44% of this region – or roughly one quarter of the entire province, is classified as the Central Mixedwood Natural Sub-Region. This region generally has short warm summers and long cold winters, with a mean annual temperature of +0.2 °C, a mean annual precipitation of 478mm (336mm for the growing season), and 97 frost-free days (Natural Regions Committee 2006). The key topographic features in this natural subregion include gently undulating plains with some upland hummocks that are imperfectly drained; these regions typically have medium nutrient regimes and mesic soil conditions, with elevations ranging from 250-1050m (average 525m). Upland sites are composed of Gray Luvisols (and other sub-gleyed groups) with a mosaic of trembling aspen (*Populus tremuloides* Michx.) and white spruce (*Picea glauca* (Moench) Voss) stands. Jack pine (*Pinus banksiana* Lamb.) can be found on xeric sites that have well-drained with coarser substrates (Natural Regions Committee 2006). In contrast, lowland forests are found in low-lying areas and are characterized by poorly drained thick organic soils (peat) composed of partially decomposed sphagnum moss or sedge (*Carex* spp.L.) vegetation. Lowland forest stands are usually composed of black spruce (*Picea mariana* (Miller) B. S. P.) and/or tamarack (*Larix laricina* (Du Roi) K. Koch). Other commonly found tree species in this natural subregion are balsam fir (*Abies balsamea* (L.) Mill), white birch *(Betula papyrifera* Marshall), and balsam poplar (*Populus balsamifera* L.). Understory species play a critical role in nutrient cycling and overstory succession in forests (Hart & Chen 2006), as well as account for the majority of biodiversity in Alberta forests (Roberts 2004; Gilliam 2007; Echiverri & Macdonald 2019) — with most boreal forest stands being composed of only about six tree species, but up to 77 understory species. In upland forests, common understory species are bearberry (*Arctostaphylos uva-ursi* (L.) Spreng.), low-bush cranberry (*Viburnum edule* (Michx.)), rose (*Rosa acicularis* Lindl.), Canada buffaloberry (*Shepherdia canadensis* (L.) Nutt.), bunchberry (*Cornus canadensis* L.), and red-osier dogwood (*Cornus stolonifera* L.), blueberry, (*Vaccinium myrtilloides* Michx.), and lichen species (*Cladonia* spp and *Cladina* spp). In wetland areas, common understory species in this region are bog cranberry (*Vaccinium vitis-idaea* L.), common labrador tea (*Ledum groenlandicum* Oeder), and feathermoss species.

Events such as fire, disease, and insect outbreaks are important drivers of boreal forest stand dynamics (Johnson 1992, Larsen 1997; Peltzer et al. 2000) as they promote the natural regeneration of understory and early successional tree species following aboveground disturbance (Larsen 1997). In general, the aboveground biomass removal facilitates the germination of buried or dispersed seeds and of vegetative propagules through reducing competition and increasing light exposure to the forest floor (Rydgren & Hestmark 1997). However, in some serotinous conifers, fire disturbances are necessary to facilitate the release of seed from cones (Lotan 1976). Natural regeneration may also occur through root suckering (as observed in trembling aspen and balsam poplar; Barnes 1966; Frey et al. 2003) or through stump sprouting (as observed in white birch; Safford et al. 1990). In addition to natural disturbances, the boreal forest in Alberta is exposed to numerous anthropogenic disturbances due to increasing land use for resource extraction such as forestry (aspen and conifer harvesting), as well as in non-renewable energy resources such as oil and gas (Natural Regions Committee 2006). Since Alberta has one of the largest bitumen sand deposits in the world (Pickell et al. 2015), oil and gas extraction has become a very prominent industrial activity in Alberta over the past 50 years. As part of the extraction bitumen mining features prominently on the landscape and unlike most natural disturbances which maintain some above ground vegetation components and substrates, the oil and gas mining involves the complete removal of vegetation components, as well as soils and the substrates overlaying the ore, thus posing a great challenge when reclaiming and revegetating these disturbed areas.

1.2 Forest Reclamation and its Challenges

In Alberta, it is required for resource industries to follow all reclamation guidelines under the Environmental Protection and Enhancement Act (Government of Alberta 2000), and specifically, to follow the Guidelines for Reclamation to Forest Vegetation in the Alberta Oil Sand Region (Government of Alberta 2009). The provided guidelines stipulate that reclaimed sites must be returned to their pre-forest state, both in terms of species composition and functionality, as a means of facilitating the development of dynamic equilibrium which in turn promotes resiliency to climatic events, topographic changes, and changes in properties of material types. However, due to the long recovery phase of forests following disturbance, coupled with oil and gas being a more recent industrial activity, the long-term effects on vegetation colonization and forest recovery are almost unknown in forest restoration practices (Dhar et al. 2018; Haeussler et al. 2021; Trepanier et al. 2021). In land reclamation, the high diversity of understory species is very important to take into consideration, however, in the past there have been challenges to restoring understories that are native and similar to natural forests (Harrington et al. 1999; Macdonald et al. 2015b).

During site reclamation, the landscape must be reconstructed before applying locally salvaged coversoil materials that are suitable for plant growth (Macdonald et al. 2015b). These landscapes are usually constructed by overlaying the extracted mined oil sand ore with overburden material that is typically a lean oil sand matter (a mixture of clay and sand with low bitumen content). Following that, coversoils are applied over the overburden dump. In Alberta, common salvaged coversoils used in forest reclamation efforts are salvaged Forest Floor Material (FFM), Peat Mineral Mix (PMM), or a mixture of the two. FFM coversoil is salvaged from upland forests and is a mixture of organic material forest floor and the upper underlying mineral soil horizons. In contrast, PMM is salvaged from lowland forests and is composed mostly of organic peat deposits and some of the underlying mineral soil (Mackenzie & Naeth 2010). FFM coversoil is typically preferred due to its native forest seedbank, but due to low FFM abundance in the region, PMM is often widely used as an additional coversoil (Mackenzie & Neath 2010; Naeth et al. 2013; Macdonald et al. 2015b). Challenges to using these coversoils in reclamation efforts include low-quality native plant propagules in coversoils (due to lengthy stockpiling periods) (Mackenzie et Naeth 2010; Archibald 2014; Macdonald et al. 2015a; Macdonald et al. 2015b) and the lack of natural seed sources adjacent to reclamation sites (Harrington et al. 1999; Gärtner et al. 2011). Lastly, both the application and spatial configuration of coversoils is a crucial aspect to consider when seeking to emulate the natural spatial heterogeneity of soil horizons in natural forests (Macdonald et al. 2012), however, our understanding of this aspect remains very poor.

Once the landscape is reconstructed with overburden materials and coversoils, the regeneration of the forest understory and the development of a tree canopy are important goals to promote boreal forest biodiversity, structure, and productivity (Roberts 2004; Hart & Chen 2006; Macdonald et al. 2012). However, the recovery of reclaimed sites is a lengthy process (taking decades) as various successional phases need to commence prior to achieving a closed tree canopy and understory as characterized by mature forests (Macdonald et al. 2012). In current reclamation practices, tree seedlings such as trembling aspen, jack pine, and white spruce are planted on reclamation sites to accelerate the reforestation process (Macdonald et al. 2015b). Successful seedling performance (establishment and growth) early on in restoration initiatives is crucial to establish a foundation of native trees and accelerate the process of crown closure, which in turn encourages native understory establishment and decreases the invasion of non-native species (Macdonald et al. 2015a). However, previous difficulties on reclaimed sites have included aggressive competitive agronomic species cover which impedes the establishment of native understory species and planted seedlings (Landhäusser & Lieffers 1998; Hart & Chen 2006; Skousen et al. 2006; Skousen et al. 2011; Evans et al. 2013; Macdonald et al. 2015a). Following natural disturbances (i.e., fire), evidence suggests that forests with increased species diversity are capable of recovering and reaching community stability more quickly in comparison to forests with lower diversity in the recovery phase (De Grand Pré & Bergeron 1997). However as mentioned previously, land reclamation differs from natural disturbance recovery due to the reconstruction of soils, and therefore, these reclamation sites provide research opportunities into how the initial stage of forest succession takes place on re-constructed sites (Roberts 2004; Trepanier et al. 2021).

1.3 The Use of Microtopography and Coversoils in Land Reclamation

Natural forests have topographical heterogeneity at the landscape, meso, and microsite scale, and this contributes to their unique vegetation biodiversity, both in the forest canopy and in the understory (Harper et al. 1965; Beckage & Clark 2003;). Microsites in natural forests provide increased habitat heterogeneity which supports the establishment of many different plant species with different environmental requirements (Harper et al.1965; Zedler & Zedler 1969; Bratton 1976; Beatty 1984; Peterson & Campbell 1993; Hart & Chen 2006; Lundholm 2009; Vodde et al. 2011; Echiverri & Macdonald 2019). As such, increasing microtopographical variation may be beneficial in forest reclamation strategies as this practice would theoretically provide suitable habitat for a wider range of native boreal species to establish (Gilland & McCarthy 2014; Melnik et al. 2017; Franklin & Buckley 2019).

Research on forest regeneration after harvesting has shown that enhancing microtopographic variation by using different site preparation techniques (i.e., mounding) can reduce spring flooding and soil warming earlier in the growing season than in depressed microsites (i.e., troughs or mound toes) (Biederman & Whisenant 2011; Melnik et al. 2017). This in turn increases establishment of deciduous tree seedlings (Lieffers et al.2017). Additionally, concave microsites that offer a greater range of moisture availability can be beneficial for mesic species to establish or provide relief during periods of moisture deficit (Lorio & Hodges 1971; Hart & Chen 2006; Biederman & Whisenant 2011; Gilland & McCarthy 2014). Increased microsite heterogeneity could also be achieved through the use of different coversoils that differ in edaphic properties and plant propagule banks, thus providing another potential means of creating a wider range of microsites and establishing a diverse source of native understory vegetation establishment (Mackenzie & Naeth 2010; Errington & Pinno 2015; Schott et al. 2014; Schott et al. 2016; Melnik et al. 2017; Dhar et al. 2018; Stack et al. 2020). More so, soils with higher organic matter content (such as PMM) can retain greater amounts of soil moisture compared to mineral-based soils (such as FFM), which may explain why increased natural colonization of tree species (such as trembling aspen) that are more sensitive to moisture stress during establishment has been observed in PMM coversoil (Pinno & Errington 2015).

Forestry and land reclamation professionals have attempted to replicate natural forest growing conditions on restoration sites, however, the efficacy of current reclamation practices in terms of achieving this is still unclear, especially so for large-scale reclamation projects and when considering temporal horizons which may span over many decades. Larger-scaled studies which have observed the recovery of forests following disturbance with different microsites with a wide range of microtopographical features are generally very limited. The goals of this thesis are to help explore this knowledge gap and how spatial heterogeneity might possibly benefit the development of planted tree establishment, natural tree regeneration, and the understory establishment on two Fort McMurray reclaimed sites 4 years after construction. This information could be very valuable and directly applicable to improving current forest reclamation practices in the area.

1.4 Research Objectives

The overarching research hypothesis of this thesis is that creating greater habitat heterogeneity (microsites) through manipulating the microtopography and substrate types on reclaimed sites will improve the survival and growth of tree seedlings as well as increase the biodiversity of a forest understory community over time. I further hypothesize that sites which exhibit increased variability in microsites will have greater natural tree regeneration and greater native colonizing species richness and diversity as compared to more homogeneous site conditions. Additionally, greater microsite variability and availability should not only lead to the early establishment of species more reflective of the original forest propagule bank, but it should also aid in the persistence of these species and continue to provide available microsites over the longer term.

1.5 Chapters Ahead

Chapter 2 presents the results from an operational-scale study of natural tree seedling regeneration and early planted seedling growth (trembling aspen, jack pine, and white spruce) in treatments with different microtopographic conditions. This chapter aims to identify if different microtopographic treatments and their associated suite of microsites yield differential patterns in seedling growth and the natural colonization of deciduous trees (trembling aspen and balsam poplar), as well determine if planted and naturally colonizing seedlings exhibit any microsite preferences during early establishment.

Chapter 3 examines the early establishment of a vegetation community and its diversity on sites with different microtopographic treatments over time following site construction. Results such as species richness, frequency, native species, species habitat preference, and overall vegetation community are evaluated.

Chapter 4 reviews and synthesizes the key findings of the two previous chapters and explores potential implications for reclamation management. Study limitations and future research opportunities are also provided.

2. Surface Microtopographical Variation Improves Natural Tree Regeneration and Early Seedling Growth on Boreal Forest Reclamation Sites

2.1 Introduction

Microsite heterogeneity at different spatial and temporal scales are important drivers that play critical roles in the functioning of ecosystems by providing a range of conditions and allows for the niche differentiation of species (Bratton 1976; Peterson & Campbell 1993; Lundholm 2009) which promotes biodiversity (Beckage & Clark 2003). Heterogeneity of microsites is driven by gradients of abiotic and biotic conditions influenced by factors such as microtopography (micro-relief and micro-aspect), soil substrates, pre-existing plant communities, seasonality and severity of disturbances, overarching climatic and edaphic processes, and their interactions (Zedler & Zedler 1969; Bratton 1976; Beatty 1984; Burke et al. 1999; Kumar et al. 2018). In natural ecosystems, variation in microtopography can be created by natural disturbances such as fire, windstorm events, or soil movements providing a range of substrates such as decaying wood, and organic- and mineral-dominated soil surfaces (Beatty & Stone 1986; Purdy et al. 2002; Vodde et al. 2011; Landhäusser et al. 2019). While plant-microsite relationships in natural ecosystems have been investigated in many studies and over many decades (i.e. Harper et al. 1965; Lundholm 2009), few have explored the role of microsite availability in the restoration of areas heavily disturbed by industrial activity. Of those few, most have focused on differences in seed germination requirements and vegetation establishment of wetlands, prairies, grasslands, or natural forest settings, following forest harvesting (Zedler & Zedler 1969; McGinnies et al. 1976; Loneragan & del Moral 1984; Fowler 1986; Økland et al. 2008; Hough-Snee et al. 2011; Gilland & McCarthy 2014; Lieffers et al. 2017). As such, there is little information on the impact of microsite variability and availability in upland forest restoration particularly at an operational scale.

The re-establishment of a tree canopy is a crucial first step in forest restoration following industrial disturbance (Macdonald et al. 2012). Tree establishment can occur via natural dispersal of seeds from nearby sources or from an available soil propagule bank (Jacobs et al. 2015); however, its success can be limited as it is dependent on the seed dispersal mechanisms of the species growing in the vicinity, the site characteristics and growing conditions of the area, and the availability of appropriate microsites with appropriate substrate and climate conditions (Primack & Miao 1992; Macdonald et al. 2012; Frouz et al. 2018, Kokkonen et al. 2018; Landhäusser et al. 2019). Furthermore, when relying on natural regeneration from a soil seed bank, longevity and coversoil placement thickness are also important (Rydgren & Hestmark 1997; Hopfensperger 2007; Landhäusser et al. 2015; Macdonald et al. 2015a). For example, some conifer species, such as white spruce (*Picea glauca* (Moench) Voss), are very dependent on well-decayed logs and moss beds for natural establishment from seed (Kokkonen et al. 2018); which is difficult to achieve when re-building forest restoration sites. Due to the aforementioned limitations associated with forest recovery using natural regeneration, the restoration of a forest canopy on reclamation sites often relies on the planting of tree seedlings (Macdonald et al. 2012).

For planted seedlings, planting check, which is associated with increased mortality and/or poor early growth, is a constraint associated with factors such as seedling and site quality and growing conditions that impact regeneration success (Lieffers & Beck 1994; Bedford & Sutton 2000; Löf et al. 2006; Evans et al. 2013; Bockstette et al. 2017; Landhäusser et al. 2019). Mechanical site preparation techniques that include soil tilling, disk trenching, and the formation of small mounds are thought improve some of these factors (Sutton 1993). In addition to using different site preparation techniques, careful consideration should be given to the selection of planting locations for the seedlings. Edaphic conditions such as water and nutrient supply and soil temperature will vary with different microsite positions, therefore selecting appropriate planting microsites that will provide suitable growing conditions for the seedlings is important on forest outplanting sites (Titus & del Moral 1998; Bruland & Richardson 2005). Unlike natural forest systems, forest reclamation sites are heavily disturbed and often lack microsite heterogeneity due to operational practices during site and soil reconstruction, such as contouring and levelling of soil surfaces with large machinery, which result in soil and topographical homogenization. However, forest reclamation sites can also provide unique research opportunities where one can explore the importance of microsite availability across different topographical positions and their effects on vegetation establishment in more detail thanks to relatively uniform initial edaphic conditions that have similar legacies such as seedbanks.

2.2 Research Objectives

The objectives of this research were: 1) to examine the impact and role of surface soil microtopography on the early growth of planted trembling aspen (*Populus tremuloides* Michx.), and conifers—white spruce (*Picea glauca* (Moench) Voss) and jack pine (*Pinus banksiana* Lamb.), and naturally colonizing trembling aspen and balsam poplar (*Populus balsamifera* L.) and 2) to identify the microsite preferences of planted and naturally colonizing trees on a large, operational-scale forest reclamation area.

2.3 Materials and Methods

2.3.1 Study Area

In the boreal forest of Alberta, areas disturbed by industrial resource extraction must be reclaimed to self-sustaining boreal forests once resource extraction is completed (Government of Alberta 2000, Government of Alberta 2009). Research was conducted at the Canadian Natural Resources Limited Albian Sands open pit oil sands mine located 70 km north of Fort McMurray, Alberta, Canada (57°15'N, 111°23'W). This mine is located within the Central Mixedwood subregion of the boreal forest (Natural Regions Committee 2006). Depending on the topography and parent material type, forest upland soils are classified as either Luvisolic or Brunisolic soils. In response to climatic conditions and forest cover type these soils can accumulate organic soil horizons (L-F-H horizons) of varying thickness (Natural Regions Committee 2006; Soil Classification Working Group 1998). Upland forest types found on sites with medium nutrient and mesic soil moisture conditions are commonly composed of varying mixtures of trembling aspen (*Populus tremuloides* Michx.) and white spruce (*Picea glauca* (Moench) Voss), while sites with poor nutrient and xeric conditions are dominated by jack pine (*Pinus banksiana* Lamb.). Lowland forests develop in low-lying areas with poorly drained mineral horizons, which commonly contain black spruce (*Picea mariana* (Miller) B. S. P.) and tamarack (*Larix laricina* (Du Roi) K. Koch) and are dominated by thick organic soils (peat) composed of partially decomposed sphagnum moss or sedge (*Carex* spp. L.) vegetation.

Climate in the region is cold with an average annual temperature of 1°C (±1.3 °C), with monthly January and July temperatures of -17.4 °C and 17.1 °C, respectively (Environment Canada 2018). Average annual precipitation is 418.6 mm in the region. The annual and growing season temperature and precipitation averages between 2015 and 2018 for this area are presented in Appendix 3 and are based on measurements from Environment Canada (2018).

2.3.2 Site Construction and Study Design

In October 2014, two large study sites (5 ha each) were established on a large overburden dump (hill landform, approximately 350 ha surface area) on the mining lease. The landscape feature was constructed from lean oil sands overburden material (a mixture of clay and sand with less than 4% bitumen content) that overlays the extractable mined oil sand ore. In this operational scale study, two sites (South and East) were established, each on a different slope aspect (south- and east-facing, respectively), but with similar slopes of 1:5 (20 %) (Appendix 1). For both sites the overall study design was a complete randomized block design. On each site, five blocks (each 100 × 100m) were established, and each block had three microtopographic treatments (33 × 100m) randomly assigned to it. The microtopographic treatments were a levelled, ridged, and hilled treatment (Appendix 2).

