Influence of Capping Depth on Forest Reclamation Success in the Athabasca Oil Sands

by

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Abstract

Surface mining in the Athabasca Oil Sands region (AOSR) of Alberta, Canada, has led to the removal of over 760 km^2 of boreal forest. Successful reclamation of these areas must return them to an equivalent capability boreal forest community. Clearwater overburden is a saline byproduct of surface mining, used in the construction of large-scale, out-of-pit landforms. To reclaim these landforms, a capping treatment is placed upon them to provide a suitable rooting media, and ensure the re-establishment of a forest vegetation community.

Using a range of peat-mineral mix (PMM) and subsoil capping thicknesses on existing Clearwater overburden reclamation sites in the AOSR, I sought to assess the long-term effects of soil cover thickness and material on boreal forest growth and development. I compared the tree height and dbh of planted trembling aspen (*Populus tremuloides*) and white spruce (*Picea glauca*) over 19 years, and at 9 and 13 years post-planting. As of 19 years post-planting at an overburden capping trial, I was able to observe a trend towards taller trees in the thickest capping treatment (100 cm, versus 50 and 35 cm), but only for the fast-growing, early-successional trembling aspen. The slower-growing, typically mid-successional white spruce did not display differential responses among the capping treatments. Examining 13-year-old sites with 100 cm of capping demonstrated high site-to-site variability in growth outcomes. Comparing the performance of the 100cm capping treatment against a thicker, 150 cm treatment at 9-year-old sites found no height or dbh differences for either species.

I additionally compared understory communities over 19 years and at 13 years after the planting of trees. Over 19-years, communities trended towards expectations for more natural boreal forest communities (increased litter cover, increased presence of forest species) but the continued presence of introduced and noxious species was notable. At 13 years, high site-to-site variability distinguished the sites of the same treatment (100 cm) but communities with different treatments (100cm, 50 cm, and 35 cm) on the same site were more similar to each other.

I conclude that trembling aspen growth suggests 100cm of PMM capping (versus 50 cm or 35 cm) is a sufficient thickness for this species, and further investment in 150 cm of capping is not justifiable at this time. White spruce did not display any differences in growth between any of the treatments sampled to date. Understory communities have progressed towards some expectations of natural boreal forest communities, but high site-to-site variability for the same treatment indicates that PMM capping treatment may not play a dominant role in this process.

Preface

Chapters 2 and 3 of this thesis will be prepared for submission as a single journal article. This will be entitled "Venskaitis, S., Macdonald, S.E., Purdy, B.G. 20xx. Influence of capping depth on long-term white spruce, trembling aspen, and understory reclamation success in the Athabasca Oil Sands". The present study was designed by myself and S.E. Macdonald, and I collected (2018) and analyzed the data (2001-2018). B.G. Purdy conceived the original study, and led data collection (2001-2012). I wrote the original draft of the chapters, and S.E. Macdonald and B.G. Purdy contributed to editing the chapters.

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Chapter 1: General Introduction

The circumpolar boreal forest dominates much of the Northern Hemisphere, covering approximately 1.16 billion ha (Olson, Watts, & Allison, 1983). This forest operates under a cycle of disturbance ecology, whereby natural disturbances trigger renewal and succession of the forest (Angelstam & Kuuluvainen, 2004; Hart & Chen, 2006; Chen & Popadiouk, 2002; Johnson, 1992). These disturbances open up the canopy, allowing light to reach to forest floor, which promotes growth of understory and young trees (Hart & Chen, 2006). Severity of disturbance is variable, with fire and storms considered stand-replacing disturbances (Angelstam & Kuuluvainen, 2004), while smaller disturbances, such as insect outbreaks, tend not to change stand dynamics as greatly (Hart & Chen, 2006).

Large-scale or stand-replacing disturbances, such as the aforementioned forest fires, result in the death of most vegetation, opening up space and releasing nutrients into the system, facilitating forest regeneration (Johnson, 1992; Bergeron et al., 2014). This regeneration can happen in many different stages, as certain tree species may dominate the forest in early stages of renewal, and others may come to dominate the forest later, as the early species reach the end of their life-span (Chen & Popadiouk, 2002). The processes of renewal and succession are aided not only by the removal of trees, but also by what is left behind. Natural disturbances will often leave behind some mature trees which can provide a seed source for future regeneration (Chen & Popadiouk, 2002). In addition, while these disturbances, especially large-scale fire, can remove some of the organic component at the surface of forest soils, some organic material will be left behind (Hart & Chen, 2006; Greene et al., 2007).