The overburden material that was used to create the landform is not considered suitable for plant growth and is required to be overlain with suitable coversoils in order to support the resource requirements (i.e. water and nutrients) for the initiation and establishment of plants and associated ecosystem function and processes (Macdonald et al. 2012). For the reclamation of the research sites, two types of cover soil materials were used (1) an upland forest floor material (FFM) and (2) a lowland peat mineral mix (PMM). Both materials had been salvaged from the lease prior to mining. Often these materials are stockpiled for long-term storage; however, for this study both materials were salvaged and only briefly stored (~6 months) prior to applying them as reclamation materials (see below). The FFM was a shallow salvaged (~30 cm depth) upland forest soil which is composed of a mixture of the organic soil horizons (L-F-H), as well as the underlying mineral A and B soil horizons. The original upland forests were dominated by jack pine and with a lesser component of trembling aspen and white spruce. The understory of these forests was mainly dominated by ericaceous dwarf shrubs (*Vaccinium* spp. L.) and lichens (*Cladonia* spp. P. Browne and *Cladina* spp.). The soil type of these forests are Eluviated Dystric Brunisols of glaciofluvial origin, which are coarse-textured and slightly acidic. The organic soil horizons of these soils are generally shallow (~5 cm). The PMM material was deeply salvaged organic lowland soils and is composed of a mixture of peat and some of the underlying mineral soil material. Lowland forests were dominated by black spruce (*Picea mariana* (Miller) B. S. P.). The FFM and PMM were both salvaged in April 2014 and stockpiled until site construction was completed in late fall 2014. Site preparation for both sites started by leveling the overburden material using D6 Caterpillar® bulldozers. The overburden material was then capped with a 35cm layer of PMM prior to the winter of 2014/2015. A second layer of FFM (15cm) was placed as a surface soil cap on the levelled and ridged treatments once the PMM layer was frozen to avoid mixing of the coversoils. The three microtopographical treatments differed in how the FFM surface soil cap was placed onto the frozen PMM layer. The levelled treatment, which is similar to common and current operational practices, was placed and levelled with the bulldozer blade (Appendix 2). The ridged treatment was constructed similarly to the levelled treatment, but instead of levelling the FFM surface soil cap, parallel ridges running perpendicular to the slope were created using the tracks of a D8 Caterpillar® bulldozer. This was done in the early spring (March 2015) when the surface FFM had thawed, while the underlying overburden and PMM layer were still frozen. This allowed the formation of the ridges with a reduced potential for compaction of the underlying PMM material. The ridges created were between 0.4 to 0.8 m tall, approximately 1.5 m wide, and were spaced 1 to 2 m apart (Appendix 2). For the hilled treatment, there was no uniform FFM surface soil cap applied, instead, FFM and PMM material was randomly selected from the stockpiles and pushed downhill using a D6 Caterpillar® bulldozer leaving large loosely piles (mound) of material in their final position. FFM and PMM mounds were placed randomly in off-set rows approximately 1.5 m apart and mounds were approximately 3.5 m wide by 5 m long and 1.5 m tall (Appendix 2). The average soil bulk density was 1.45 (± 0.02) g cm^{-3} in the levelled treatment, 1.34 (± 0.02) g cm^{-3} in the ridged treatment, and 1.03 (± 0.02) g cm^{-3} in the hilled treatment (Melnik et al. 2017).

Following the soil placement, one-year-old commercially grown nursery seedlings of three tree species (trembling aspen, white spruce, and jack pine grown from local seed sources and native to the area). The target planted density was approximately total of 3200 stems per hectare across each the site. A species mixture of 60% trembling aspen (1920 stems ha^{-1}), 20% jack pine (640 stems ha^{-1}), and 20% white spruce (640 stems ha^{-1}) was planted. Seedlings were free planted at an approximate equal spacing and planters were encouraged to avoid giving preference to any particular microsite. Average seedling heights at planting were 29 (±2cm), 22 (±1cm), and 27 (±1cm) for aspen, jack pine, and white spruce, respectively. Due to logistical issues, seedlings were planted in the spring of 2015 only on the South site, while the East site was planted in the late spring of 2016.

2.3.3 Data Collection and Calculations

To assess seedling responses to the microtopographic treatments, 180m^2 belt transects (90m long by 2m wide) were established in each microtopographic treatment on both research sites. Data on seedling height and root collar diameter (RCD) were collected on all trembling aspen, jack pine and white spruce seedlings found within these transects. Additionally, within these belt transects, there was a count of all naturally colonized balsam poplar seedlings (no growth response collected). Aspen natural seedling establishment that occurred on these sites was impossible to differentiate from the planted individuals after a few growing seasons (four on South and three East sites). As a result, both naturally regenerated and planted individuals were combined in the 2018 data collection. Based on repeated tree counts in the transects, a performance trajectory of a treatment was assessed by selecting the tallest 17 aspen saplings in each transect. The selected number of the best performing saplings represents a stand density of 944 stems ha^{-1} and falls into the range of stem densities that can be found in natural, closed canopy, mature aspen stands (Cumming et al. 2000). Aspen stem volume (V) was calculated for the top performing individuals using the volume of a cone:

$$V = \pi \times \left(\frac{RCD}{2}\right)^2 \times (height/3)$$

Since the South and East site were planted one year apart, growth measurements were conducted at the end of each growing season between 2015 and 2018 for the South site, while on the East site, measurements were conducted at the end of each growing season between 2016 and 2018. The microsite positions and substrate type of each seedling was also noted during the data collection in the ridged and hilled treatments. Microsite positions were defined as a *ridge* and *trough* position in the ridged treatment and as a *mound* and a *mound toe* position in the hilled treatment. The *mound toe* positions included the positions directly at the toe of a mound and the areas in-between mounds. The *mound* position in the hilled treatment was further characterized by substrate type (e. g. peat mineral mix (PMM) or forest floor material (FFM)), while the *mound toe* position in the hilled treatment and the *ridge* and *trough* positions in the ridged treatment all had only PMM as a substrate type (Appendix 2). Transects established in the hilled treatment contained overall similar numbers of FFM and PMM mounds.

To assess vegetation competition at the treatment level, colonizing vascular plant cover was estimated in quadrats ($1m^2$) placed along each transect at the peak of the 2018 growing season. Projected total cover (%) by species was estimated and microsite position (hilled treatment only), and coversoil type were noted for each quadrat. Species cover measurements were assigned into three different groups: shrubs, forbs, or graminoids. In both the levelled and ridged treatments, there was a total of 12 measurement quadrats in each transect. The quadrats in the levelled treatment were randomly located along the belt transect, while in the ridged and hilled treatments, quadrat locations were assigned randomly within specific microsite positions along the belt transect to assure that all microsites (and substrates) within a treatment were represented in the appropriate proportions (see below). Due to the relatively narrow troughs in the ridged treatment, sample quadrats characterized both the *ridge* and *trough* microsites combined. To capture both substrate types (FFM and PMM) and microsite positions (*mound* and *mound toes*) in the hilled treatment, six mounds (3 FFM and 3 PMM) were randomly selected along each belt transect. To characterize the vegetation of the whole mound, 5 quadrats were established (north, east, south, and west aspects and on crest) of each mound and one quadrat each was located on each cardinal direction at the toe of the mound. In order to estimate the overall hilled treatment vegetation, means were calculated based on both the 5 quadrats collected on the mounds and the 4 quadrats on the mounds' toes. Since the mound toe positions were much more prevalent than mounds positions in the hilled treatment, the averages for the microsites were corrected using a microsite availability ratio (see below) to estimate final overall treatment vegetation cover.

Since the microsite positions in each the ridged and hilled treatments were not equally available, we had to correct for their abundances within a treatment to be able to directly compare tree establishment at the microsite scale in these treatments. To achieve that, we determined a microsite availability ratio by identifying different microsites positions at 25cm increments along a 90 m line transect located in the center of the belt transect. The data indicated that the *mound toe* microsite position occurred 2.4 times more than the actual *mound* (FFM or PMM substrate) microsite position in the hilled treatment. Therefore, seedling densities found in the *mound* position were multiplied by 2.4 in the hilled treatment and by 4.5 in the *trough* microsite position in the ridged treatment.

Due to logistical issues, initial data collected on the East site to determine average planting density of aspen in each microsite positions in the ridged treatment was inadequate. We therefore estimated the initial planting densities by microsite using the same proportion of seedlings that were planted a year earlier in the ridged treatment on the South site (i.e. 25% of the planted aspen was in the *ridge* positions while 75% of the seedlings were planted in the *trough* position).

2.3.4 Statistical Analyses

All data were analyzed using R Project statistical software (R Core Team 2018). Linear mixed effect models from the *nlme* R package (Pinheiro et al. 2018) were used with treatment as a fixed effect and blocks as a random effect. To evaluate the microtopographic treatments at an operational scale, one-way analyses of variance (ANOVA) were used to explore treatment differences in aspen (natural and planted) density in 2018, naturally colonizing balsam poplar density in 2018, growth of top performing aspen (height, RCD, and stem volume), planted conifers (height and RCD), total vegetation cover by group type in 2018 on each research sites. Further, the planted seedlings responses to the microtopographic treatments were analyzed separately for the two sites, due to the difference in planting time (see above) and potential site differences were only evaluated qualitatively.

Responses at the microsite scale in both the ridged and hilled treatments were analyzed with one-way ANOVAs only in planted and natural aspen seedling by comparing changes in aspen (natural and planted) density, and in vegetation cover for each microsite in the hilled treatment. Planted conifers and balsam poplar were not included in this microsite level analysis, because there were not enough seedlings within the transects for each microsite to execute a meaningful analysis. All datasets were tested to ensure that the assumption of homogeneity of variances (Levene's test in the R *car* package (Fox et al. 2019)) was not violated for ANOVA assumptions. As a result of the operational scale experimental design and the limited replication, we used an alpha of 0.1 to consider significant differences and used Fishers least significant difference (LSD test) for post hoc comparisons.

2.4 Results

The initial average planting density for aspen on the South site was 1981 stems ha^{-1}, slightly above the target aspen density of 1920 stems ha^{-1}. There were slight differences in average initial densities of planted aspen among treatments with 1822, 1844, and 2278 stems ha^{-1} for the hilled, ridged, and levelled treatments, respectively; however statistically these differences were not significant (treatment $p = 0.37$; Table 2-1:). Four growing seasons later (2018), the average aspen density in the hilled treatment was higher (3133 stems ha^{-1}) than in the levelled treatment (1966 stems ha^{-1}), while the ridged treatment (2578 stems ha^{-1}) density did not differ other treatments (treatment $p = 0.008$; Table 2-1:Figure 2-1a). However, the difference between the initial and 2018 aspen density was significant different among all microtopographic treatments, with aspen stem density about 72% higher in the hilled treatment, followed by a 40% increase of stems in the ridged treatment and a 14% reduction in the levelled treatment (treatment $p < 0.001$; Table 2-1)

The average height of aspen (planted and ingress combined) in 2018 was not different among the treatments with 145.6cm for the hilled, 153.6cm for the ridged, and 133.7cm for the levelled treatments (treatment $p = 0.19$;Table 3-1:). However, after four growing seasons, the range in aspen heights (natural and planted combined) was greatest in the hilled treatment (from 7cm to 375cm), followed by the ridged treatment (14cm to 336cm), and the levelled treatment (41cm to 281cm) (Figure 2-1a), where the hilled treatment and to a lesser extend the ridged treatment contained a larger cohort of smaller trees compared to the levelled treatments, most likely a result of continued ingress of natural aspen regeneration from seed. Naturally colonizing balsam poplar exhibited similar establishment and growth patterns than found for the aspen; however most likely due to lower number and greater variation among replicates, they were statistically not different among treatments ($p > 0.14$), with a density of 533 (±606) (SD), 300 (±155) and 22 (±30) seedlings ha^{-1} in the hilled, ridged, and levelled treatment, respectively.

Apart from the trees, the total cover of colonizing vegetation was significantly higher in 2018 in the levelled (47.2%) and ridged (46.9%) treatments compared to the hilled treatment with 27.8 % ($p = 0.03$;Table 2-2). When separated by groups, total graminoid cover was slightly higher in the ridged (8.2%) and levelled (7.1%) treatments compared to the hilled treatment (3.8%); however, this difference was only marginally significant ($p = 0.11$; Table 2-2). Total forb (30%) and shrub (4.7%) cover did not differ among the three treatments (both $p > 0.02$; Table 2-2). The dominant forbs on site were competitive non-native agronomic and ruderal species which included *Medicago sativa* L., *Melilotus alba* Desr.*,* and *Sonchus sp* L (refer to Chapter 3 for more details). The dominant shrub species were native *Salix sp* L., *Prunus pensylvanica* L.f, and *Rubus idaeus* L.

The East site, which had also four years of natural vegetation colonization, but only three growing seasons for the planted seedlings (see methods), had somewhat lower initial planting densities of aspen than the south site in the previous year with an average of 1385 seedlings ha^{-1} that did not differ among treatments ($p = 0.12$; Table 2-1). In 2018, total average aspen densities (planted and natural combined) were 2778, 2044, 1200 stems ha^{-1} for the hilled, ridged, and levelled treatments, respectively (treatment $p = 0.006$; Table 2-1). Similar to the South site, the hilled treatment on the East site had significantly higher aspen ingress (difference between initial and 2018 densities) of 1144 stems ha^{-1} compared to the ridged and levelled treatments ($p = 0.04$) which were not different from each other with an average ingress of 522 stems ha^{-1} and 201 stems ha^{-1}, respectively (Table 2-1). In 2018, the average height of aspen seedlings (planted and ingress combined) was on average 80.4 cm and was not different among treatments (treatment *p*

= 0.446; Table 2-1). The range of seedling heights on the East site was not as large as on the South site, but generally the hilled treatment had a wider distribution of seedling heights (14cm to 215cm) compared to the other treatments (b). Balsam poplar natural regeneration also followed a similar trend, with the highest densities found in the hilled (422 ± 231(SD) stems ha^{-1} and the ridged treatment (378 ± 243 (SD) stems ha^{-1} compared to the levelled treatment (44 ±72 (SD) stems ha^{-1} (treatment p = 0.03).

In 2018, the hilled treatment had the lowest total vegetation cover (31.7%), which was significantly different than both the levelled (52.8%) and ridged treatments (44.3%) (treatment p=0.01; Table 2-2). While graminoid (7.4%) and shrub (4.4%) cover did not differ among treatments (both treatment $p > 0.77$), forb cover was different for all three treatments with the lowest cover in the hilled treatment (21.8%) and the highest in the levelled treatment (40.5%) (treatment p = 0.005; Table 2-2).

Due to the significant ingress of natural seedlings, we compared only the top 17 performing aspen seedlings in the different treatments (see methods). All variables such as height, root collar diameter (RCD), and stem volume of these saplings were impacted by the microtopographic treatments on the South site (all treatment $p < 0.001$;Figure 2-2 a,c,e; Appendix 4), while no differences were found for the East site (all treatment $p < 0.78$; Figure 2-2 b,d,f; Appendix 4). On the South site, the top performing aspen were the tallest and had the greatest RCD in the hilled treatment (227.7cm, 29.2mm) followed by the ridged treatment (186.1cm, 19.9mm) and the levelled treatment (156.7cm**,** 15.7mm). When assessing the average stem volume, it was different among all three treatments with aspen in the hilled treatment having the highest average stem volume (567 cm^{-3}) compared to aspen in the ridged (221 cm^{-3}) and levelled (129 cm^{-3}) treatments.

Overall mortality for both planted conifer species was less than 1% on both sites and there was no natural ingress of spruce or pine on either research site. On the South site, both the height and RCD of the jack pine after four growing seasons responded in similar patterns to the microtopographic treatments than aspen. Pine grew tallest and had the greatest RCD in the hilled and ridged treatments compared to the levelled treatment (both treatment $p < 0.02$;Figure 2-3a; Appendix 5). After four growing seasons, white spruce seedlings planted on the South site had greater root collar diameters in the hilled treatment compared to the other two treatments ($p = 0.009$, data not shown), while seedling height was not affected by the microtopographic treatments (treatment $p= 0.22$; Figure 2-3c). On the East site after three growing seasons neither species showed differences in height or RCD in response to the microtopographic treatments (Figure 2-3b, d; Appendix 5).

When exploring the microsite preference of establishing aspen at a more detailed microsite level, the difference between the initial and 2018 density indicated aspen ingress on *mound* microsites with an average addition of 1145 stems ha^{-1} on FFM and 2693 stems ha^{-1} on PMM mounds, while there was a decline in aspen density from the initial density in the *mound toe* microsite (-289 stems ha^{-1}) (microsite $p = 0.014$;). Similarly to the South site, aspen seedling density in 2018 on the East site indicates aspen seedling ingress on *mound* microsites, but the PMM *mounds* with an addition of 1598 stems ha^{-1} was significantly greater compared to the FFM *mound* microsites (an addition of 908 stems ha^{-1}) and the *mound toe* microsite (addition of 100 stems ha^{-1}) (microsite $p < 0.03$; Figure 2-3). Based on the availability of the different microsites, naturally colonizing aspen had an overall higher preference for the PMM *mounds* than for the FFM *mounds* on both sites. On the South site, natural aspen regeneration was 2.3

times more likely on mounds with a PMM substrate over FFM, whereas on the East site this ratio was 1.8.

When comparing aspen height response at the microsite level, there were no significant differences between different microsite positions in 2018 on the South site (microsite p= 0.18, data not shown). However, both the FFM and PMM *mound* microsite positions had taller aspen (86.0 (± 12.4) (SD) cm and 84.1(± 11.9) (SD) cm respectively) than the *mound toe* (65.5(±6.3) (SD) cm) on the East site (microsite p= 0.02; data not shown). Total vegetation cover on the South site was greater in the FFM and PMM *mound* positions (52.7% and 47% respectively) compared to the *mound toe* position (40.6%) (microsite p = 0.066), whereas on the East site overall vegetation cover was not significantly different among the three microsite positions in the hilled treatment (average cover 51.7%) (microsite p = 0.68).

In the ridged treatment on the South site, aspen ingress occurred only in the *trough* microsite, which had an average ingress of 877 aspen stems ha^{-1}, while there was an overall decline of 650 stems ha^{-1} in the *ridge* position; however, this difference was only marginally significant (microsite p = 0.11; Table 2-4). On the East site, both microsite positions experienced small increases in seedling density since planting (113 stems ha^{-1}), but no statistical differences between microsites were detected (microsite p = 0.49; Table 2-4).

2.5 Discussion

This operational-scale experiment demonstrates that site preparation techniques that increase microtopographic and microsite variability in reclamation areas can be very beneficial to forest reclamation efforts. Compared to common operational practices that often consist of homogenizing reclamation surfaces and materials by contouring, levelling, and mixing, this study highlights the benefits of enhancing microtopographical variation and providing a variation of substrate materials for early forest regeneration, which in our study included ridges and large mounds of loosely piled coversoil materials that differed in edaphic characteristics. The main trend found in this study was that larger microtopographical features along with the use of different coversoil materials created greater gradients of microsite conditions which allowed for improved establishment and growth of planted seedlings, particularly trembling aspen and jack pine, at the site level. Another significant benefit of a greater variation in microtopography appeared to be the enhanced recruitment of long-distance wind-dispersed tree species, such as aspen and balsam poplar. When comparing the growth responses to the treatments among the planted species, trembling aspen appeared to be more responsive to the treatments, followed by jack pine, with white spruce showing little response. The differential response was particularly noticeable for seedlings on the more sun-exposed South site. Some of the differences in species responses are potentially related to the different growth strategies and tolerances among the species with the early successional fast-growing aspen and pine being competitive species that demand high resource availability, while the later successional slower-growing white spruce can be considered more of a stress tolerator (Grime 1974). However, even though the microsite treatments had little early effects on the growth of white spruce, many forest regeneration studies have shown delayed growth responses to site preparation treatments in white spruce potentially due to slow early seedling establishment (Haeussler et al. 1999; Boateng et al. 2006; Boateng et al. 2009; Lieffers et al. 2018). Apart from the differences in growth strategies we believe that the more pronounced effects of our treatments on the South site points also to the possibility that slope aspect can be an important factor affecting the efficacy of microtopographical treatments when reconstructing forest sites. Since the two research sites had similar initial edaphic conditions (Melnik et al. 2017), the sun exposure on the South site provided an overall greater solar energy input than on the East site, suggesting that seedlings on the South site were most likely exposed to conditions, that could have significant positive or negative effects on their growth and overall productivity depending on the treatment. However, we observed only small treatment differences in seedlings on the East site and the data do not indicate that significant treatment differences would appear in the near-term on this site. Interestingly despite the greater potential exposure, the hilled treatment on the South site generated more natural regeneration of aspen and poplar and better growth of aspen than the East site, potentially related to the affinity of aspen to warmer growing conditions (Perala 1990; Zasada & Phipps 1990; Landhäusser & Lieffers 1998). This suggests that the provision of greater variation in topographical features and surface soil material types is most likely more effective on sites that are exposed to more stressful growing conditions, such as sun-facing slopes or sites with limited rooting space. It is also important to note that for the 2015-2018 period, the growing conditions for the region were somewhat below normal for precipitation and above normal for average temperature (Appendix 3), therefore the sites, particularly the South site, were most likely experiencing water limiting conditions during portions of the growing season. Different site preparations, such as mounding, can lead to the warming of the soil earlier in the growing season and has promoted aspen establishment outside its original range (Haeussler et al. 1999; Fraser et al. 2003, Landhäusser et al. 2010). While the warmer conditions could provide increased risk to the establishing seedlings, both *Populus* species have quickly expanding lateral root systems that can access soil moisture from microsites within a topographically variable landscape that accummulate water (Snedden 2013).

In addition to the topographical features, substrate type played also a significant role in the establishment and growth of seedlings. When comparing the total aspen seedling density (natural and planted combined) for each mound substrate (PMM or FFM) in the hilled treatment in 2018, the natural regeneration and growth of trembling aspen was the greatest on the PMM soil on both research sites. It is well understood that coversoil materials based on salvaged FFM and PMM have very different edaphic characteristics (Walczak et al. 2002; Errington & Pinno 2015; Rees et al. 2020; Stack et al. 2020), however we believe that in the context of this study, the greater natural aspen colonization and growth of aspen on the PMM substrate in both the hilled and ridged treatment was most likely driven by its greater availability of soil moisture (Pinno et al. 2012; Pinno & Errington 2015; Tremblay et al. 2019; Stack et al. 2020). Previous research has shown that the natural regeneration of trembling aspen and balsam poplar from seed are highly dependent on microsite soil moisture availability and requires a substrate with a high moisture-holding capacity; such as soils with high organic matter content or mineral soils with a finer texture (Wolken et al. 2010; Schott et al. 2014; Pinno & Errington. 2015; Lieffers et al. 2017; Landhäusser et al. 2019). Further, the difference in the placement of the coversoil among the microtopographic treatments influenced the pattern of natural regeneration responses. In the ridged treatment (particularly on the South site), there was greater natural regeneration of aspen in the lower *trough* microsite position when compared to the elevated *ridge* position. This higher regeneration in the *trough* is likely the result of the exposed PMM in the trough, providing greater moisture availability than the *ridges* composed of FFM substrate. In the hilled treatment (on both research sites), the highest natural regeneration was observed in the PMM *mound* microsite position. The combination of higher water availability of the PMM substrate and the higher micro-elevation of the *mound* position provided optimal conditions for aspen growth, both of which contribute to favorable conditions for the natural establishment and growth of aspen (Fechner and Burr 1981, Landhäusser et al. 2010, Landhäusser et al 2019).

Another factor in the seedlings response to our treatments was likely the parallel recolonization of vegetation after site reconstruction. Interestingly, vegetation cover was the lowest in the hilled treatment, which could have reduced competition effects for the planted and recruited tree seedlings. Generally competing vegetation on restoration sites has shown to negatively affect planted seedlings growth, especially early on after site construction and can be a challenge in forest restoration (Macdonald et al. 2015b; Tremblay et al. 2019). However, in both sites, the hilled and ridged treatments generated more natural colonization of aspen and poplar after site construction than the levelled treatment. Natural tree establishment by wind-dispersed seed has been shown to be more effective on soil surfaces that have high surface roughness, improving seed capture and early establishment (Titus & del Moral 1998, Landhäusser et al 2019). Another contributing factor to higher natural colonization of aspen and poplar particularly in the hilled treatment could have been the overall lower vegetation cover in this treatment, which provided a greater availability of suitable seedbeds over a longer period compared to the other two treatments. This suggestion is supported by the size distribution of aspen seedlings we observed in the hilled treatments, where a significant number of smaller seedling size classes were present, indicating a continuous recruitment of aspen seedlings over the measurement period. Natural regeneration of conifer species was not found within the first four growing seasons on both research sites. The dominance of natural regeneration by aspen and balsam poplar over the conifers can most likely be attributed to the differences in their ability of long-distance seed dispersal. Seeds of both aspen and balsam poplar can travel very long distances via wind, while most conifers have seed dispersal distances that are mostly driven by the height of the seed dispersing trees (Stewart et al. 1998) and are often limited by periodic seed production and/or the presence of serotiny (Gärtner et al. 2011; Lieffers et al. 2018).

In our study, mechanical site preparation increased the variety and availability of microsites in and positively influenced growing conditions, such as increased soil moisture availability and soil temperature, decrease in herbaceous competition, and increased shelter from unfavourable abiotic disturbances (wind, high sun exposure etc.) (Melnik et al. 2017), all of which are driving tree growth and establishment (Titus & del Moral 1998; Pinno & Errington 2015; Landhäusser et al. 2019). Overall, the goal to increase microtopographical variation applying different techniques during site and soil reconstruction has received only limited attention in forest reclamation practices, mostly due to a history of applying agriculture driven approaches to forest reclamation. Microsite-based forest regeneration using mechanical site preparation has had a long history and is extensively used as part of forest operations (Sutton 1993; Löf et al. 2012). Site preparation and microsite availability have shown to benefit the establishment, growth, and productivity of planted and naturally regenerated trees (Sutton & Weldon 1993: Haeussler et al. 1999; Boateng et al. 2006; Landhäusser 2009; Kokkonen et al. 2018; Franklin & Buckley 2019). Microsite factors driving plant responses on forest reclamation sites are often not directly comparable with those found in forest operations or after most natural disturbances, due to the large differences in the severity of site and soil disturbance and the required need for landscape, soil, and vegetation reconstruction (Macdonald et al 2012;

Macdonald et al. 2015b; Jacobs et al. 2015). However, this study showed that increasing microtopography, particularly on sites exposed to stress conditions, can help improve the establishment and growth of recently planted tree seedlings, while simultaneously increasing the natural recolonization of native species on restoration sites. Besides the obvious ecological benefits of considering these techniques in reclamation practices, the application of a treatment such as the hilled treatment to a reclamation site required significantly less time for soil and site reconstruction and reduced the number of travel passes of equipment across the site, therefore not only reducing operational costs, but also negative impacts on coversoils, such as excessive compaction.

2.6 Figures and Tables

Table 2-1:Average (± SD) of initial aspen planting density, total aspen density and height in 2018 (planted and natural ingress combined), the change in density for each microtopographic treatment on the South and East site. Different letters indicate significant differences among means (α=0.1; n=5).

Research site	Microtopographic treatment	Planting density (stems ha^{-1})	Total density in 2018 (stems ha^{-1})	Height in 2018 (cm)
South	Hilled	1822 (±173) ^{a}	3133 (±769) ^{b}	146 (±26) ^{a}
	Ridged	1844 (±661) ^{a}	2578 (±607) ^{ab}	154 (±16) ^{a}
	Levelled	2278 (±487) ^{a}	1966 (±775) ^{a}	134 (± 17) ^{a}
East	Hilled	1633 (±234) ^{x}	2778 (±758) ^{z}	76 (±4) ^{x}
	Ridged	1522 (±337) ^{x}	2044 (±887) ^{y}	77 (± 14) ^{x}
	Levelled	999 (±620) ^{x}	1200 (±279) ^{x}	89 (±25) ^{x}

Table 2-2: Average (± SD) of total vegetation cover (%), graminoid cover (%), bryophyte cover (%), forb cover (%) and shrub cover (%) in 2018 for each microtopographic treatment on the South and East site. Different letters indicate significant differences among means (α=0.1; n=5).

Research site	Microtopographic treatment	Total vegetation (%)	Graminoids (%)	Forbs (%)	Shrubs (%)
South	Hilled	30(±5) ^{a}	5(±0) ^{a}	20(±5) ^{a}	5(±0) ^{a}
	Ridged	50(±10) ^{b}	10(±5) ^{a}	35(±5) ^{a}	5(±0) ^{a}
	Levelled	50(±20) ^{b}	10(±5) ^{a}	35(±20) ^{a}	5(±5) ^{a}
East	Hilled	30(±5)^{x}	10(±0) ^{x}	20(±0) ^{x}	5(±0) ^{x}
	Ridged	45(±10)^{y}	10(±0) ^{x}	30(±5) ^{y}	5(±5) ^{x}
	Levelled	55(±10)^{y}	10(±5) ^{x}	40(±10) ^{z}	5(±5) ^{x}

Table 2-3: Average (± SD) of initial aspen planting density, aspen density in 2018 (natural and planted combined) for two microsite positions and coversoil types in the hilled treatment on the South and East site. All density measurements were corrected for the availability of each of the microsite types in the treatment (see methods for details) (α=0.1; n=5).

Research site	Microsite	Coversoil	Planting density (stems ha^{-1})	Total density in 2018 (stems ha^{-1})
South	Mound	FFM	641 (±437) [a]	1786 (±1385) [a]
	Mound	PMM	641 (±331) [a]	3334 (±1087) [b]
	Mound toe	PMM	1289 (±273) [b]	1000(±498) [a]
East	Mound	FFM	266(±211) [x]	1174 (±173) [x]
	Mound	PMM	934 (±389) [y]	2532 (±1385) [y]
	Mound toe	PMM	1133 (±217) [y]	1233 (±308) [x]

Table 2-4: Average (± SD) of initial aspen planting density, aspen density in 2018 (natural and planted combined), and the change in aspen density for each microsite in the ridged treatment. All density measurements were corrected for the availability of each of the microsite types in the treatment (for more details see methods) (α=0.1; n=5).

* = estimated planted density (see methods).

Research site	Microsite	Planting density (stems ha^{-1})	Total density in 2018 (stems ha^{-1})
South	Ridge	1650 (±1069) [a]	1000 (±1045) [a]
	Trough	1478 (±585) [a]	2355 (±571) [b]
East	Ridge	1930 (*)	2000 (±935) [x]
	Trough	1444 (*)	1600 (±808) [x]

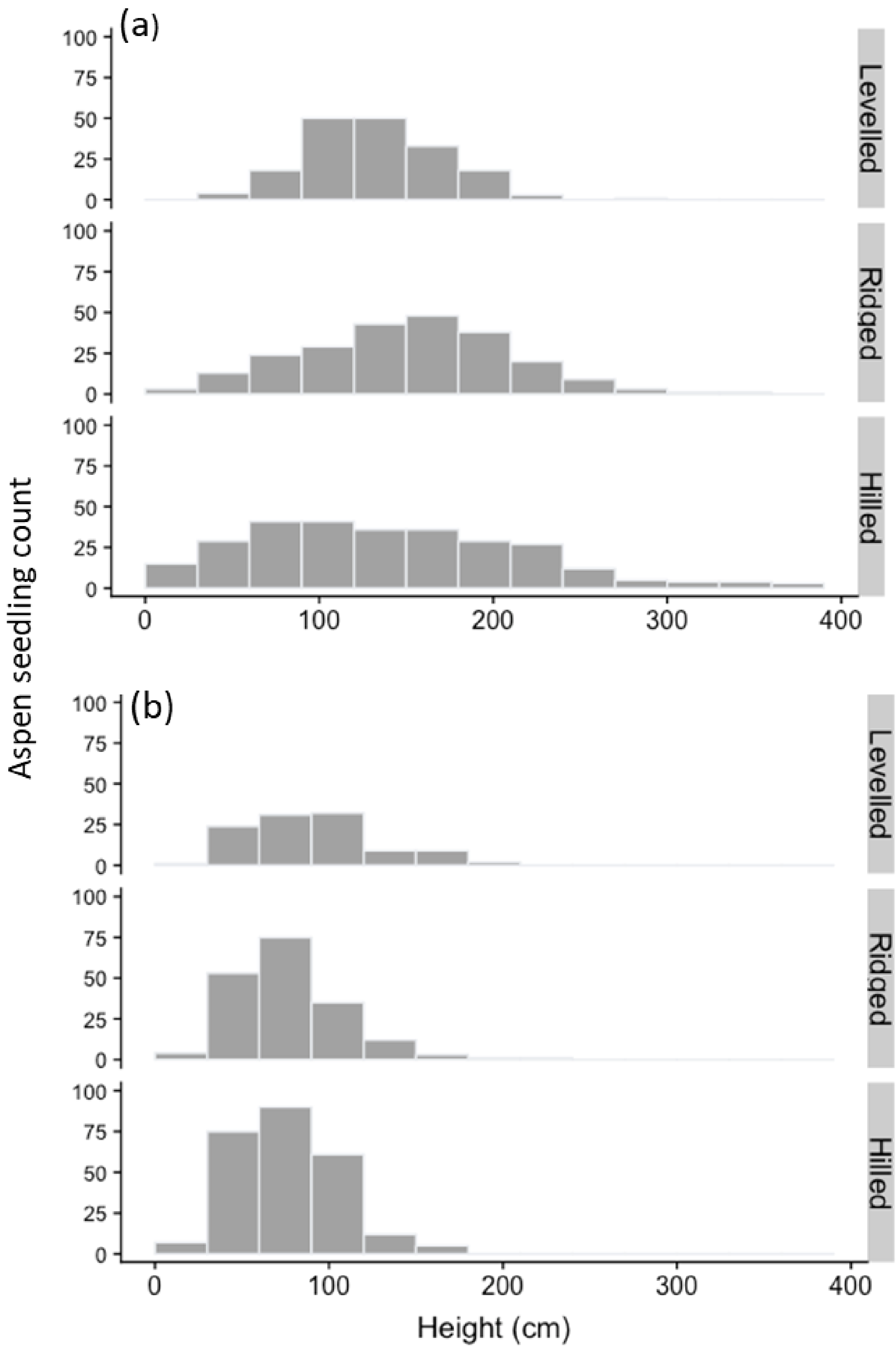

Figure 2-1: Height distribution of trembling aspen individuals on the a) South site and b) East site in 2018 in response to microtopographic treatments. A bin width of 30cm was used, which reflects approximately the average initial height of the aspen seedlings at the time of planting in 2015 (South site) and in 2016 (East site).

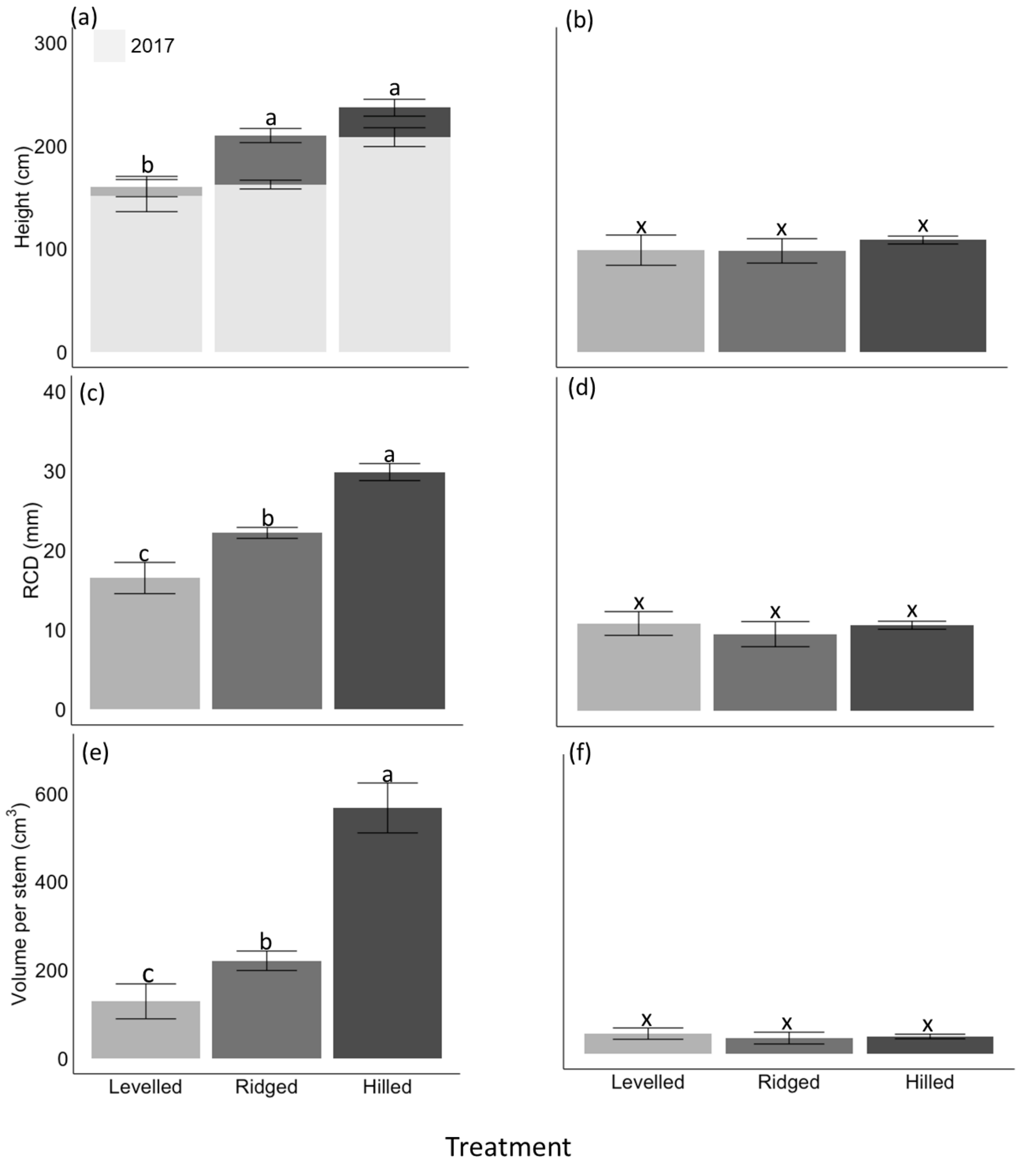

Figure 2-2: Average total height, root collar diameter (RCD), and average stem volume of the top performing trembling aspen stems per transect (see methods) measured in 2018 on South site (a, c, e) and East site (b, d, f) in response to three microtopographic treatments. Research sites were analyzed separately due to the different planting schedules. Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=5).

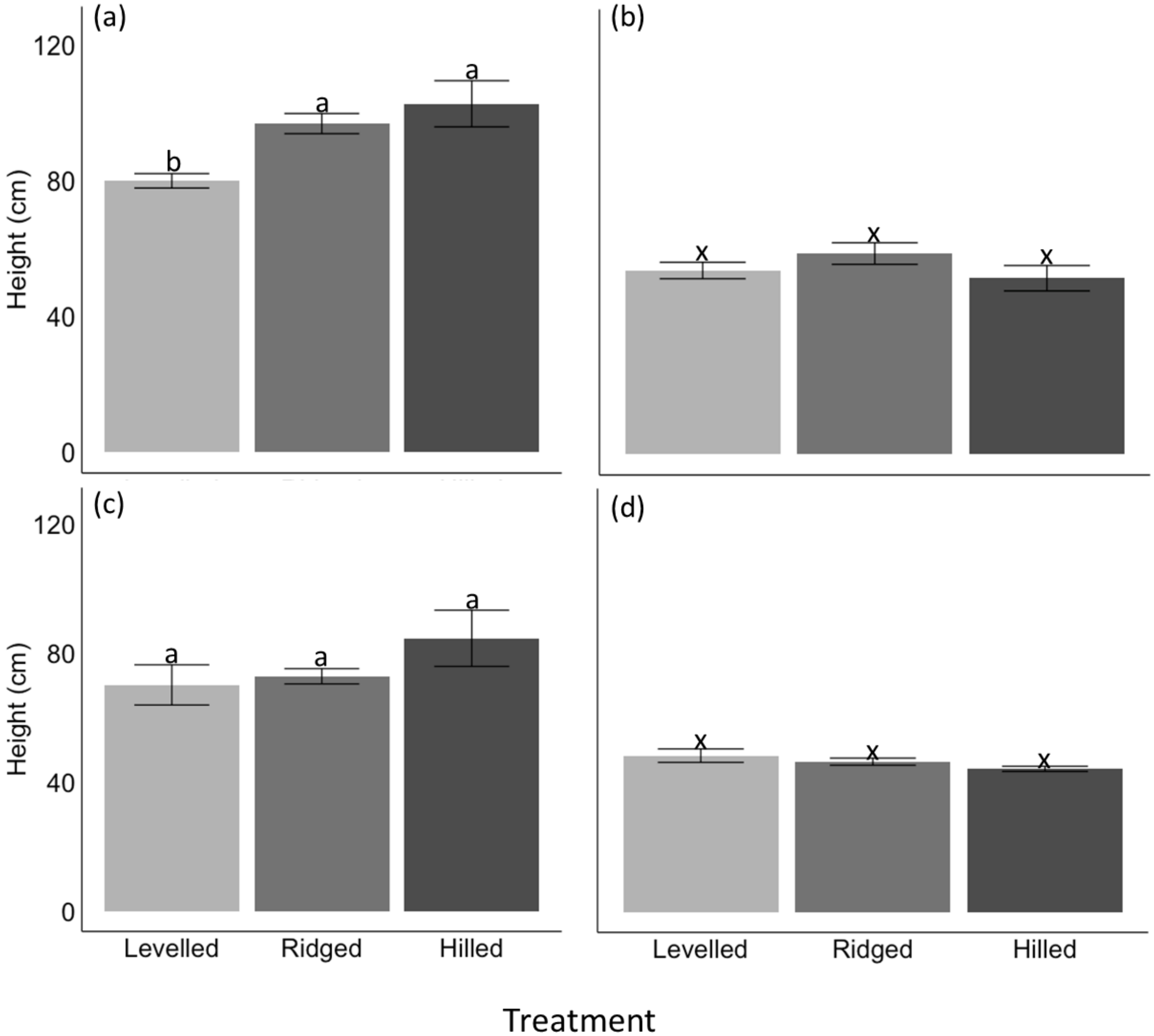

Figure 2-3: Average height of jack pine (top row) and white spruce bottom row in 2018 on the South (a, c) and East site (b, d). Research sites were analyzed separately due to different planting times. Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=5).

3. Surface Microtopographical Variation Enhances Diversity of Colonizing Vegetation on Boreal Forest Reclamation Sites

3.1 Introduction

Microtopographic heterogeneity often plays a significant role in creating a variety and range of microsite conditions which create important habitats for a wide range of species. In addition to microtopographical variation, other factors that influence microsite variability in natural forests are edaphic properties, climatic processes, and natural disturbance regimes (Zedler & Zedler 1969; Bratton 1976; Vodde et al. 2011; Kumar et al. 2018). Microsite variability on the landscape is frequently accompanied by different abiotic growing conditions such as gradients of water, light, and nutrient availability, ranges of edaphic conditions, physical shelter from extreme growing conditions (e.g., light, temperature, or wind), and biotic conditions such as species composition and interspecific and intraspecific species interactions. This natural variation in both microsites and growing conditions play an important role in influencing the biodiversity of ecosystems such as forests (Beatty 1984; Chipman & Johnson 2002; Macdonald et al. 2015b). More specifically, forest stands with a wider range of microtopographical variation generally correlate with vegetation communities that have greater species diversity (Harper et al. 1965; Huenneke & Sharitz 1986; Beatty & Stone 1996; Beckage & Clark 2003; Hart & Chen 2006).