Large-scale anthropogenic disturbances have been introduced to this system through industry, such as forestry and mining (Macdonald, Quideau, & Landhäusser, 2012; Simard et al, 2001; Bergeron et al., 2001). These disturbances are generally more severe than natural processes, especially in the case of mining, where the natural environment is completely removed and must be entirely rebuilt (Macdonald, Quideau, & Landhäusser, 2012). Industrial disturbances also operate on different timelines than natural disturbances, as industrial operations often continue for many years or decades before sites become available for reclamation; for example, mines in the Athabasca Oil Sands which have been operating since 1967 (Brandt et al., 2013). As a result, and because these disturbances lack natural precedent, human intervention is required to return affected forests to a more natural state (Macdonald, Quideau, & Landhäusser, 2012; Macdonald et al., 2015a). Soils must be rebuilt, and the potential lack of seed bank must be addressed by planting native trees and and using other approaches to facilitate establishment of understory vegetation (Audet, Pinno, & Thiffault, 2014; Macdonald, Quideau, & Landhäusser, 2012; Macdonald et al., 2015a). This thesis will address the reclamation of boreal forest after disturbance by oil sands mining in the Athabasca Oil Sands Region (AOSR) of Alberta.

1.1 Overview of Mining and Reclamation in the Athabasca Oil Sands

The Athabasca Oil Sands in Northern Alberta, Canada contain vast bitumen deposits. The 3% of these deposits that are available to surface mining lie under 475 000 ha of boreal forest, with approximately 142 800 ha approved by government for mining operations (Rooney, Bailey, & Schindler, 2012; Alberta Environment and Parks, 2017). In order to access these bitumen deposits, the overlying forest and peat are removed (Rowland et al., 2009). This peat can be used in the reclamation of other sites, and is either stockpiled for use later, or can be directly placed on a new reclamation site if available (Béasse, Quideau, & Oh, 2015; Fung & Macyk, 2000). Underneath the peat is a layer referred to as the overburden which is also removed, but it can also be used in later reclamation (Rowland et al., 2009; Kessler et al., 2010). Once the overburden is removed, mining of the bituminous sands begins (Rowland et al., 2009; Fung & Macyk, 2000).

After oil sands mining of an area is complete, reclamation must begin. The Environmental Protection and Enhancement Act of Alberta requires companies to return the mined area to an equivalent land capability to that which was removed (Government of Alberta, 2005; Alberta Environment and Parks, 2010). Overburden material is placed first to create landform topography capable of supporting both upland and wetland reclamation sites (Chapman, Barbour, & O'Kane, 2006). To improve upon the nutrient content and water-holding capacity of overburden materials and facilitate vegetation growth on upland sites, a layer of material better-suited to rooting is placed over the overburden; this is known as a capping treatment (Macdonald, Quideau, & Landhäusser, 2012; Rowland et al., 2009). In some cases, tailings sand, the material left after bitumen is extracted from the bituminous sands, is used instead of overburden (Macdonald, Quideau, & Landhäusser, 2012); this material has different requirements for reclamation and was not studied in this thesis. After the capping treatment has been applied, trees are planted at the site, generally seedlings about one year in age (Fung & Macyk, 2000; Government of Alberta, 2018). Fertilizers and cover crops may be applied, to stabilize the soils and aid in the establishment of vegetation (Fung & Macyk, 2000).