Most of the plant species diversity found in natural forests originates in the strata below the tree canopy (i.e., the understory of shrubs, herbs, mosses, and lichens) (Gilliam 2007). This phenomenon applies particularly to the boreal forest where the species richness of the tree canopy is relatively low compared to the richness of the understory species (Gilliam 2007; Jones & Landhäusser 2017; Chen et al. 2018). While some of the most diverse boreal forests can hold up to 6 species in the overstory, the understory can hold more than 77 species (Hart & Chen

2006), therefore understories are an important component of these forests as understory communities provide a significant contributions to the function and processes of this ecosystem and play important roles in nutrient cycling and forest succession (Hart & Chen 2006; Hart & Chen 2008; Zhang et al. 2017; Errington & Pinno 2021).

Industrial activities such as resource extraction (i.e., surface mining) in forest regions are severe land disturbances that involve not only the removal of vegetation but also the complete removal of soils and the underlying substrates (overburden), and with that all topographical features at the micro and macroscale. In some jurisdictions, these disturbed areas are required to be reconstructed and restored to the point of achieving desired goals at the landscape and stand levels, such as a hydrological function of landscape and forest cover with similar ecological functions, processes, and resilience as there was before the industrial disturbance (Macdonald et al. 2012; Jacobs et al. 2015). However, there have been many challenges in attaining a diverse and resilient native understory on these industrially disturbed forest reclamation sites in the past (Dhar et al. 2018; Buss et al. 2018; Errington & Pinno 2021). Some of the challenges associated with poorly established vegetation communities are associated propagule availability and lack of appropriate microsites for early species establishment (Cornett et al.1997), unfavourable growing conditions (Macdonald et al. 2012), and a highly competitive environment due to the preferential establishment of ruderal species in the earlier stages of forest recovery (Buss et al. 2018).

The re-establishment of forest understories has generally been an overlooked and unexplored topic in forest reclamation with very few relevant studies (Melnik et al. 2017; Zhang et al. 2017; Chen et al. 2018). Many of the studies are focused on early general revegetation patterns rather than focusing on species-specific development of forest understories. Consequently, long-term effects on vegetation recolonization are little explored in forest restoration practices (Dhar et al.

2018; Haeussler et al. 2021; Trepanier et al. 2021) partly due to the slow recovery of forests. As a result of this knowledge gap, current land reclamation practices have often adopted hands-off regeneration approaches for understory recovery such as leaving sites for natural recolonization (Frouz et al. 2011) or providing plant propagules through forest floor transfers (Jones & Landhäusser 2018; Errington & Pinno 2021). However, these practices have been met with mixed success, which points to other factors than propagule availability potentially driving understory establishment success (Errington & Pinno 2021). There have only been a few observations that explored microtopographical variation and the use of different substrate types on species diversity and the overall recovery of reclamation sites and generally these studies focused on non-forest ecosystems (Bruland & Richardson 2005; Moser et al. 2007; Biederman & Whisenant 2011; Gilland & McCarthy 2014).

Current land reclamation practices create landforms that are often rebuilt very with homogenous and uniform substrate types (e.g. coversoils) and a placement that results in very little microtopographical variation. These homogenous conditions often do not represent microsite variation found in natural systems and might lack the growing conditions for a range of understory species (even if the propagules are available) while it allows for generalists such as ruderal species to rapidly occupy these sites. Consequently, the emergent vegetation is not reflective of the original propagule bank (Macdonald et al. 2015a; Melnik et al. 2017; Buss et al. 2018, Stack et al. 2020). Using different types of coversoils salvaged from different local forested ecosystems such as organic forest soils, or fine and coarse mineral forest soils provide not only a range in different edaphic conditions, but also contains different native propagules banks that might not only increase species richness (Nichols et al. 1998; Bartels & Chen 2010;

Errington & Pinno 2015; Melnik et al. 2017) but also potentially redundancy in species group types and with that resilience of a re-developing forest community.

3.1.1 Research Hypotheses and Objectives

In this research project, it is hypothesized that providing a greater range of microsites will improve and increase the biodiversity of forest understory vegetation communities. Furthermore, sites that have an increase in the availability and variability of microsites will have overall greater species richness and diversity mainly driven by a greater component of native plant species, compared to more homogeneous sites. Additionally, greater microsite variability and availability should not only lead to the early establishment of species reflective of the original forest propagule bank, but it should also continue to provide microsites over the longer term to maintain and potentially increase the number of species that are reflective of a forest understory.

3.2 Materials and Methods

3.2.1 Study Area

The research was conducted at the Canadian Natural Resources Limited Albian Sands open-pit oil sands mine lease located 70 km north of Fort McMurray, Alberta, Canada (57°15'N, 111°23'W). This mine is located within the Central Mixedwood subregion of the boreal forest (Natural Regions Committee 2006). For more details on the nutrient and moisture regimes, coversoil types, and dominant tree species found in this subregion, please refer to Chapter 2 of this thesis. Furthermore, climatic conditions such as annual and growing season precipitation and temperature for 2015 to 2018 can be found in Appendix 3.

3.2.2 Site Construction and Study Design

In October 2014, two large study sites (5 ha each) were established on a large overburden dump (hill landform) on the mining lease. The landscape feature was constructed from lean oil sands overburden material that overlays the extractable mined oil sand ore. In this operational scale study, two research sites (South and East) were established, each on a different slope aspect (south- and east-facing, respectively), with similar slopes of 1:5 (20 %) (Appendix 1). The study for both sites was set up as a complete randomized block design. On each research site, five blocks (each 100 × 100m) were established, and each block had three microtopographic treatments (33 × 100m) randomly assigned. The microtopographic treatments applied in each block were a levelled, ridged, and hilled treatment (Appendix 1 & Appendix 2).

For the reclamation of the research sites, two types of salvaged coversoil materials were used (1) upland forest floor material (FFM) and (2) lowland peat mineral mix (PMM). The FFM and PMM were salvaged in April 2014 and stockpiled until site construction which was completed that following fall (2014). The FFM was a shallow salvaged (~30 cm depth) upland forest soil that included the organic soil horizons (L-F-H) as well as the underlying mineral A and B horizons. Please refer to Chapter 2 for more details on the origin, growing conditions, and species commonly associated with these two types of coversoils used in this study.

Before the winter of 2014-2015, the landform built from overburden material was capped with a 35cm layer of PMM coversoil. In early spring 2015, FFM was placed as a second layer and as a surface soil cap (15cm) on the levelled and ridged treatments once the PMM layer was frozen to avoid mixing of the coversoils. The levelled treatment (operationally most common practice) was placed and carefully levelled producing little microtopographical variation, (Appendix 2). The ridged treatment was constructed similarly to the levelled treatment, but instead of levelling the FFM surface soil cap, parallel ridges running perpendicular to the slope were created. The ridges created were 0.4 to 0.8 m tall, approximately 1.5 m wide, and were spaced 1 to 2 m apart (Appendix 2). For the hilled treatment, there was no FFM surface soil cap applied directly on top of the PMM base layer. Instead, FFM and PMM material was randomly selected from the stockpiles and pushed downhill and leaving a large loose pile (mound) of material in their final position. FFM and PMM were mounds placed in offset rows approximately 1.5 m apart. Each mound measured approximately 3.5 m wide by 5 m long and 1.5 m tall (Appendix 2). Microsites defined in the ridged treatment were the *ridge* and the t*rough* and in the hilled treatment were the *mound* and *mound toe* (Appendix 2). The bulk density for each treatment was 1.45 ± 0.02, 1.34 ± 0.02, and 1.34 ± 0.02 g/cm^3 for levelled, ridged, and hilled treatments respectively (Melnik et al. 2017). For additional details on the construction of treatment, please refer to Chapter 2 of this thesis.

3.2.3 Measurements and Data Collection

To assess vegetation competition at the treatment level, the colonizing vascular vegetation cover (%) was estimated in quadrats ($1m^2$) randomly placed along transects in each treatment plot (length $180m^2$; see Chapter 2) at the peak of the 2015 and 2018 growing season. In 2015, there were 15 quadrats randomly sampled along each transect for all treatments, and the number of individuals per species was counted as many plant species were just developing, and the ground cover was not an appropriate measure. In 2018, the approach of sampling was adjusted to capture more accurately the differences in vegetation (see below) and the type of microsite and their availability in each treatment. In 2018, 12 quadrats were placed along each transect in the levelled and ridged treatment. The quadrats in the levelled treatment were randomly located across the belt transect, while in the ridged and hilled treatments, quadrats data were collected categorically (microsite specific), but microsites were still selected at random within the belt transect. This type of categorical collection in the ridged and hilled treatment was applied to capture all microsites within treatments and get a good representation of the overall vegetation cover of each treatment, but also to be able to compare vegetation cover at the microsite level. Furthermore, in the ridged treatment, the location of each quadrat sampled had to cover both the part of the *ridge* and the *trough* microsites. To capture both coversoil type and microsite positions (*mound* and *mound toes*) in the hilled treatment at a finer scale, six mounds (3 FFM and 3 PMM) were randomly selected in the hilled treatment and on each mound 5 quadrats were placed to capture the four cardinal slope aspects and the crest of the mound, and an additional 4 quadrats were placed at the four cardinal directions at the toe of each mound.

As mentioned previously, as species started to establish on the sites in 2015 and the individual plants were very small, therefore the vegetation composition and abundance were recorded as the number of individuals of each species rather than cover. In 2018, when species were well established the projected percent cover was estimated for each species in each quadrat, as well as the microsite position (hilled treatment only) and coversoil type (hilled treatment only) of each quadrat. All data were collected at the species level and species that could not be identified in the field were collected, pressed, and later identified in the lab at the genus level at a minimum. Based on the Flora of Alberta (Moss & Packer 1983), each species was assigned one of four natural habitat preference types: forest, mesic (moisture-loving), ruderal, or generalist. The generalist habitat preference was assigned to species that indicate to have a preference for more than more habitat types listed above (e.g. moist woods). Each species observed in this study was also classified as native or non-native to Alberta (Appendix 6) (Moss & Packer 1983; Brouillet et al. 2010).

The data from the initial propagule bank of the salvaged FFM and PMM coversoils' samples were collected in August 2014 and are presented in more detail in Melnik et al. (2017). Briefly, there were 18 bulk soil samples collected based on soil samples from three soil cores (3.3 L, 14.5 cm diameter × 20cm tall), with two cores collected from the top 20 cm of the stockpile surface and a third taken below 20 cm depth for each FFM and PMM stockpiles. To encourage maximum seed, vegetation, and bud bank germination from these collected soil samples, Melnik et al. (2017) followed Ter Heerdt et al. (1996) greenhouse plant incubation study protocols. The greenhouse study lasted 16 weeks long and was terminated once there was no germination of new seedlings. Melnik et al. (2017) provide greater in-depth results of the propagule bank collection and greenhouse experiment protocols and 2015 vegetation data collection.

3.2.4 Species Richness & Diversity Indices, and Dominant Species Calculations

For species richness, binary variables were assigned to each species in each quadrat sampled (in either 2015 or 2018) to indicate their presence or absence. Since species richness does not consider the number of individuals for each species present, Shannon-Wiener (Barnes et al. 1998) and Simpson (Simpson 1949) indices were used for quantifying the species richness and the evenness of these species across treatments and research sites. To calculate these indices, the following formulas were used using the vegan package in R (Oksanen et al. 2016) using the average species richness of each block. The equations below were used to calculate the Shannon-Wiener (H) and Simpson (D) indices (Simpson 1949; Magurran 1988), where p_i is the proportional abundance of each species and s is the number of species:

$$H = -\sum_{i=1}^{s} p_i \ln(p_i)$$

$$D = \sum_{i=1}^{s} {p_i}^2$$

The ten most dominant species were calculated based on the occurrence of each species by taking the sum of each species' presence (1) or absence (0) values of all quadrats collected in each transect and then averaged for each treatment (sites combined: n=10). Since the number of quadrats sampled was different among different treatments and measurement periods, a ratio of average species count per quadrat was calculated to compare averages (e.g., 6% frequency means that this species occurred in 6% of quadrats sampled).

3.2.5 Categories of Species Establishment and Persistence

Categories were assigned to each species to break down and highlight which species were established in specific treatments and which of them persisted over the measurement periods

(Table 3-3). Category 1 was assigned to species that were found in the initial propagule bank but never established on the research sites in either measurement period. Category 2 was assigned to species that were found in the initial propagule bank, as well as on the research sites in 2015, but only in *specific* microtopographic treatments (2a) or in both 2015 and 2018 (2b). Category 3 was assigned to species that were found in the initial propagule bank and present in *all* microtopographic treatments in 2015 (3a) or both in 2015 and 2018 (3b). Category 4 was assigned to species that were not present in the initial propagule bank but were present in a *specific* microtopographic treatment in 2015 (4a) or both in 2015 and 2018 (4b). Category 5 was assigned to species that were not in the propagule bank and most likely colonized from outside in *all* microtopographic treatments in 2015 (5a) or both 2015 and 2018 (5b).

3.2.6 Statistical Analyses

All data were analyzed using R Project statistical software (R Core Team 2018; version 3.5.0). Linear mixed-effect models from the *nlme* R package (Pinheiro et al. 2018) were used with block used as random effect. Three-way ANOVA were used to determine if there was a treatment, site, or time interaction (repeated measures) when comparing the average native species count, diversity indices (Shannon-Wiener and Simpson), and average species richness. All datasets were tested to ensure that the assumptions of normality (using the Shapiro-Wilk normality test from the R *stats* package (R Core Team, 2018)) and homogeneity of variances (Levene's test in the R *car* package (Fox & et al. 2019)) were not violated. Post hoc comparisons of the least mean squares (Tukey *p*-value adjustment) were performed to detect the main effects and interaction between microtopographic treatments, research sites, and measurement periods.

Three-way PERMANOVA (Permutational Multivariate Analysis of variance) tests were performed to test the interaction of microtopographic treatments, research sites, and measurement periods (repeated measures) on species richness using the R software package *vegan* (Oksanen et al. 2016). The PERMANOVA was calculated using the Bray-Curtis dissimilarity matrix method with 9999 permutations (Anderson 2001). To visualize the difference between vegetation community species richness and frequency, Non-metric Multidimensional Scaling (NMDS) ordinations were created (using the Meta MDS function in *vegan* package) which is an iterative method to converge the final ordination configuration with minimizing stress. Stress tests were used to indicate the goodness of fit of the two-dimensional ordination configuration was a good representation of the community in their multidimensional places and their corresponding similarities and dissimilarities between species. Stress tests give values representing the difference between distance in the reduced dimension compared to the complete multidimensional ordination. An alpha of 0.1 was used to consider differences among the interactions and the main effects significant. Post hoc comparisons were used using a permutational (N=999) permanova the pairwise test using "RVAideMemoire" R package (Hervé) to detect the main effects and interactions between microtopographic treatments, a p-value adjustment of "Holm" was used for these pairwise comparisons.

No statistical analyses were performed on all total count of species found across both research sites (Table 3-1:), including habitat preference types, species unique to treatment, and top 10 species (Table 3-4).

3.3 Results

3.3.1 Overall Vegetation

A combined total of 105 species were observed among the initial propagule bank and both research sites over the two measurement periods (Appendix 6). Out of these species, 33 of them were present in the initial propagule bank (both FFM and PMM coversoils results combined) (Table 3-1:). On both research sites, there was a total of 54 species in 2015, with an observed 33% increase (80 species.) in 2018 (Table 3-1:; Appendix 6). The hilled treatment in 2015 contained the greatest number of species (83% of total species found), followed by the levelled treatment (69%), and the ridged treatment (66%) (Table 3-1:; Appendix 6). In 2018, the species overall increased in the three treatments, but the response pattern remained similar with the hilled treatment containing 95% of the species found followed by the levelled treatment with 71%, and the ridged treatment with 66% (Table 3-1:). Interestingly, the hilled treatment vegetation cover was lower on both sites even though it had the highest amount of species count (Table 3-2). In 2018, the hilled treatment was significantly lower in vegetation cover on both sites (South site 28%; East Site 32%) than the levelled (South 47%; East 53%), and ridged treatment (South 47%, East 44%; p=0.03) (Table 3-2).

3.3.2 Species Unique to Treatment & Categories of Establishment

In 2018, 15.2% of all species that were found across all treatments on both research sites were uniquely found only in the hilled treatment, while there were only 2.8 % and less than 1% unique to the ridged and levelled treatments, respectively (Table 3-1:). Out of the 33 species that were detected in the original propagule bank, eight of them were never established (Category 1, Table 3-1:;Appendix 6), all eight species are native and common to Alberta's forests in this natural region (Table 3-1:; Appendix 6). These species were: paper birch (*Betula papyrifera*

Marshall), Canada bunchberry (*Cornus canadensis* L.), Labrador tea (*Ledum groenlandicum* Oeder), Wild lily of the valley (*Maianthemum canadense* Desf.), Bishop's Cap *(Mitella nuda* L.), Dewberry *(Rubus pubescens* Rafinesque), Slender wedge grass (*Sphenopholis intermedia (*Rydb.) Rydb.), and Blueberry (*Vaccinium myrtilloides* Michx.). Four species that were found in the propagule bank and only established in selected microtopographic treatments in 2015 but were absent in 2018 (Category 2a) were also all native species: fireweed (*Epilobium ciliatum* Rafinesque), small bedstraw (*Galium trifidum* L.), cursed crowfoot (*Ranunculus sceleratus* L.)*,* and common cattail (*Typha latifolia* L.) (Table 3-3; Appendix 6). Twelve other species fell into Categories 3a., 4a., and 5a. (Table 3-3; Appendix 6) which were described as species that were present in 2015 but not in 2018. Out of these 12 species, 75% of these had annual life strategies and the other 25% were perennials. Category 3a included yellow cress (*Rorippa palustris* (L.) Besser) that was found in the original propagule bank and had been established on all treatments in 2015. Category 4a were species that were colonized from an outside source (not in the propagule bank) and only found on a specific (one or two) microtopographic treatment in 2015: Red (or white) baneberry (*Actaea rubra* (Aiton) Wildenow), slough grass (*Beckmannia syzigachne (Steudel) Fernald*)*,* american dragonhead (*Dracocephalum parviflorum* Nutt.), touch-me-not balsam (*Impatiens noli-tangere* L.), Pineappleweed (*Matricaria matricarioides* DC)*,* false solomon's seal (*Smilacina* spp. Desf.), starflower (*Trientalis borealis* Rafinesque), and pussy willow (*Salix discolor* Muhlenberg). Category 5 were species: russian pigweed (*Axyris amaranthoides* L.), golden corydalis (*Corydalis aurea* Willdenow), prostrate knotweed (*Polygonum aviculare* L.) (Table 3-3; Appendix 6).

3.3.3 Dominant Species

The ten most dominant species observed across the sites and treatments in 2018 were two forest species: Slender wheatgrass (*Agropyron trachycaulum* (Link) Malte), and Red raspberry (*Rubus idaeus* L.), one generalist species: Hairgrass (*Agrostis scabra* Willd), and seven ruderal species: Narrow -leaved hawksbeard (*Crepis tectorum* L.), Foxtail barley (*Hordeum jubatum* L.), Alfalfa (*Medicago sativa* L.), White sweetclover (*Melilotus alba* Desr.), Yellow sweetclover (*Melilotus officinalis* (L.) Lam.), Wild buckwheat (*Polygonum convolvulus* L.), Sowthistle spp. (*Sonchus* spp. L.) (Table 3-4)—These ten species were present in both measurement periods and all had a lower frequency in 2015 compared to 2018. Ruderal species like *M.sativa, M.alba*, and *M.officinalis* had very little presence in 2015, but increased prominently and dominated the sites in 2018 (Table 3-4).

3.3.4 Species Habitat Preferences

When exploring the species compositions in relation to their associated habitat types, of the 33 species detected in the original propagule bank, 18% were associated with forests habitats, 24% with ruderal environments, 21% with mesic environments, while 36% were considered generalists (belonging to more than one these habitat types) (Table 3-1:). The main findings were that the hilled treatment had the highest proportion of forest species (15% in 2015; 21% in 2018), followed by the levelled treatment (11% in 2015; 18% in 2018), and the ridged treatment (11% in 2015; 17% in 2018) (Table 3-1:). The proportion of ruderal species decreased in all treatments from 2015 to 2018, with the hilled treatment having the lowest proportion (35.5% in 2015; 28.9% in 2018), followed by the ridged treatment (42.8% in 2015; 35.8% in 2018), and the levelled treatment (45.9% in 2015; 33.3% in 2018) (Table 3-1:). In both 2015 and 2018, the proportion of mesic species was the highest in the hilled treatment (18% in 2015; 22% in 2018), followed by the ridged treatment (17% in both 2015 and 2018), and then the levelled treatment (8% in both 2015 and 2018) (Table 3-1:). The richness of generalists was the greatest in the levelled treatment in both 2015 (35%) and 2018 (33%), followed by the hilled (both 30% across both measurement periods) (Table 3-1:).

3.3.5 Average Species Richness

One of the main findings for average species richness analysis was a treatment and site interaction (treatment × site p=0.042) (Figure 3-4:). All three microtopographic treatments did not differ between both research sites. The East site had a higher number of species since it had a stronger response for species richness than the South site. The hilled treatment generally had a greater average species richness among all treatments but only the East site (35.2 ± 3.8 (SD)) differed significantly from the levelled (East: 19.1 ± 3.4 (SD) South: 20.2 ± 2.1 (SD) and ridged treatments (East: 23.2 ± 2.7 (SD); South: 18.2 ± 2.6 (SD); p <0001) (Figure 3-4:). However, the South site hilled treatment (29.6 ± 4.9(SD)) was significantly greater than the South ridged and South levelled treatments (both p<0.0001) but was only marginally different from the East site ridged treatment (p=0.101) (Figure 3-4:). The ridged and levelled treatments were not significantly different from each other on either research site (East p=0.345; South p=0.917) (Figure 3-4:).

From 2015 to 2018, all microtopographic treatments significantly increased in average species richness (all p <0.001) with the hilled showing the highest increase among all treatments (Figure 3-3). In 2015, the hilled treatment had a higher species richness (20.2 ± 2.3 (SD)) than the ridged (13.5 ± 1.5(SD); p=0.001) and levelled treatment (12.5 ± 2.1(SD), p=0.0001) (Figure 3-3). The ridged and levelled treatments were not different from each other (p=0.850) (Figure 3-4:). In 2018, the hilled had the highest average species richness (44.7 ± 1.3 (SD)) and was significantly greater than the ridged (27.9 ± 1.1 (SD); $p<0.001$) and levelled treatments (26.8 ± 0.6 (SD); $p<0.0001$). The ridged and levelled treatments were not different from another ($p=0.874$). There was a treatment and measurement period interaction (treatment × year $p=0.009$) which is explained by the hilled treatment having responded with a stronger magnitude in 2018 compared to the other treatment in 2015 (Figure 3-3).

The difference in species richness is further supported by the nonmetric multidimensional scaling analysis, which shows a significant increase in species richness from 2015 to 2018 ($p=0.001$) (Figure 3-1), difference between research sites ($p=0.006$) with treatment interaction (treatment × period $p= 0.02$; stress=0.156; treatment × site $p<0.001$; stress=0.115) (Figure 3-2). There was no three-way interaction between the treatment, site, and measurement period (treatment × site × period $p=0.228$).

The treatments in 2018 generally had a smaller magnitude of species richness differences between blocks (less variation) than in the 2015 measurement period (ellipses in 2018 were much smaller and clustered together). In 2015, the levelled treatment differed from the hilled treatment ($p=0.015$) but not from the ridged ($p=0.486$) (Figure 3-1). The hilled and the ridged also had differences in vegetation communities ($p=0.021$) (Figure 3-1). In 2018, the hilled treatment had a different vegetation community than the levelled and ridged treatment (both $p=0.015$). The levelled and the ridged treatments had similar vegetation communities during each measurement period ($p=0.486$). Overall, the East site had less variation in the plant community than the South site (the ellipse shows more clustered averages in the East than the South site) (Figure 3-1).

3.3.6 Diversity Indices

The Shannon-Wiener and Simpson indices which are quantitative measures that reflect the number of different species and how evenly distributed the individuals among the species are. A three-way interaction was present for both the Shannon-Wiener and Simpson indices (H: treatment × site × year p=0.070; D: treatment × site × year p=0.069) (Figure 3-7 & Figure 3-8:). For both indices, the significant three-way interaction is driven by the hilled treatment having significantly higher indices than the levelled on the East site, but not on the South site in 2015 measurement period. There was a significant increase in both indices from 2015 to 2018 for all treatments except for the Simpson in the ridged treatment on the East site (p=0.293) (Figure 3-8:). For both diversity indices, significant differences were only detected among treatments on the East site in 2015. On the East site, the levelled treatment in 2015 had the lowest Shannon-Wiener index (1.97 ± 0.58(SD)) and Simpson index (0.82 ± 0.08 (SD)) among all treatments on both sites but only was significantly different than the hilled treatment on the East site (H: 3.27 ± 0.05(SD), p=0.0004) (Figure 3.7). All other treatments in 2015 on either site did not differ statistically in both the Shannon-Wiener and Simpson diversity index (Figure 3-7 & Figure 3-8:).

3.3.7 Native Species

Of the 33 species detected in the initial propagule bank, 76% (27 species) of the species were native to Alberta (Table 3-1:). Overall, there were more native species found on the research sites than the propagule bank. When examining the total count of native species found across all transects combined for each treatment, 78% of species found in the hilled treatment were composed of native species (in both 2015 and 2018), followed by the ridged treatments (72% of native species in both 2015 and 2018), and in the levelled treatment (65% in 2015 and 72% in 2018) (Table 3-1:). For native species richness, the microtopographic treatments followed the same pattern as the average species richness (both research sites) results stated above. From 2015 to 2018, all microtopographic treatments increased in native species richness and were significantly different from 2015 to 2018 measurement periods (treatment x period p=0.009) (Figure 3-5). The treatment and measurement period interaction are explained by the hilled treatment having a much greater difference compared to the levelled and ridged treatments in 2018. Comparably to species richness results, the native species richness in the hilled treatment (2015: 15.2 ± 2.0 (SD); 2018: 29.8 ± 1.2 (SD)) was significantly greater than the ridged treatment 2015: 8.9 ± 1.1 (SD); 2018: 17.3 ± 1.2 (SD), both p<0.001) and the levelled treatment (2015: 8.2 ± 1.4 (SD); 2018: 16.6 ± 0.6 (SD), both p<0.0001). The levelled and ridged did not differ from each other in 2015 (p=0.894) or 2018 (p=0.984) (Figure 3-5).

Overall, the East site responded stronger to the microtopographic treatment than the South site for native species richness (treatment × site p=0.015) (Figure 3-6). On the East site, the hilled treatment had the greatest native species count per transect (25.3 ± 2.3 (SD)) and was significantly different than levelled (12.1 ± 2.1 (SD); and ridged (15.5 ± 1.7 (SD); both p < 0.0001), whereas the ridged and levelled treatments did not differ on either research sites. On the South site, the native species count in the hilled treatment also performed higher (19.7 ± 3.2 (SD)) than the levelled (12.7 ± 1.3 (SD); p<0.001) and ridged treatment (10.7 ± 1.5 (SD); p<0.0001) (Figure 3-6).

3.4 Discussion

The results of this study highlight that an increase in microtopographic variation on forest restoration sites benefited the forest recovery of these sites by allowing a more diverse understory community to establish. Additionally, treatments with a higher microtopographic variability had vegetation communities that were composed of more native species and more species that were associated uniquely with that treatment. In more detail, the hilled treatment had the greatest species richness, including the highest number of native species, and represented most closely the initial propagule bank in both the 2015 and 2018 measurement periods. The ridged treatment also showed these positive effects however the magnitude of the treatment response was not as great as observed in the hilled treatment. This could be explained by the hilled treatment not only having an increase in surface roughness but also having greater exposure to the two different coversoils (FFM and PMM). The combination of these two factors likely provided a wider range of microsite types and was able to create conditions that favoured the establishment of a greater range of species than the levelled or ridged treatment. However, both the hilled and ridged treatment provided a wider range of microsite types than the levelled treatment. Both these treatments (i.e., hilled and ridged) had convex and concave microsites that offered likely different soil moisture and temperature conditions (Melnik et al. 2017) which would have contributed to different growing conditions.

Throughout the study, there was a significant increase in all diversity variables (average species richness, native species, Shannon-Wiener and Simpson indices) across all treatments between 2015 and 2018. Notably, the diversity indices increased from 2015 to 2018, however, significant differences were only observed in 2015, similar to the species richness results. Simpson diversity index accounts for both the abundance and evenness of a plant community and tends to be more responsive to differences in species dominating the community, whereas the

Shannon-Wiener diversity index is more responsive to differences in rare species in a plant community (Pitkänen 1998; Boateng et al. 2000; Nagenda 2002). This indicates that the different microtopographic treatments might have influenced the establishment of both less common species and dominant species cover during the first few years of establishment.

Another interesting result is that species that were only present in the 2015 measurement period (category 3a, 4a, 5a) were mostly composed of annual species, whereas the ten most dominant species in 2018 were mostly perennial species. The change from a plant community that is based on annual species to a community that is based on perennial species has also been observed in other reclamation studies (Landhäusser et al. 1996; Dhar et al. 2020), suggesting that short-term vegetation studies are of limited value when exploring the trajectory of vegetation recovery on reclamation sites. Over time, these sites should witness other shifts in the vegetation community patterns. The highest plant species diversity in a natural boreal forest stand can occur up to 40 years after fire disturbance (Hart & Chen 2006), therefore the diversity on these two sites most likely has not reached its peak in vegetation diversity and we should continue to see an increase over time. Other expected future vegetation community shifts include the closure of the canopy, which will also be a change in the forest structure with the development of an understory over time (Errington & Pinno 2021). Depending on stand density, canopy closure is usually achieved in 5-15 years for deciduous and pine stands and 40+ years for spruce stands (Lieffers et al. 2003; Errington & Pinno 2015). There are a lot of unknowns for the future vegetation community changes in forestry reclamation therefore the push in the industry and scientific community to research and understand how forest understory develops on reclamation sites is an understatement.

Even with the greater dominance of ruderal species in the plant community, greater species diversity was still observed in the hilled treatment (2018), which had also a higher proportion of forest and native species compared to the levelled and ridged treatments. However, an increase in species richness might not be indicative of a successful forest understory recovery. In the early establishment years, reclamation sites are almost always at risk of high vegetation competition due to the availability of growing space, therefore, giving opportunities for ruderal/non-native species to colonize rapidly and spread aggressively (Macdonald et al.2015b; Buss & Pinno 2018). When considering this, it is important to note that the majority (70%) of the most dominant species in this study were ruderal species and non-native to boreal forest ecosystems. This significant dominance of these ruderal species could have been caused by the limited establishment of native perennial but competitive species such as *Calamagrostis canadensis* (Michx.) Beauv. and *Epilobium angustifolium* L. due to the lack of vegetative and seed propagules. In other studies that had homogenous coversoil applications similar to the levelled treatment investigators observed that new vegetation germination was negatively affected by increased compaction and competing ground vegetation (Skousen et al. 2011; Macdonald et al. 2015b). In terms of compaction in this study, the levelled and ridged treatments had a greater bulk density than the hilled treatment which was likely caused by the greater number of bulldozer passes needed to contour the site compared to the hilled treatment, which needed only one pass pushing loose soil materials into mounds downhill. This reduction in machinery passes resulted also in reduced time and lower operational costs to build the site (Rob Vassov, personal communication, July 2018). Other studies have observed higher bulk densities in sites that had numerous machinery passes during construction (Zenner et al. 2007; Mundell et al. 2008). Additionally, soil compaction has been shown to have an impact on the vegetation community, such as higher bulk density has been shown to impact root penetration on early establishment vegetation (Gilland & McCarthy 2014; Melnik et al. 2017) and increase the establishment of ruderal species with strong taproots that are well adapted to higher soil bulk density conditions (Ewing 2002).

One highly competitive and non-native ruderal agronomic species that was commonly encountered in all microtopographic treatments across the research sites was *Medicago sativa* L., interestingly this species was marginally less abundant in the hilled treatment, potentially a result of the establishment of a greater number of native competitors in this treatment. Results from other studies also suggest that mechanical site preparation that promoted levelled and less rough surfaces or shallow windrows favoured the establishment of ruderal species, such as *M. sativa* (Biederman & Whisenant 2011; Macdonald et al. 2015a). However, these ruderal species usually do not play a large role in the vegetation community and often reduce in abundance after 5 years of restoration (Errington & Pinno 2015; Trepanier et al. 2021) as well will reduce in diversity over time with canopy closure (Macdonald et al. 2015b). The results of this microtopographic study indicate that treatments with higher microtopographical variability could potentially provide an advantage in reaching this forest successional stage at an earlier time.

Another factor that played a large role in achieving higher species diversity on sites was likely the simultaneous use of two coversoils (PMM and FFM) in the construction of the microtopographic treatments. However, the amount of PMM coversoil exposure differed by treatment. For example, both the ridged and the hilled treatment had greater exposure of PMM coversoil (i.e., the troughs of the ridges and in between all mounds in addition to PMM mounds), whereas the levelled treatment had none. Providing exposure to different substrates did not only provide a more diverse topography but it also provided two substrates that were edaphically different and contained two different propagule banks. Edaphically, PMM coversoil creates wetter and cooler growing conditions, has lower bulk density and higher organic carbon content (Errington & Pinno 2015; Melnik et al. 2017; Zhao & Si 2019), higher pH and nitrogen content (Stack et al. 2020) than coversoil, whereas FFM provides greater nutrients availability such phosphorus and potassium (Stack et al. 2020). With an increase in moisture availability in PMM coversoil, previous studies have claimed that lower microsites (i.e., mound toes or ridge troughs) generate an increase in early diverse vegetation establishment due to more favourable establishment growing conditions which could have benefited the hilled and ridged treatment in this case (Lorio & Hodges 1971; Hart & Chen 2006; Biederman & Whisenant 2011; Gilland & McCarthy 2014).

Like previously mentioned, due to different origins, the two coversoils provided two types of seedbanks on the research sites which helped increase the species richness on sites. PMM provided a seed and propagule bank that originated from lowland forest/peatland whereas the FFM provided a more upland forest species seedbank source. FFM is the preferred coversoil used in upland forest reclamation due to FFM having a greater amount of forest indicator species than PMM (Errington & Pinno 2015) and overall have greater species diversity (Mackenzie & Naeth 2010). However, due to its lower availability in the region, PMM is commonly used in tandem with FFM. Propagules found in the PMM are more likely adapted to lowland growing conditions which are usually associated with hydric conditions (Macdonald et al. 2015a), therefore vegetation establishment from this coversoil might be low to thrive in an upland forest restoration site. Having said that, the provision of moisture receiving (concave) microsites (i.e., mound toes with greater water availability and cooler temperatures) can create growing conditions where these mesic species are able to establish on uplands sites.

Another interesting finding in this study was the exploration of the early establishment and persistence over the first four growing seasons of native species. Species belonging to Category 1 could be considered as species of special interest, as they were never established on the research sites although they were present in the original propagule bank. These species (*Betula papyrifera*, *Cornus canadensis, Ledum groenlandicum*, *Maianthemum canadense, Mitella nuda, Sphenopholis intermedia, Rubus pubescens,* and *Vaccinium myrtilloides*) are all native boreal forest species (Archibald 2014; Macdonald et al. 2015b) and some species (e.g., *Cornus canadensis, M. canadense,* and *M.nuda*) are considered later successional species that are shade tolerant and/or understory obligate and therefore require a closed overstory for establishment (Beatty 1984; Peterson & Campbell 1993; Hart & Chen 2006). We can speculate that the lack of forest canopy at this restoration stage might have inhibited the establishment of these shade tolerant species. This also aligns with the observation of disturbance recovery processes of the boreal forest where some species taking longer to establish after disturbance, such as needing a closed canopy for to different light requirements (Hart & Chen 2006). The importance of canopy closure and vegetation understory development is something that is discussed extensively in the literature (Beatty 1984; Macdonald et al. 2015b; Messier et al. 1998). Future research should be focused on shade-intolerant/early successional species that never established on sites (e.g., *B. papyrifera*, *L. groenlandicum, V. myrtilloides*, and *R. pubescens*). Other species of interest are species belonging to Categories 2a and 3a which are species that were present in 2015 but had disappeared in 2018. These species were almost all native boreal forest and mesic species (e.g., *Galium trifidum* L. *Ranunculus sceleratus* L., *Rorippa palustris* (L.) Besser, *Salix discolor* Muhlengberg). Speculations for these species not being persistent on sites could be explained by the weather being warmer and drier throughout this study than the average weather in Fort

McMurray region (Environment Canada, 2018). Growing conditions might also not have been as favourable for these mesic species and could have been possibly outcompeted by the species that are better adapted to survive in drier and warmer conditions.

An increase in microtopography and the increase in diverse vegetation establishment were observed regardless of the overall slope aspect. However, on the East site initially higher species richness was observed than on the South site and this could have been driven by the less extreme growing conditions with an eastern exposure (less direct radiant heat and with that less evaporation) (Melnik et al. 2017). Overall species richness on the East site was higher than on the South Site, particularly for the hilled and ridged treatment, but differences in species and native species richness and diversity between sites disappeared by 2018. The South site gained more species, driven likely by the microtopographic treatments with more relief and the developing tree canopy (see previous chapter) provided sheltered microsites (i.e., from solar radiation and wind. Distance to outside seed source is also an important factor influencing recolonization of reclamation sites (Jones & Landhäusser 2018), interestingly, the closest potential seed source (measured from the edge of intact mature forest stands to the center of the research site) was 850m for the East site while it was only 200m for the South site.

3.5 Conclusion and Implications

Understanding the colonization and early successional pathways of forest understory on reclamation sites is a topic that needs to gain more traction in forest restoration research since the understory vegetation holds most of the diversity in these forests and is a critical driver of canopy closure and overstory succession. It is evident from this study that reclamation sites that offer a greater range of microsite types and availability benefit the recolonization of vegetation and its richness. More specifically, sites providing microtopographic features on reclamation landscapes promoted higher species richness and more native species than sites that had homogenous conditions. Other important factors to consider when revegetating reclamation sites are potential outside seed sources, seed dispersal distance, seedbed conditions, growing conditions during dispersal, and the availability of microsites which are not occupied by other species (Peterson & Peterson 1992; Lieffers et al.2003; Jones & Landhäusser 2018). Even though this study spanned only four growing seasons, the greater number of native forest species that persisted in the hilled treatment is encouraging and could provide a good starting point for developing more diverse understories that are accelerated by a faster canopy closure of the tree canopy. However longer-term data is necessary to confirm this trajectory. Species that were present in the seedbank but not present on sites (Category 1 species) also suggest that the establishment requirements for these species from a propagule bank need to be examined more closely to help capture more of these species in future reclamation sites.

3.6 Figures & Tables

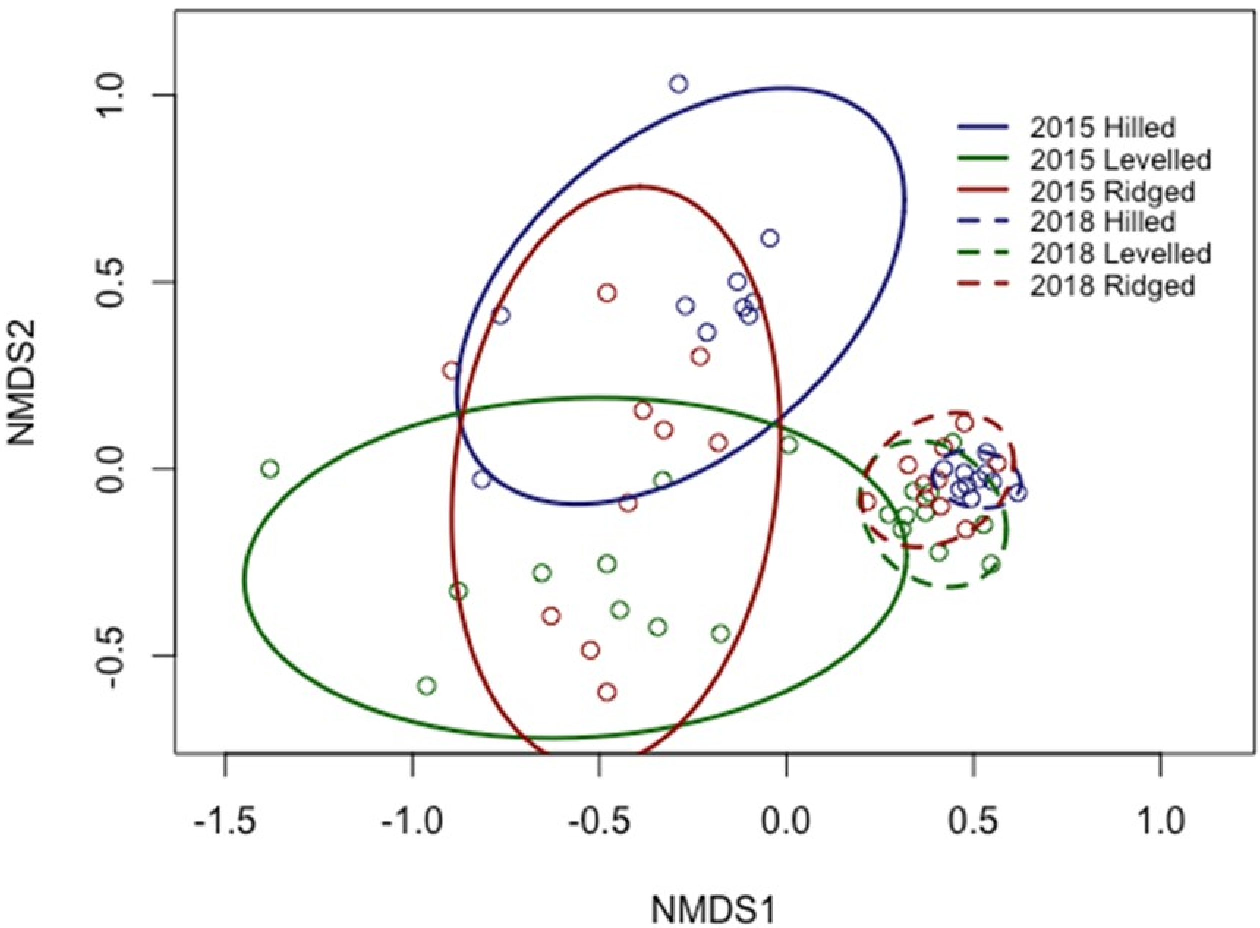

Figure 3-1 NMDS showing species richness for different microtopographic treatments in both measurement periods (research sites combined (α=0.1; n=10), treatment × site × period p=0.02; Stress: 0.156; Ellipses are showing CI= 95%). The NMDS1 & NMDS2 axis, show the range of the distances reached between the treatments and the measurement period with the Bray-Curtis dissimilarities matrix.