There are a few different overburden formations in the Athabasca Oil Sands, and this study examined reclamation of Clearwater overburden sites. Clearwater overburden is a particular overburden formation found in the Athabasca Oil Sands, the remnant of an ancient inland sea (Fung & Macyk, 2000; Kelln at al., 2009). Covering this saline material with a capping treatment is meant to protect against salt ingress (Kelln et al., 2009; Macdonald et al., 2015a). Although naturally saline sites do exist in the boreal forest, saline soils have been shown to negatively impact tree growth (Lilles et al., 2010). The capping treatments applied in this case, are additionally utilized to minimize salt ingress from the underlying overburden (Kessler et al., 2010). Various capping depths and compositions have been trialed for this purpose, and will be discussed further in this thesis. Two of the main types of capping treatment are called peat-mineral mix (PMM) and forest floor material (FFM). PMM treatments are composed of an upper layer of peat and a lower mineral layer, both of which are sourced from more recently disturbed mine sites (Macdonald et al., 2015a; Macdonald, Quideau, & Landhäusser, 2012). FFM layers also have a lower mineral layer, but the upper layer is composed of forest floor LFH material, sourced from nearby upland forests before they are mined (Macdonald et al., 2015a). The mineral layer is sourced from lower horizon material, and is composed of a variety of materials and textures (Macdonald, Quideau, & Landhäusser, 2012). It is important to note that the peat is sourced from lowland sites, while the FFM is sourced from upland sites, but both are used for reclamation in new upland sites (Macdonald et al., 2015a). This study surveyed PMM reclamation sites.

1.2 Reclamation Targets for Clearwater Overburden and Measuring Success

The goal for reclamation of these upland sites is to target a type d ecosite (per Beckingham & Archibald, 1996) (M. Yarmuch, personal communication). Ecosites are categorized based on the nutrient and moisture regime of the soil; for example, a type d ecosite has a mesic moisture and medium nutrient regime (Beckingham & Archibald, 1996). Typical soils for this ecosite are as follows, taken from *Field Guide to the Ecosites of Northern Alberta* (Beckingham & Archibald, 1996). The A horizon (5-15 cm thick) is composed of Ae(gj), possibly with a layer of Ahe above. Ae(gj) is an eluviated type, and Ahe is similar, but also enriched with organic matter (Soil Classification Working Group, 1998). The B horizon (25-55 cm thick) is composed of Bt(gi) and in some cases also Bm. Bt(gj) is a layer enriched with illuviated clay (Soil Classification Working Group, 1998). Bm has been subject to chemical weathering, which means that carbonates have been removed, but there is likely no illuviation in this layer (Soil

Classification Working Group, 1998). The C horizon (30+ cm) is type C(kg), which means that it contains calcium carbonate from the parent material (Soil Classification Working Group, 1998). An LFH layer is usually present, roughly 2-10 cm thick for this soil profile. Parent material for type d ecosites are either glacial till or glaciolacustrine materials with moderately fine to fine texture (Beckingham & Archibald, 1996). Essentially, this means the typical soils experience a downward movement of nutrients and minerals, and lower horizons tend to be slightly basic.

The typical overstory species for this ecosite type are trembling aspen (*Populus tremuloides* Michx.) and white spruce (*Picea glauca* (Moench) Voss) (Beckingham & Archibald, 1996). Aspen is a fast-growing species which is often among the first trees to colonize sites and become dominant early after disturbance (Mitton & Grant, 1996; Peterson & Peterson, 1992; Bergeron et al., 2014). White spruce demonstrates a more conservative growth strategy; it can colonize sites at multiple time periods following disturbance, and it often only becomes dominant at the mid-successional stage or later during stand development (Gärtner, Lieffers, & Macdonald, 2011; Purdy, Dale, & Macdonald, 2002; Bergeron et al., 2014). In fact, white spruce often grows underneath an aspen canopy during the early and mid-stages of stand development (Lieffers & Stadt, 1994). In a study of naturally-saline sites, aspen shows a greater negative growth response to soil salinity than white spruce, with the response increasing with greater salinity (Lilles et al., 2010). While white spruce appeared to be less responsive to variation in salinity, growth for both species declines over time, as compared to non-saline sites (Lilles et al., 2010).

To initiate stand development for d ecosites during reclamation, aspen and white spruce are often planted in roughly equal proportions, at a combined density of about 2000 stems/ha. Historically, sites were planted in rows, in some cases alternating rows of aspen and white spruce, and in others alternative blocks of aspen or white spruce. Planting stock is typically one-year-old seedlings, and the major oil sands companies of the region collaborate on seed collection and stock growing (Government of Alberta, 2018; Schoonmaker et al., 2014).