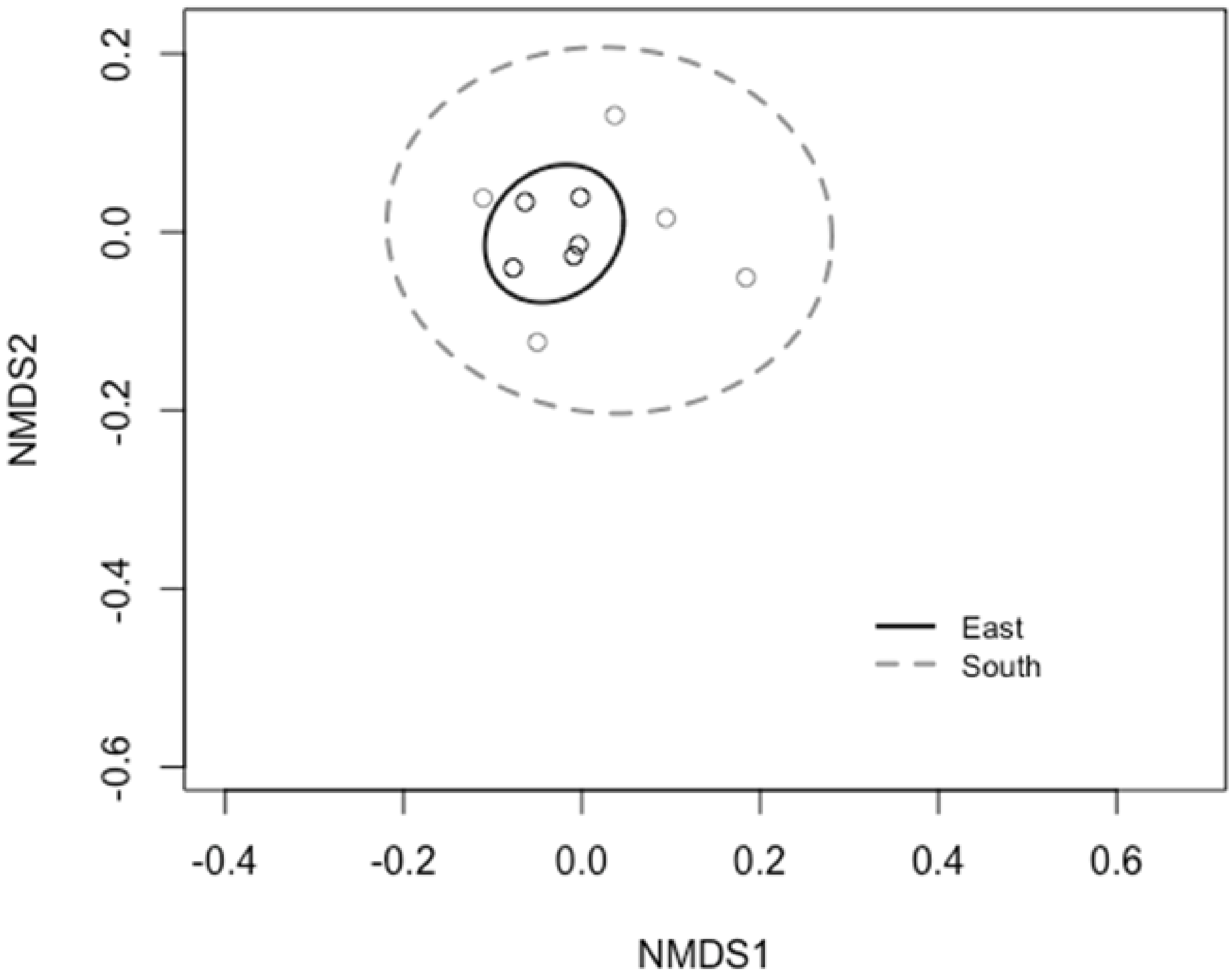

Figure 3-2 NMDS showing species richness for the East and South research sites (Site main effect p=0.006; treatments and measurement periods combined (α=0.1; n=5). Stress: 0.115. Ellipses are showing CI= 95%). The NMDS1 &NMDS2 axis, show the range of the distances reached between the research sites plant community with the Bray-Curtis dissimilarity matrix.

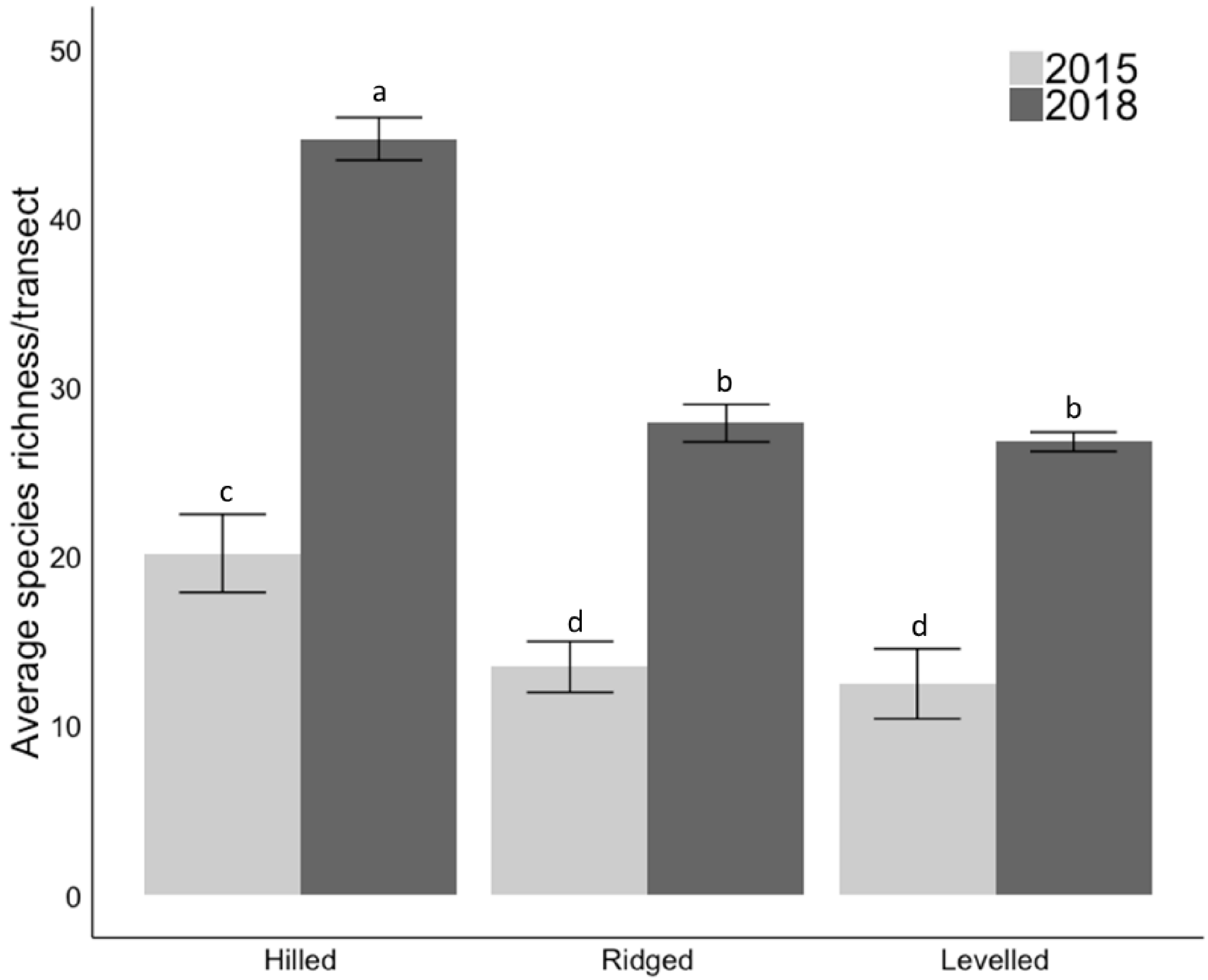

Figure 3-3: Species richness per transect ($180m^2$) among all microtopographic treatments in 2015 and 2018 measurement periods. Research sites were combined for this analysis since the treatment × site × year interaction was not significant (p=0.228). Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10).

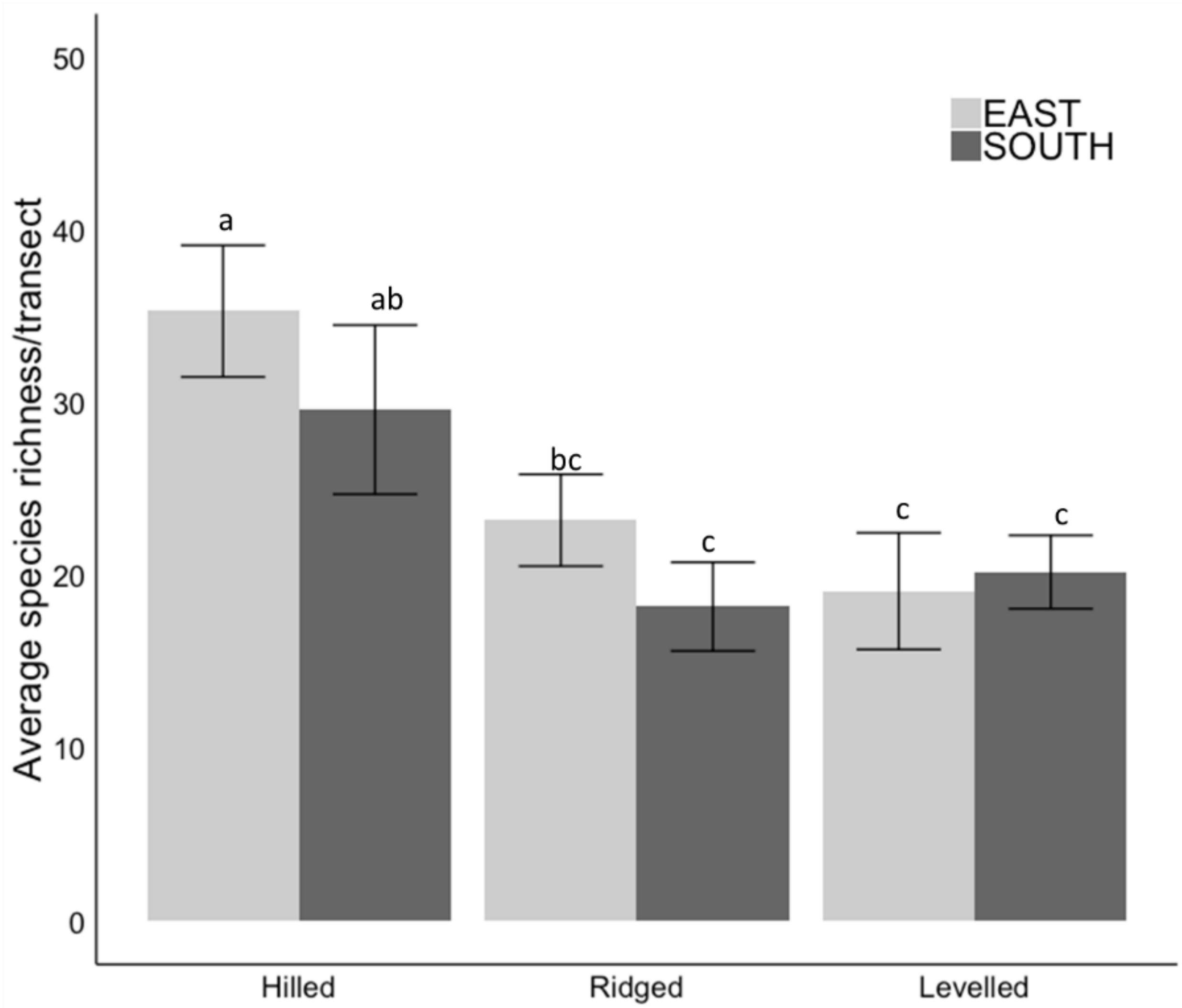

Figure 3-4: Average species richness per transect (180m2) among all microtopographic treatments on East and South sites. Measurement periods were combined for this analysis since the treatment × site × year interaction was not significant (p=0.228). Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10).

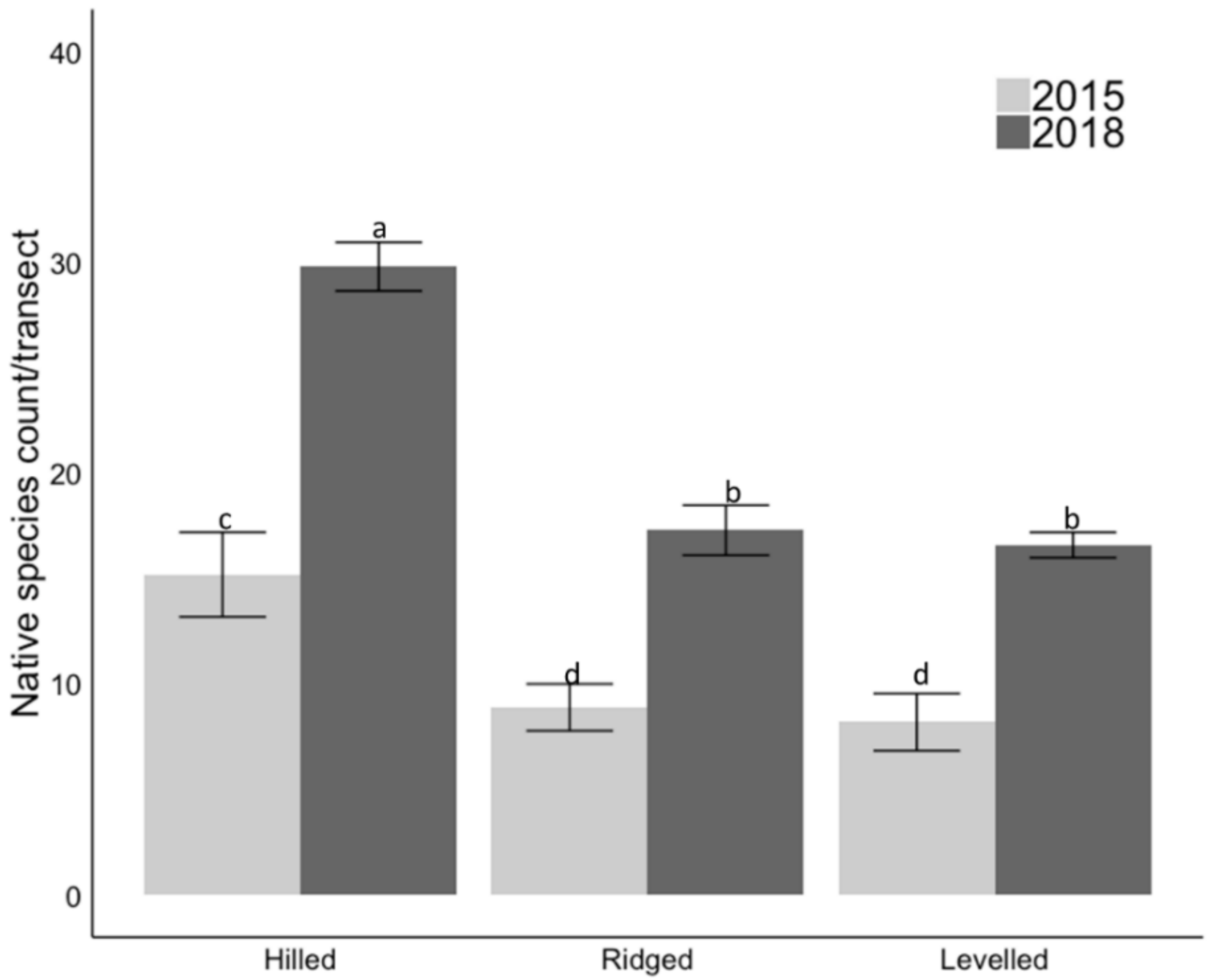

Figure 3-5: Average richness of native species per transect (180m^2) among all microtopographic treatments in 2015 and 2018 measurement periods. Research sites were combined for this analysis since the treatment × site × year interaction was not significant (Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10)

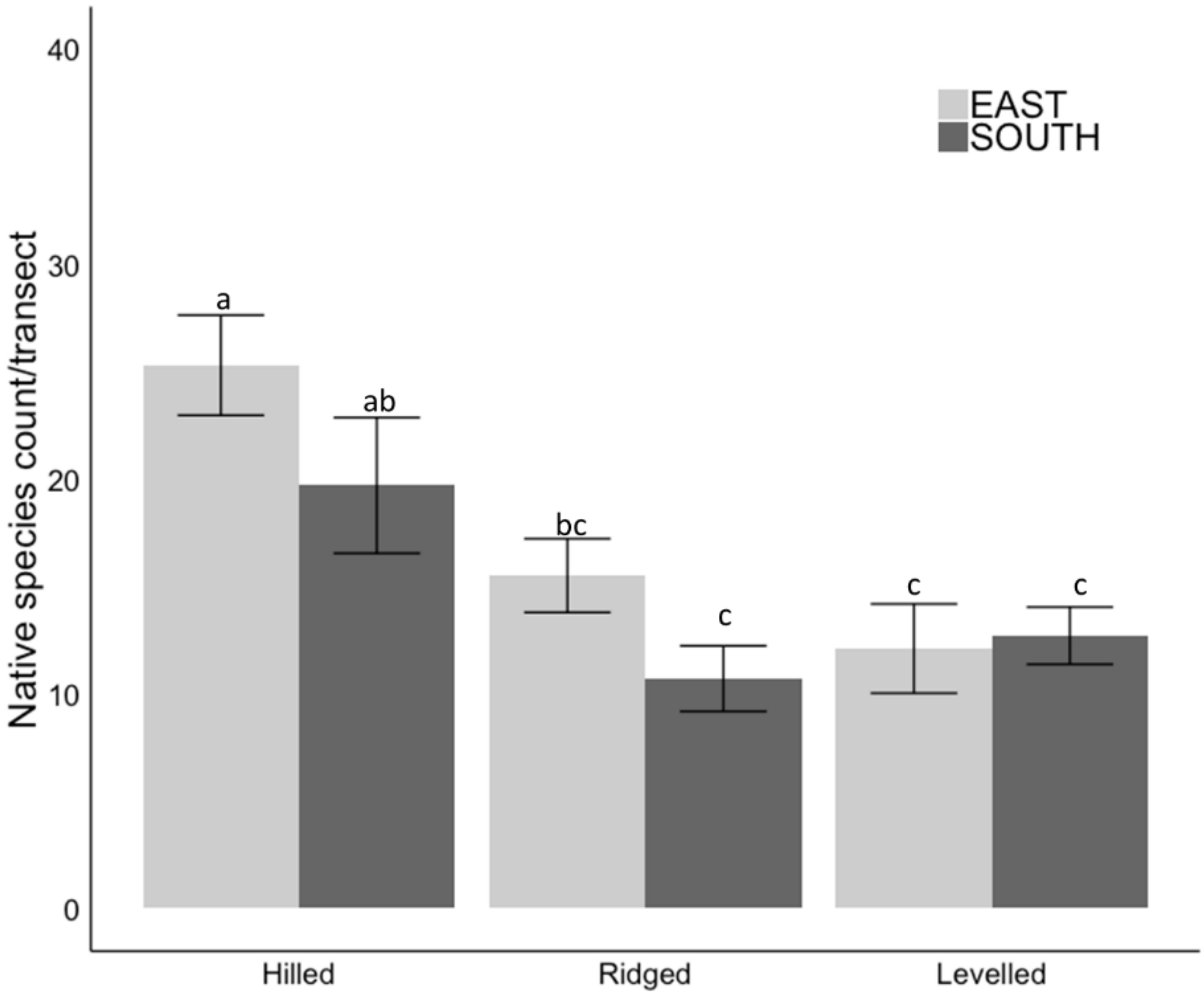

Figure 3-6: Average native species richness per transect ($180m^2$) among all microtopographic treatments in 2015 and 2018 measurement periods. Measurement periods were combined for this analysis since the treatment × site × year interaction was not significant. Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10).