Measuring the success of forest reclamation is limited by the intangible nature of defining successful reclamation. The *Land Capability Classification System for Forest Ecosystems in the Oil Sands* (2006) is used to determine the potential of the reclaimed soils to support forest vegetation. This system ranges from Class 1 High Capability to Class 5 Non-Productive, with classes 1-3 considered capable of supporting productive forests (Alberta Environment and Parks, 2006). Classes are determined by the soil nutrient and moisture regimes (Alberta Environment and Parks, 2006), much like the natural ecosites (Beckingham & Archibald, 1996). Site index at breast height age 50 (SI) is a useful tool to compare site potential of reclaimed areas to the natural counterparts the SI is based upon. Typical SI values for aspen and white spruce in type d ecosites are 18.2 ± 0.2 m and 16.8 ± 0.2 m, respectively (Beckingham & Archibald, 1996). This means that a 50-year old stand of aspen would have a mean height of 18.2 ± 0.2 m, and a 50-year old stand of white spruce would have a mean height of 16.8 ± 0.2 m. Using tree age and height, SI can be calculated for reclaimed sites, and compared to these typical values. Previous research on saline sites has shown that, compared with typical non-saline boreal sites, SI values for aspen and white spruce decrease with time, indicating salinity fosters a long-term decline (Lilles et al., 2012).

In addition to the reestablishment of trees, fostering a healthy understory community is an important part of forest reclamation. Understory vegetation contributes greatly to the biodiversity of boreal forests, contributes to soil stabilization, particularly for sites whose trees have not yet established themselves, and also provide microsites for new tree growth (Macdonald et al., 2015a). In contrast to the establishment of trees on reclamation sites, native understory vegetation is not often actively seeded or planted in the AOSR, but is expected to passively colonize newly reclaimed sites (Macdonald et al., 2015a). In the case of FFM capping treatments, many native upland seeds are present in the forest floor material, and will be able to grow on the newly reclaimed sites (Macdonald et al. 2015a; Mackenzie & Naeth, 2010). This is of course, provided that this material was not stockpiled for a lengthy period of time before placement, as stockpiling decreases the number of viable seeds able to germinate on a reclamation site (Dhar, Comeau, & Vassov, 2019; Mackenzie, 2013). With PMM treatments however, native lowland seeds would be present when the material has a limited storage time between soil salvage and reclamation; these species are less suited to the reclaimed upland sites (Hahn & Quideau, 2013). Native understory species must therefore arrive through dispersal from neighbouring natural or maturing reclaimed sites, if any exist in close proximity (Macdonald et al., 2015a). In the absence of native species, it is likely that weedy and invasive species will colonize reclamation sites and then may persist for several years (Audet, Pinno, & Thiffault, 2014; Errington & Pinno, 2015). Salinity can additionally impact the understory species composition, directing it away from typical upland forest communities (Purdy, Macdonald, & Lieffers, 2005).

Measuring reclamation success of an understory vegetation community is more challenging than tree growth, and tends to rely on community composition comparisons between reclaimed sites and natural sites, or even natural sites with recent natural disturbance (Errington & Pinno, 2015; Rowland et al., 2009). The reason for comparison to natural sites and natural disturbances is because of the expectation for community succession to be similar to natural forest understory succession (Errington & Pinno, 2015; Macdonald et al., 2015a). After reclamation, these communities are usually expected to increasingly resemble their pre-disturbance condition upon canopy closure (Macdonald et al., 2015a). One of the more tangible qualifiers for successful return to a more natural, pre-disturbance site is the disappearance of the weedy and invasive understory species that often colonize disturbed sites, and reappearance of native forest species (Macdonald et al., 2015b; Macdonald et al., 2015a; Mackenzie & Naeth, 2010; Brown & Naeth, 2014).

Previous research has studied the hydrology of various reclamation sites, addressing the integration into the wider landscape, and some of which also assess the relevance to tree and vegetation growth (e.g. Carrera-Hernández et al., 2012; Chapman, Barbour, & O'Kane, 2006; Strilesky, 2019; Meier & Barbour, 2002). The majority of research into the growth of trees and the development of an understory community on reclamation sites in the Athabasca Oil Sands has occurred during the first decade after planting, often just examining seedlings (e.g. Hoffman, 2017; Pinno & Errington, 2015; Onwuchekwa et al., 2014; Sloan & Jacobs, 2013; Landhäusser at al., 2012; etc.). Little research has been dedicated to the growth of trees on sites older than this, in part due to the comparatively lower abundance of long-term reclamation sites, which is a concern. Trees grow over many decades, and the interim between the planting of a reclamation site and reclamation certification can last over 25 years (Government of Alberta, 2013; Brocke & Ferster, 2007). To date, only one site has achieved this certification (Atkinson, 2017) to stand as a representative of successful reclamation in the long-term.

Of the few long-term studies conducted, it is made clear that early-stage results bear some difference to the later-stages of post-reclamation examined. For instance, Rowland et al. (2009) identified understory trends and responses to fertilizer that required 15-20 years to be realized, and Pinno and Hawkes (2015) noted changes in vegetation community and soil nutrition that took place over 20 years. Older sites are also considered to better-support comparisons with natural systems. For example, Farden et al. (2013) chose to use 21-year-old pine to evaluate reclamation success, with the expectation that pine in natural stands at this age already indicate any long-term trends. Furthermore, studies of soil salinity indicate that effects may only be seen in the long-term (Lilles et al., 2012). For these reasons, to accurately measure reclamation success, long-term study of the growth of planted trees, and the resultant understory development, are critically important.

1.3 Long-term Forest Response to the Peat-Mineral Mix Capping Treatment

The research sites studied for this thesis were located within Syncrude Canada Ltd.'s Mildred Lake oil sands lease, roughly 40 km north of Fort McMurray, in northeastern Alberta, Canada. The main site included in this study was a capping trial research site, which was developed to test the efficacy of three different depths of PMM capping treatments. This site was planted with aspen and white spruce in 1999, and regular surveys of tree growth and understory composition were conducted, including the present survey 19 years post-reclamation in 2018. This site was previously studied for the purpose of hydrological development after reclamation (Chapman, Barbour, & O'Kane, 2006; Meier & Barbour, 2002), to compare soil salinity of the three treatments (Kessler et al., 2010), and to investigate rooting behaviour of the planted trees (Lazorko & Van Rees, 2012). A detailed assessment of tree growth over time versus capping depth has not yet been completed.

The lack of replication for this long-term research site necessitated the addition of two age-based cohorts of sites with similar capping treatments, to better-compare treatment results. These cohorts comprised an additional eight sites, planted with aspen and white spruce between 2005 and 2010, and were 8-13 years post-reclamation when sampled in 2018.

The goals of this thesis were to investigate the growth responses of aspen and white spruce on a variety of PMM capping treatments, document and compare the understory vegetation communities present, and examine the changes in tree growth and understory community at the long-term research site. Understanding the responses of the forest overstory and understory to the capping treatments is critical to understanding the relative success of reclamation relative to targets. In particular, long-term research sites, such as the one included in this research, provide valuable information about reclamation success over time, as the constructed forest grows and changes. Assessing reclamation sites at one moment in time, or in the first few years after their establishment, will not provide for sufficient understanding of the dynamics of a reclaimed forest system, and the true efficacy of a capping treatment over the life-span of a forest. The information contained in this thesis can be used to indicate where reclamation techniques have succeeded in promoting healthy forest growth, and where some improvements might be needed.

In Chapter 2 I examine the growth of white spruce and aspen on four different thicknesses of PMM capping treatments. I expected to find an optimal capping depth for aspen, but a lack of preference in white spruce. I compared growth over time at the long-term research site, and among sites with particular capping treatments and comparable reclamation ages.

In Chapter 3 I investigate the development of forest understory communities at PMM reclamation sites. I expected to find a great variety in the communities developing at these sites, but a temporal trajectory at the long-term research site. I compared vegetation cover percentages for the species found at each site, and noted changes in composition over time.

Chapter 2: Tree growth 2-19 years post reclamation on Clearwater overburden

2.1 Introduction

Re-establishing forests after industrial disturbance is an important component of reclamation, but presents many challenges. In particular, successful tree growth and survival requires creating suitable site conditions (Macdonald, Quideau, & Landhäusser, 2012). Soil moisture and nutrients are strong determinants of site suitability (Alberta Environment and Parks, 2006), and meeting these requirements is the main role of capping treatments, the surface soil material placed upon a reclaimed site (Macdonald, Quideau, & Landhäusser, 2012; Li & Fung, 1998).