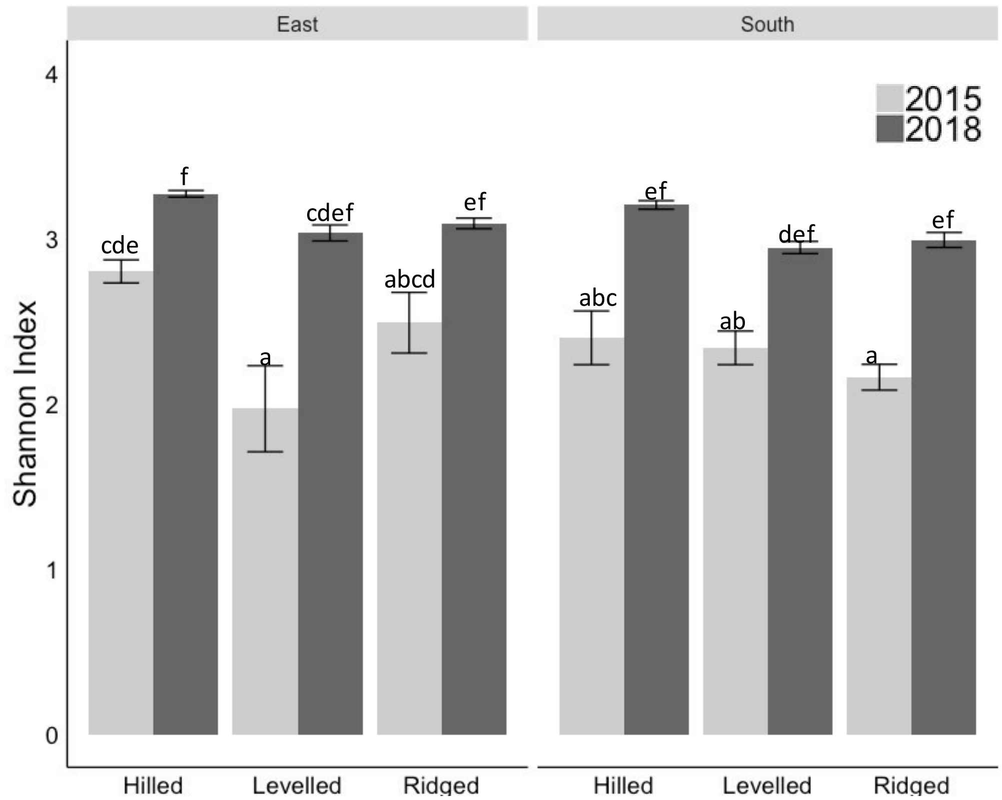

Figure 3-7: Shannon-Wiener diversity index among all microtopographic treatments in 2015 and 2018 measurement periods. Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10).

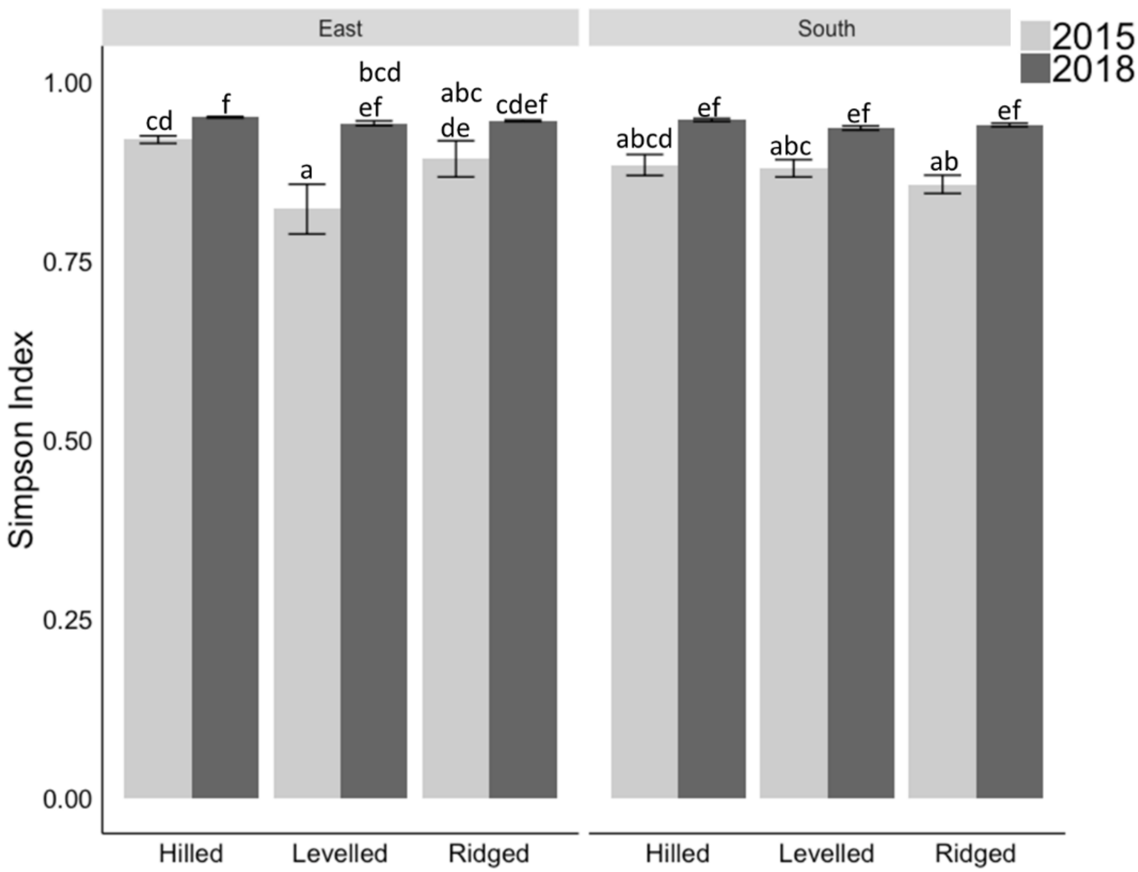

Figure 3-8: Simpson diversity index among all microtopographic treatments in 2015 and 2018 measurement periods. Means with different letters are significantly different and error bars indicate one standard error (α=0.1; n=10)

Table 3-1: Total count of species native species and species unique to treatment, species associated with forest, ruderal, and mesic sites and generalists for each microtopographic treatment in propagule bank study (FFM & PMM results combined)(Melnik et al. 2017), 2015 and 2018. Both research sites (South and East) are combined as they were reconstructed using the same salvaged cover soil material (n=10).

Count	**Initial Propagule Bank (Melnik et al. 2017)**	**2015**			**2018**		
		Hilled	**Ridged**	**Levelled**	**Hilled**	**Ridged**	**Levelled**
All species	33	45	35	37	76	53	57
Species unique to treatment	8	10	2	4	16	3	1
Native	27	35	25	24	35	38	41
Forest	6	7	4	4	16	9	10
Ruderal	8	16	15	17	22	19	19
Mesic	7	8	6	3	17	8	9
Generalist	12	14	10	13	21	17	19

Table 3-2 Average (± SD) of total vegetation cover (%), graminoid cover (%), bryophyte cover (%), forb cover (%) and shrub cover (%) in 2018 for each microtopographic treatment on the South and East site. Different letters indicate significant differences among means (α=0.1; n=5). This table is also presented in Chapter 2.

Research site	**Microtopographic treatment**	**Total vegetation (%)**	**Graminoids (%)**	**Forbs (%)**	**Shrubs (%)**
South	Hilled	30(±5) [a]	5(±0) [a]	20(±5) [a]	3.3 (±0) [a]
	Ridged	50(±10) [b]	8.2 (±5) [a]	35(±5) [a]	4.3 (±0) [a]
	Levelled	50(±20) [b]	10(±5) [a]	35(±20) [a]	6.4 (±5) [a]
East	Hilled	30(±5) [x]	5(±0) [x]	20(±0) [x]	3.7 (±0) [x]
	Ridged	45(±10) [y]	10(±0) [x]	30(±5) [y]	4.8 (±5) [x]
	Levelled	55(±10) [y]	10(±5) [x]	40(±10) [z]	4.6 (±5) [x]

Table 3-3 Count of species that followed different colonization patterns (see Category description) across both research sites

Category	Category Description	Number of species
1	Species in the original propagule bank and never established on the research sites (2015 or 2018)	8
2	Species that were in the original propagule bank and established on site, but only in SPECIFIC microtopographic treatments a) of those species, the ones that were present in 2015 only	4
	b) Species in 2a. that persisted into 2018	2
3	Species that were in the original propagule bank and established on ALL microtopographic treatments a) of those species, the ones that were present in 2015 only	1
	b) Species in 3a. that persisted into 2018	16
4	Species that were NOT in the propagule bank but were established on site in SPECIFIC microtopographic treatments a) of those species, the ones that were present in 2015 only	8
	b) Species in 4a. that persisted into 2018	36
5	Species that were NOT in the propagule bank that most likely colonized from outside in ALL microtopographic treatments a) of those species, the ones that were present in 2015 only.	3
	b) Species in 5a. that persisted into 2018	27

Table 3-4 The ten most dominant species among all treatments on both research sites by habitat preference for the 2015 and 2018 measurement periods. Percentage values indicate the average frequency these species present were found in sample quadrats along transects in each respective treatment (n=10).

Habitat Preference	**Species**	**2015**			**2018**		
		Hilled	**Levelled**	**Ridged**	**Hilled**	**Levelled**	**Ridged**
Forest	*Agropyron trachycaulum*	15(±15)%	30(±30)%	15(±15)%	70(±15)%	70(±15)%	70(±25)%
	Rubus idaeus	10(±10)%	10(±25)%	5(±5)%	35(±15)%	25(±25)%	35(±15)%
Generalist	*Agrostis scabra*	20(±15)%	5(±15)%	10(±10)%	60(±50)%	5(±55)%	5(±50)%
Ruderal	*Crepis tectorum*	10(±10)%	10(±15)%	5(±5)%	50(±15)%	50(±10)%	50(±15)%
	Hordeum jubatum	0%	0%	0%	40(±10) %	35(± 10)%	40(±20)%
	Medicago sativa	0%	10(±20)%	5(±5)%	50(±15)%	60(± 25%)	60(±15%)
	Melilotus alba	5(±5)%	0%	0%	60(±15)%	65(±15)%	60(±15)%
	Melilotus officinalis	0%	0%	0%	70(±15)%	70(±25)%	85(±10)%
	Polygonum convolvulus	45(±40)%	0%	20(±20)%	35(±20)%	25(±25)%	30(±25)%
	Sonchus spp.	40(±25)%	0%	10(±10)%	85(±5)%	80(±15%)	90(±10)%

4. Synthesis and Future Research

4.1 Research Summary

The focus of this thesis was to examine how microsite heterogeneity using varying microtopography and different substrate types influences the growth and establishment of tree seedlings and colonizing vegetation on reclamation sites. It was hypothesized that reclamation areas which provide a greater variety and range of microsites should improve growth of planted seedlings (Chapter 2) and increase the species diversity within the forest vegetation community (Chapter 3) in comparison to sites with more homogenous growing conditions. Greater spatial heterogeneity should not only lead to the early establishment of native species reflective of the original forest propagule bank, but these microsites will also continue to provide habitats in the long term for future establishment and maintain species that are reflective of a native forest understory.

The studies presented in Chapter 2 and Chapter 3 examined the effect of microtopography on three different microtopographic treatments—levelled, ridged, and hilled—on the tree growth and vegetation community. Treatments were applied on two sites that had different aspects (south and east facing slopes) and constructed with PMM and FFM coversoils four years after site construction. Chapter 2 of this thesis focused on assessing the relationship between these three microtopographic treatments and the growth of planted seedlings: trembling aspen, jack pine, and white spruce, while also evaluating the natural regeneration of trembling aspen and balsam poplar. Chapter 3 investigated the effect of microtopographic treatments on the establishment of the vegetation community, such as species richness, and data was used as a point of comparison to the results of the propagule bank study and the initial data collection (2015).

As predicted, results from both chapters presented evidence that treatments offering a greater range of spatial heterogeneity through mechanical site preparation (i.e., hilled or ridged) benefited planted seedling growth, natural regeneration of deciduous trees, and the establishment of native vegetation. The main trends found in chapter 2 were that larger microtopographical features (i.e., mounds or ridges) created greater gradients of growing conditions, such as moisture/temperature regimes or shelter from wind/solar radiation, which provided suitable growing conditions for tree growth and establishment. When comparing the growth responses to the treatments among the three planted species four growing seasons in, trembling aspen appeared to be more responsive to the higher microtopographic treatments, followed by jack pine, with white spruce showing little response. Another significant advantage that was observed from the addition of microsites was the improvement of long-distance wind-dispersed tree species recruitment, such as trembling aspen and balsam poplar. In chapter 3, the main findings were that treatments which provided more microsites and higher exposure to both coversoils (hilled treatment) both promoted higher species richness (Shannon-Wiener and Simpson indices), and showed an increase presence of native species when compared with treatments that had homogenous conditions (levelled treatment). This increase of species richness was most likely aided by the availability of two different propagule banks and the difference of seedbed microsite (FFM and PMM) establishment conditions that are suitable to larger number of species.

Studying both tree growth and forest vegetation establishment elements concurrently provided an opportunity to gain a clearer understanding of how land reclamation sites can recover over time at both the tree growth and vegetation establishment levels. Results showed that variation in microtopography can benefit both the tree growth and the vegetation establishment in early phase of reclamation. Interestingly, the hilled treatment was able to highlight the advantage of having the highest native species richness but by also having the lowest vegetation cover among all treatments. This means the hilled treatment microsites could have a significant reduction in competition effect for both the planted seedlings and native vegetation establishment. Some other early results that emerged from this study that could support the dependency between the tree growth and the understory development are that some species might require a forest canopy or sheltered growing conditions to be maintained. For example, Category 1 species (such as *Cornus canadensis*, *Maianthemum canadense*, *Mitella nuda*) were present in the propagule bank but never established on the sites. However, these types of species tend to be shade tolerant species in natural forests, therefore they could emerge at a later time in the restoration process once the site is more matures and has a denser tree canopy.

Another important take away from both the tree and vegetation studies is that they displayed different results for different research sites (South and East sites). Overall, the planted seedlings performed better and more natural tree recruitment occurred on the South site whereas the East site resulted in a more diverse and native vegetation community. Seedlings most likely grew better on the South site by having the advantage of higher exposure to solar radiation while microsites provided milder growing conditions that could have alleviated these stressful growing conditions. For example, mound toes or trough microsites were able to offer shelter from wind and/or sun as well provide a higher moisture availability to the trees and the colonizing species. As for the East site, the higher species richness found was most likely due to having milder growing conditions and offering more suitable establishing conditions for a larger number of species originating from the soil seedbank.

4.2 Study Limitations and Future Research

Even though this study spanned only four growing seasons, the greater number of native forest species and improved tree growth found in the hilled treatment are encouraging results showing a trajectory of success for these research sites. That being said, results only captured the initial forest recovery response, therefore only depicting a short time frame in the progression of the forest establishment and succession to maturity—a process which can span over decades. Consequently, in order to better understand long-term recovery, the recommended next steps for this study would be to continue examining the response of tree growth, tree ingress, and the shift in the vegetation community in relation to the different microtopographic treatments over the next few decades. This is especially valuable when studying later successional species such as white spruce or other slow-growing conifers, which are important boreal forest species.

Other future evaluations on these research sites could include assessing the effect of various microtopographic positions to other species groups such as lichen, moss, and fungi. Additionally, data that could also further benefit the study to better understand the revegetation of these sites would be considering factors such as outside seed sources, seed dispersal distance, seedbed conditions, and growing conditions during dispersal. Some important native species were present in the propagule bank but never established on sites (Category 1 species), which suggests that this is an area for future research. Examining the microsite preference of these species could inform required growing conditions for these native species. As mentioned above, a longer-term study (planted or natural) would be beneficial in order to see if these species require a more closed overstory to emerge from the seedbank.

Another suggested next step of this research would be to assess the relationship between tree canopy closure (Chapter 2) and the development of understory vegetation (Chapter 3) on these reclaimed sites. In this research project, these two components were looked at independently, but it would be valuable to look at how they interact with one another over longer temporal horizons. Understanding the factors influencing the forest understory establishment along with tree canopy establishment is a crucial component to forest restoration, because the understory is the foundational component of functioning forest ecosystems but has not been well-studied in land reclamation settings. In order to accelerate the canopy closure while simultaneously maintaining the development of a native understory to obtain resilient and sustainable multi story forest stands, long term evidence is required.

A main take-away from this study is that the South and East sites each resulted in different outcomes for both the tree growth and vegetation established due to their different growing conditions. Therefore, it would have been beneficial to investigate other slope aspects, such as west, north, or even a non-sloped site to do a full comparison of results and explore the role of microtopography for different slope aspects. Furthermore, the climate data for the region during the study was warmer and drier than the average climate, meaning sites were more stressed and most possibly had water limiting conditions (especially the South site due to more solar radiation) which most likely influenced the differences observed between the two sites. Some additional data that could have helped to contribute to a better understanding of the growing conditions (ie., solar radiation) between all the different microsites between the East and South site could help separate the effect between both. For future research, it would be interesting to assess if the South site maintains its advantage with the increased tree growth or if the East site maintains a higher species diversity over time.

4.3 Management Implications

This study highlighted that using a variety of microsites and coversoils on reclamation sites can provide a greater range of habitat types and diverse seedbanks which resulted in an acceleration of tree growth, natural colonization of deciduous trees, and revegetation of native species. Although all three microtopographic treatments were suitable for plant establishment and tree growth, the treatments providing heterogeneous growing conditions (hilled and ridged) significantly benefitted with overall better tree growth and native vegetation establishment. From these observations, it is suggested that future reclamation practices incorporate the use of microtopography (when possible) in order to fully gain this advantage. With improved tree growth and increased density, forests should theoretically attain canopy closure earlier which will encourage the understory vegetation establishment, which is an important forest succession stage that often gets overlooked in reclamation.

Additionally, this study showed that while microtopography might have played different roles on different site aspects, both research sites still performed better with heterogeneous sites than homogenous ones. For aspects such as South facing, implementing microsites and different coversoils can help alleviate stressful growing conditions on sites that are exposed to extreme growing conditions. For example, the PMM (mounds) in the hilled treatment could have contributed to cooler growing conditions and have higher water retention which resulted in better tree growth and recruitment of tree seedlings. In current forest reclamation practices, a common challenge is that there is a low availability of FFM coversoil, therefore being able to use PMM and understand its advantage when paired with microsites can be extremely valuable in practice. It is important to note that by using both FFM and PMM seedbanks on sites increases the establishment species diversity on reclamations sites which can reduces the invasion of non-native species that affect the overall reclamation success.

To conclude, mechanical site preparation into ridges or mounds is more advantageous than just a levelled surface. Additionally, besides the many ecological benefits of microtopography use mentioned above, it is important to note that the hilled treatment required significantly less time to construct and had a reduced the number of equipment travel passes across the site. This not only reduced operational costs compared to the ridged and levelled treatments, but also mitigated other negative impacts on coversoils, such as excessive compaction.