The Athabasca Oil Sands Region (AOSR) is a current and future source of industrial disturbance requiring reclamation. The minable oil sands area covers roughly 475 000 ha, much of which will need to be reclaimed to an equivalent capability boreal forest (Rooney, Bayley, & Schindler, 2012; Government of Alberta, 2005; Alberta Environment and Parks, 2010). There are a variety of materials and sites that require reclamation, each with their own challenges. Clearwater overburden, the focus of this study, is a saline material excavated during the mining process (Fung & Macyk, 2000). Salinity in high concentration can stunt tree growth, and decreases survival (Lilles et al., 2012). Thus, in the case of Clearwater overburden reclamation, a capping treatment is used to minimize salt ingress into the rooting zone; a thicker layer of capping is presumed to allow for greater depth of salt-free soil (Kessler et al., 2010). However, thicker capping treatments require more materials, and are thus more expensive; balancing cost and effectiveness are important for the practical applicability of reclamation prescriptions.

Materials used for capping are sourced from areas newly disturbed by mining operations; thus there are limits on total available capping materials. Typically, Clearwater overburden capping treatments contain two main layers, a nutrient-rich upper layer, and a mineral layer (Rowland et al., 2009; Turcotte, Quideau, & Oh, 2009). The upper layer can be composed of peat from excavated peatlands (known as peat-mineral mix, or PMM soils), or forest floor material harvested from upland forests (known as FFM soils) (Macdonald et al., 2015a). Peat is notable for its high water-holding capacity and low nutrient content (Béasse, Quideau, & Oh, 2015; van Breemen, 1995), whereas forest floor material has higher nutrient content, and can provide an upland seed bank (Mackenzie & Naeth 2010; Mackenzie & Quideau, 2012). The mineral layer material is sourced from the lower horizons of these same areas, and can contain a variety of materials and textures (Macdonald, Quideau, & Landhäusser, 2012). As the majority of the minable oil sands area is peatland (Raine, Mackenzie, & Gilchrist, 2002), there is generally greater availability of peat and thus a higher prevalence of use of PMM as a capping material for reclamation. However, it is important to note that the majority of reclamation on Clearwater overburden targets establishment of upland forest (Rooney, Bayley, & Schindler, 2012; Rowland et al., 2009).

Assessing the success of reclamation is complex, but comparisons to natural reference systems are useful (Government of Alberta, 2013). For example, standardized growth curves, such as those for site index (SI; tree height at breast height age 50 years) can be used to provide a concrete comparison of outcomes between reclamation sites and the natural sites they are intended to emulate (Huang et al., 2014). Land capability classes are a tool for predicting the success of reclamation based on the nutrient and moisture regimes of the capping treatment (Alberta Environment and Parks, 2006), and could be used to provide insight into why a site may or may not have successful growth of planted trees.

Forest reclamation is an evolving science, with many new research projects being developed to increase our understanding of reclamation best practices, such as capping treatments (Macdonald et al., 2015a; Rowland et al., 2009). However, as trees and forests grow and develop over many decades, and sites may be let to grow and establish for over 25 years before reclamation certification is applied for, it is important to document longer-term outcomes of capping treatments as well (Government of Alberta, 2013; Brocke & Ferster, 2007). In addition, effects of salinity on tree growth and survival can take decades to fully manifest (Lilles et al., 2012). Thus, in qualifying the success of reclamation, it is important to understand how the trees on a given site respond over a long period of time. Despite a variety of previous studies of forest reclamation in the AOSR, relatively few have followed tree growth beyond the first few years after tree planting (e.g. Pinno & Errington, 2015; Onwuchekwa et al., 2014; Sloan & Jacobs, 2013; Landhäusser at al., 2012; but see Farden et al., 2013). It is, therefore, important to incorporate longer-term studies of reclamation sites into the broader context of studies determining success of reclamation sites and prescriptions.