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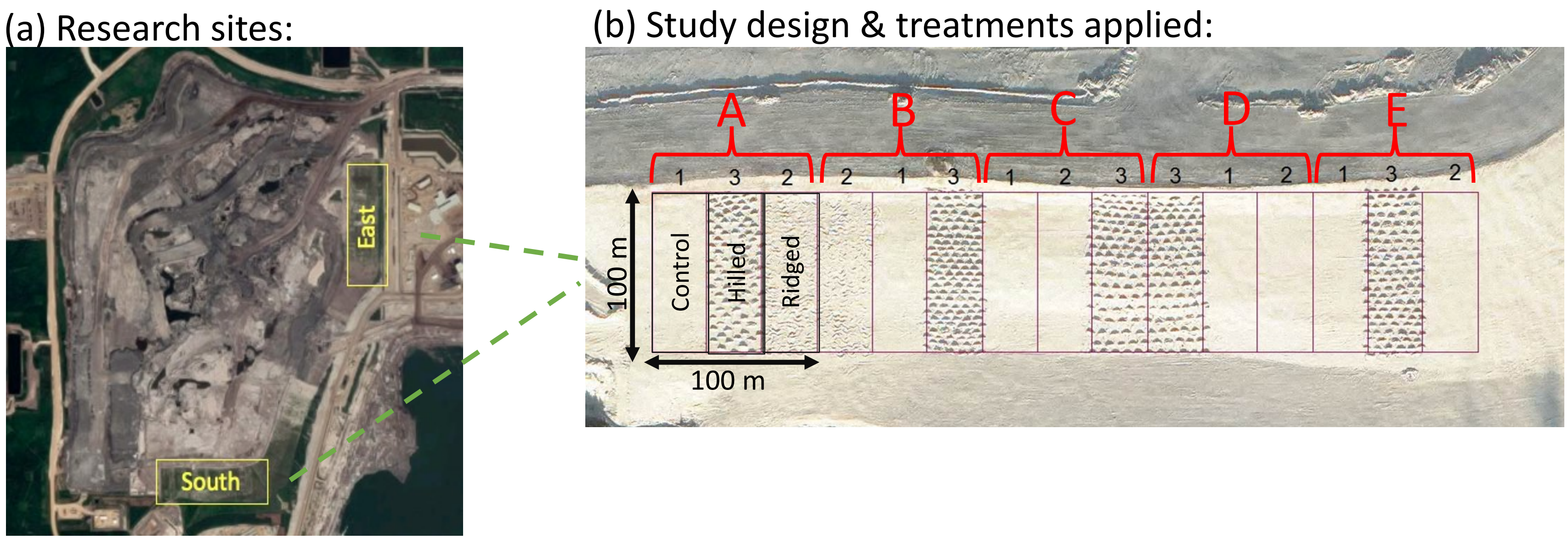
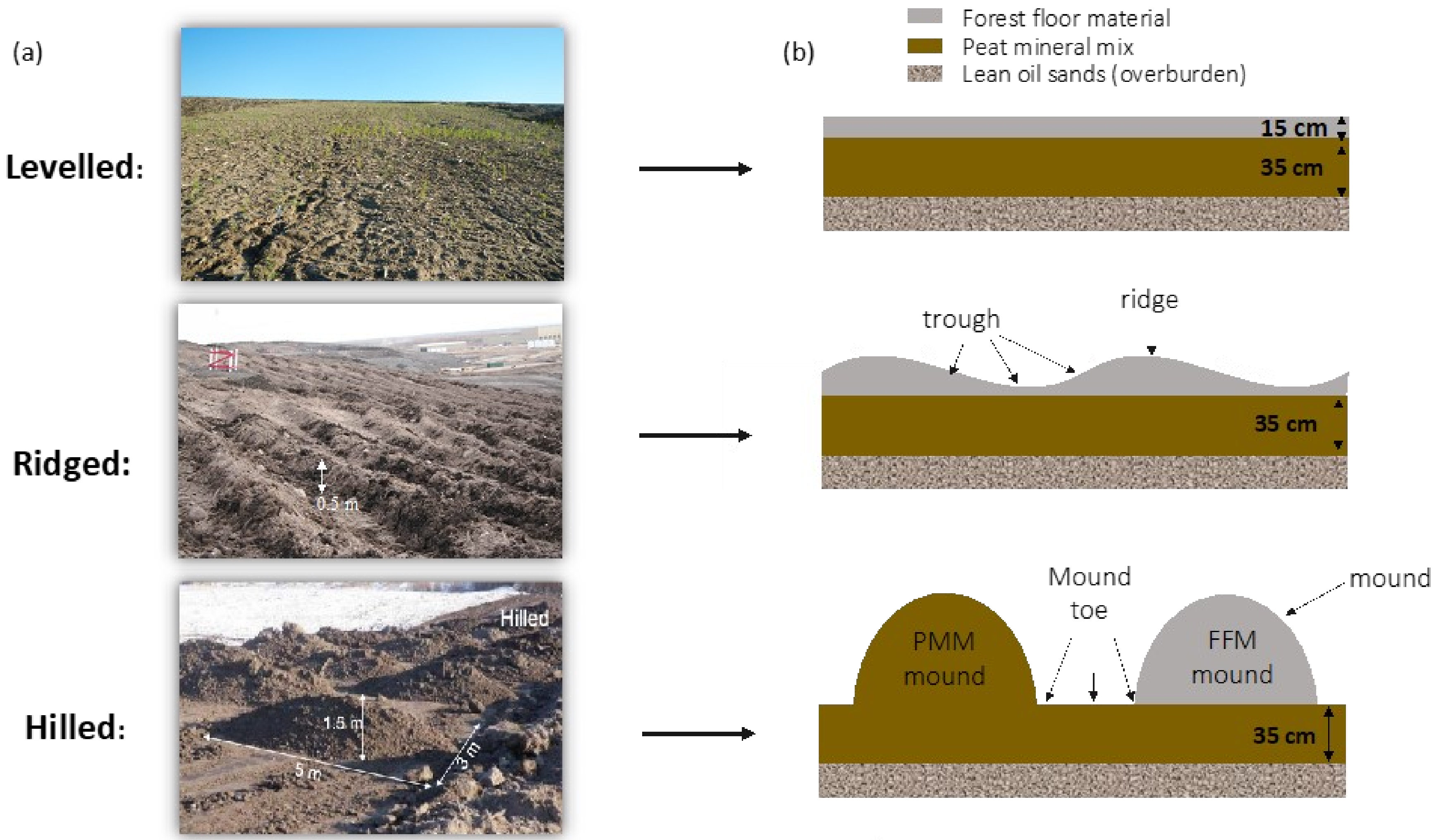
Appendices

Appendix 1:(a) Aerial view of the overburden dump with the South and East sites outlined in yellow; (b) Aerial view showing the South site in the winter of 2016 with five replicate blocks (A-E) and each block with the three microtopographic treatments.



Appendix 2:Images of the three microtopographic treatments on the research sites (a) and the associated material layering scheme (b) for the microtopographic treatments with a description of microsite type within the ridged and hilled treatments and with bars indicating the scale and size of microsite features.



Appendix 3:Fort McMurray's, mean temperature total precipitation annually and growing season from year 2015 to 2018 (Environment Canada, 2018)

Year	Annual		Growing season	
	Temperature (°C)	Total precipitation (mm)	Temperature (°C)	Total precipitation (mm)
2015	3	274.6	14.4	194.8
2016	3	398.6	15.7	321.5
2017	2.8	210.4	15.6	120.9
2018	0.9	271.7	14.6	255.6

Appendix 4:Average (± SD) of height (cm), root collar diameter (mm) and stem volume (cm^3) of top performing aspen in 2018 for each microtopographic treatment on the South and East site. (α=0.1; n=5).

Research Site	Treatment	Height (cm)	Root collar diameter (mm)	Stem volume (cm^3)
South	Hilled	223 (±17)^a	29(±2)^a	567(±126)^a
South	Ridged	186(± 12)^b	20(±2)^b	221(±49)^b
South	Levelled	156(± 25)^c	16(±4)^c	129(±88)^c
East	Hilled	109(±9)^x	11(±1)^x	39(±12)^x
East	Ridged	98(±26)^x	10(±4)^x	35(±29)^x
East	Levelled	99(±33)^x	11(±3)^x	45(±29)^x

Appendix 5: Average (± SD) of height (cm), root collar diameter (mm) of jack pine (Pj) and white spruce (Sw) in 2018 for each microtopographic treatment on the South and East site. (α=0.1; n=5).

Research Site	**Treatment**	**Species**	**Height (cm)**	**Root collar diameter (mm)**
South	Hilled	Pj	103(±15) [a]	19(±3) [a]
South	Ridged	Pj	97(±7) [a]	18(±2) [a]
South	Levelled	Pj	80(±5) [b]	12(±1) [b]
South	Hilled	Sw	85(±20) [a]	18(±3) [a]
South	Ridged	Sw	73(±5) [a]	15(±1) [b]
South	Levelled	Sw	70(±14) [a]	13(±2) [b]
East	Hilled	Pj	52(±8) [x]	8(±3) [x]
East	Ridged	Pj	59(±7) [x]	10(±2) [x]
East	Levelled	Pj	54(±5) [x]	9(±2) [x]
East	Hilled	Sw	44(±2) [x]	8(±1) [x]
East	Ridged	Sw	46(±3) [x]	8(±1) [x]
East	Levelled	Sw	49(±5) [x]	9(±2) [x]

Appendix 6:All species found on propagule bank, 2015 and 2018 measurement period for each microtopographic treatment. Life strategy (P = perennial, A= annual, B= biennial), native to Alberta, category of maintenance and establishment (see methods for more details).

Species (scientific name)	Life Strategy	Native to Alberta	Habitat Type	Category	Propagule Bank	Treatment: Levelled		Treatment: Ridged		Treatment: Hilled	
						2015	2018	2015	2018	2015	2018
***Achillea millefolium* L.**	P		generalist	4b		✓	✓		✓		✓
***Achillea sibirica* Ledeb.**	P	✓	generalist	4b		✓					✓
***Actaea rubra* (Aiton) Wildenow**	P	✓	forest	4a						✓	
***Agropyron trachycaulum* (Link) Malte**	P	✓	forest	5b		✓	✓	✓	✓	✓	✓
***Agrostis scabra* Willd.**	P	✓	generalist	3b	✓	✓	✓	✓	✓	✓	✓
***Alnus viridis* (Chaix.) D.C.**	P	✓	forest	3b	✓		✓		✓		✓
***Amelanchier alnifolia* Nutt.**	P	✓	forest	4b							✓
***Aquilegia brevistyla* Hook.**	P	✓	generalist	4b					✓		
***Artemisia campestris* L.**	P	✓	forest	5b			✓		✓		✓
***Aster borealis* (T. & G.) Prov.**	P	✓	mesic	4b					✓	✓	✓
***Aster ciliolatus* Lindl.**	P	✓	forest	5b			✓		✓		✓
***Aster* sp. L. (unknown #3)**	P	✓	forest	4b							✓
***Aster* sp. L. (unknown #1)**	P	✓	forest	4b					✓		✓
***Aster* sp. L. (unknown #2)**	P	✓	forest	4b			✓				✓
***Aster* sp. L. (unknown #4)**	P	✓	forest	4b							✓
***Axyris amaranthoides* L.**	A		ruderal	5a		✓		✓		✓	
Beckmannia syzigachne (Steudel) Fernald	A	✓	mesic	4a		✓					
***Betula papyrifera* Marshall**	P	✓	forest	1	✓						
***Bromus ciliatus* L.**	P	✓	generalist	5b			✓		✓		✓

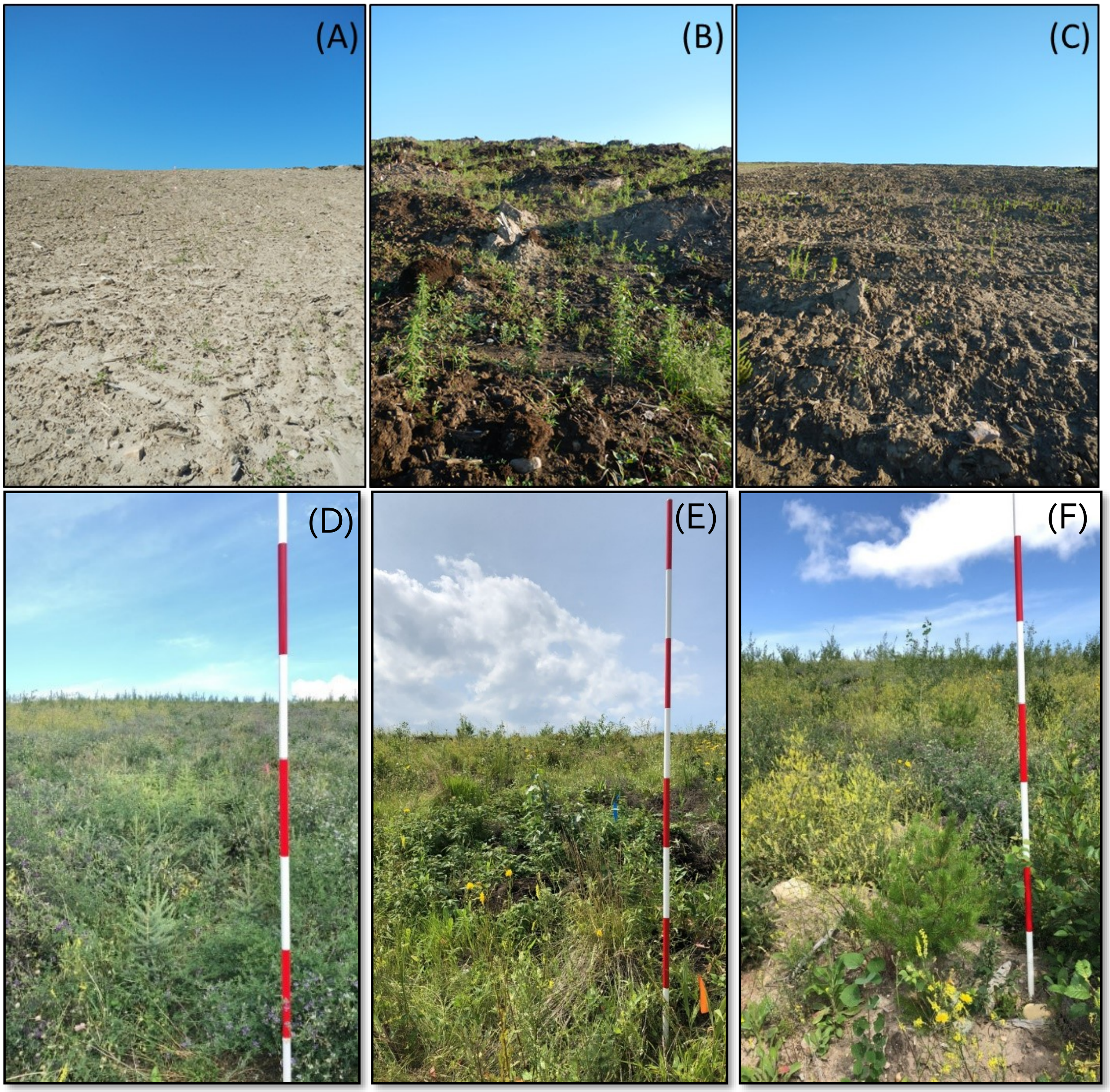
***Calamagrostis canadensis* (Michx.) Beauv.**	P	✓	generalist	5b		✓	✓	✓	✓	✓	✓
***Carex aenea* Fern.**	P	✓	forest	5b			✓		✓		✓
***Carex aquatilis* Wahlenb.**	P	✓	mesic	4b			✓				✓
***Carex bebbii* Olney ex. Fern.**	P	✓	mesic	5b			✓		✓		✓
***Carex interior* Bailey**	P	✓	mesic	4b			✓				
***Carex* sp. (unknown)**	P	✓	mesic	3b	✓	✓	✓	✓	✓	✓	✓
***Chenopodium album* L.**	A		ruderal	3b	✓	✓	✓	✓	✓	✓	✓
***Chenopodium capitatum (*L.) Ambrosi**	A	✓	ruderal	4b				✓		✓	
***Cornus canadensis* L.**	P	✓	generalist	1	✓						
***Cornus stolonifera* Michx.**	P	✓	generalist	4b		✓		✓			✓
***Corydalis aurea* Willdenow**	A, B	✓	generalist	5a		✓		✓		✓	
***Crepis tectorum* L.**	P		ruderal	3b	✓	✓	✓	✓	✓	✓	✓
***Deschampsia cespitosa* (L.) Beauv.**	P	✓	mesic	4b							✓
***Dracocephalum parviflorum* Nutt.**	A, B	✓	forest	4a						✓	
***Elymus canadensis* L.**	P	✓	ruderal	4b							✓
***Epilobium angustifolium* L.**	P	✓	generalist	3b	✓	✓	✓	✓	✓	✓	✓
Epilobium ciliatum Rafinesque	P	✓	forest	2a	✓	✓				✓	
***Equisetum arvense* L.**	P	✓	generalist	3b	✓	✓	✓	✓	✓	✓	✓
***Erigeron canadensis* L.**	A		ruderal	3b	✓		✓		✓	✓	✓
***Erigeron lonchophyllus* Hook.**	A	✓	mesic	4b					✓		
***Fragaria virginiana* Duschene**	P	✓	forest	4b			✓				✓
***Galium boreale* L.**	P	✓	generalist	4b			✓				✓
Galium trifidum L.	P	✓	mesic	2a	✓			✓		✓	
***Geranium bicknelii* L.**	A, B	✓	ruderal	3b	✓	✓	✓	✓	✓	✓	✓
***Geum* sp. L.**	P	✓	generalist	4b			✓				✓
***Hordeum jubatum* L.**	P	✓	ruderal	4b		✓	✓		✓	✓	✓
Impatiens noli-tangere L.	A	✓	forest	4a				✓			

***Juncus balticus* Willd.**	P	✓	mesic	3b	✓		✓		✓		✓
***Lathyrus ochroleucus* Hook.**	P	✓	generalist	5b		✓	✓	✓	✓	✓	✓
Ledum groenlandicum (Oeder)	P	✓	generalist	1	✓						
***Lepidium densiflorum* Schrad.**	A, B	✓	ruderal	3b	✓		✓		✓		✓
Maianthemum canadense Desf.	P	✓	generalist	1	✓						
***Matricaria matricarioides* DC.**	A/B		ruderal	4a		✓					
***Medicago sativa* L.**	P		ruderal	5b		✓	✓	✓	✓	✓	✓
***Melilotus alba* Desr.**	A, B		Ruderal	5b		✓	✓	✓	✓	✓	✓
***Melilotus officinalis* (L.) Lam.**	A,B		Ruderal	4b		✓	✓	✓	✓		✓
***Mentha arvensis* L.**	P	✓	mesic	4b					✓	✓	✓
***Mitella nuda* L.**	P	✓	generalist	1	✓						
***Oryzopsis* sp. Michx.**	P	✓	forest	4b							✓
***Phleum pratense* L.**	P		ruderal	5b			✓		✓		✓
***Plantago major* L.**	P		ruderal	5b		✓	✓	✓	✓		✓
***Poa palustris* L.**	P	✓	generalist	5b		✓	✓	✓	✓	✓	✓
***Poa pratensis* L.**	P	✓	ruderal	5b			✓		✓		✓
***Polygonum aviculare* L.**	A	✓	ruderal	5a		✓		✓		✓	
***Polygonum convolvulus* L.**	A		ruderal	5b		✓	✓	✓	✓	✓	✓
***Polygonum lapathifolium* L.**	A	✓	ruderal	5b		✓		✓	✓	✓	✓
***Populus balsamifera* L.**	P	✓	generalist	5b			✓		✓		✓
***Populus tremuloides* Michx.**	P	✓	forest	5b			✓		✓		✓
***Potentilla fruticosa* L.**	P	✓	forest	4b							✓
***Potentilla norvegica* L.**	A	✓	generalist	5b		✓	✓	✓	✓	✓	✓
***Prunus pensylvanica* L.f.**	P	✓	forest	5b		✓	✓	✓	✓	✓	✓
***Ranunculus sceleratus* L.**	A,P	✓	mesic	2a	✓			✓		✓	
***Ribes glandulosum* Grauer**	P	✓	generalist	4b			✓		✓		
***Ribes oxyacanthoides* L.**	P	✓	generalist	2b	✓		✓				✓
***Ribes* sp. L.**	P	✓	generalist	4b					✓	✓	✓

***Rorippa palustris* (L.) Besser**	A, B	✓	mesic	3a	✓	✓		✓		✓	
***Rubus idaeus* L.**	P	✓	forest	3b	✓	✓	✓	✓	✓	✓	✓
***Rubus pubescens* Rafinesque**	P	✓	generalist	1	✓						
Salix discolor Muhlenberg	P	✓	mesic	4a						✓	
***Salix exigua* Nutt.**	P	✓	mesic	5b			✓		✓		✓
***Salix* sp. L. (unknown #3)**	P	✓	mesic	4b			✓				✓
***Salix* sp. L. (unknown #4)**	P	✓	mesic	4b							✓
***Salix* sp. L. (unknown #5)**	P	✓	mesic	4b							✓
***Salix* sp. L. (unknown #7)**	P	✓	mesic	4b							✓
***Salix* sp. L. (unknown #8)**	P	✓	mesic	5b			✓		✓		✓
***Salix* sp. L. (unknown #2)**	P	✓	mesic	4b			✓				✓
***Salix* sp. L. (unknown #5)**	P	✓	mesic	4b					✓		
***Salsola kali* (pestifer) L.**	A		ruderal	5b		✓	✓		✓	✓	✓
***Scutellaria galericulata* L.**	P	✓	mesic	4b						✓	✓
***Sherpherdia canadensis* (L.) Nutt**	P	✓	generalist	5b			✓		✓		✓
***Silene pratensis* (Rafn) Godron & Gren**	A, B, P		ruderal	4b							✓
***Smilacina* spp. Desf.**	P	✓	forest	4a						✓	
***Sonchus* spp. L.**	P		ruderal	3b	✓	✓	✓	✓	✓	✓	✓
Sphenopholis intermedia (Rydb.) Rydb.	P	✓	mesic	1	✓						
***Stachys palustris* L.**	P	✓	generalist	4b			✓				✓
***Taraxacum officinale* Weber**	P		ruderal	3b	✓	✓	✓	✓	✓	✓	✓
***Tragoporon dubius* Scop.**	A,B		Ruderal	4b			✓				✓
***Trientalis borealis* Rafinesque**	P	✓	generalist	4a						✓	
***Trifolium hybridum* L.**	P		ruderal	3b	✓	✓	✓	✓	✓	✓	✓
***Trifolium pratense* L.**	B, P		ruderal	5b			✓		✓		✓
***Typha latifolia* L.**	P	✓	mesic	2a	✓			✓			✓
***Urtica dioica* L.**	P		generalist	3b	✓		✓		✓	✓	✓

Vaccinium myrtilloides Michx.	P	✓	forest	1	✓						
***Vicia americana* Muhl.**	P	✓	generalist	5b		✓	✓	✓	✓	✓	✓
***Viola adunca* J.E. Smith**	P	✓	forest	5b			✓	✓	✓	✓	✓

Appendix 7: Control (A), hilled (B) and ridged (C) treatment plots at the end of the first growing season in August 2015 compared with Control (D), hilled (E), and ridged (F) treatments in 2018.



Appendix 8: Overview of the treatment plots on the south-facing slope in early spring 2015 before the first growing season following treatment construction. Photo credit: Rob Vassov

Appendix 9: A control treatment plot in the foreground with the hilled treatment plot in the background at the end of the first growing season in August 2015.