The main objectives of this study were to examine a long-term forest reclamation site, assessing the treatment response over nearly two decades, and to compare a suite of sites with similar capping treatments, at comparable ages of 9-13 years post-reclamation. Each of these upland sites were comprised of landforms underlain with saline Clearwater shale overburden that had been capped with similar PMM soils and planted with roughly equal densities of white spruce (*Picea glauca* (Moench) Voss) and trembling aspen (*Populus tremuloides* Michx.); thus, I focused on the growth of these two species. The 19-year-old site, known as the South Bison

Hills Instrumented Watershed (SBH) contains three different thicknesses of capping treatments, 35 cm, 50 cm, and 100 cm. I predicted (1) that aspen, being sensitive to increasing salinity (Lilles et al., 2012), would have the best growth in the thickest treatment and poorer growth in the thinner treatments. In contrast, I predicted that white spruce would demonstrate no differences among capping depths, being better-able to avoid soil salinity than aspen (Lilles et al., 2012). For the 13-year-old sites, which all had the same 100 cm capping treatment, I predicted (2) similar growth outcomes to each other. Finally, for the 9-year-old sites, where both the 100 cm treatment and a thicker 150 cm treatment were used, I predicted (3) that the 150 cm capping depth would provide no added growth benefits for aspen or white spruce, given previous estimates that 75 cm of capping for Clearwater overburden is sufficient to meet nutrient and water requirements (Syncrude Canada Ltd., 2013). The results of this research can establish the longer-term success of different capping treatments, and aid in determining the value of investment in increasing capping thicknesses. This can be used to specify which capping treatment best makes use of available resources, while supporting adequate white spruce and aspen reclamation on Clearwater overburden.

2.2 Methods

2.2-1 Research Area and Sites

Research was conducted at reclamation sites in the Mildred Lake Mine Area (57°02'29"N, 111°36'34"W), operated by Syncrude Canada Ltd. This oil sands mining operation is located within the Athabasca Oil Sands Region (AOSR), roughly 40 km north of Fort McMurray, in northeastern Alberta, Canada. This area has a mean annual precipitation of 418.6 mm, and a mean annual air temperature of 1.0 °C (Government of Canada, 2018).

The research area is situated within the Central Mixedwood natural subregion of the Boreal Forest (Natural Regions Committee, 2006). This subregion, which covers approximately one quarter of the province of Alberta, is composed of roughly 60% upland forest and 40% wetland and lowland forest areas (Natural Regions Committee, 2006). The upland areas are commonly dominated by forests in which the canopy is comprised mainly of trembling aspen (*Populus tremuloides* Michx.), white spruce (*Picea glauca* (Moench) Voss), and in certain areas, jack pine (*Pinus banksiana* Lamb.) (Natural Regions Committee, 2006). Typical soils are grey luvisols (Natural Regions Committee, 2006).

This study focused on Clearwater overburden reclamation. For each of the sites included in this research, Clearwater overburden forms the topographical structure of the site, and a capping treatment lies overtop, providing a rooting media for vegetation. There are four capping treatments included in this study, and each consists of different thicknesses of two layers: an upper, organic-rich peat layer composed of a peat/mineral mix, and a lower, clay-rich mineral layer (Figure 2.1). Treatment C35, the thinnest treatment, has 35 cm of capping in total: 15 cm of peat layer over 20 cm of mineral layer. Treatment C50 is 50 cm thick, with 20 cm of peat layer over 30 cm of mineral layer. Treatment C100 is 100 cm thick, with 20 cm of peat layer over 80 cm of mineral layer. The thickest treatment is C150 with a total thickness of 150 cm, is composed of 30 cm of peat layer over 120 cm of mineral layer.

After the landform was created using the overburden material and the capping treatment was applied over the overburden, native trees were planted onto the sites. Reclamation outcomes for the research sites targeted a type d ecosite (M. Yarmuch, personal communication), which is a common forest type in the Central Mixedwood subregion, and is dominated by aspen and/or white spruce (Beckingham & Archibald, 1996). These sites were planted with roughly equal densities of aspen and white spruce. My research focused on white spruce and aspen growth at three different subgroups of sites, whose specific establishment conditions are described below. The main site was a research site established with three capping treatments, and sampled at various points in time over the 19 years since its reclamation. Additional sites each had only one capping treatment (C100 or C150), and were sampled at 13 years post-reclamation, or at nine years post-reclamation. Specific establishment conditions are described below (see also Table 2.1).

2.2-1.1 South Bison Hills Instrumented Watershed

The main study site is located at the SW30 Overburden Research Site, also known as the South Bison Hills Instrumented Watershed (SBH). Here, Clearwater overburden was used to construct a hillslope with a 5:1 slope (Kelln et al., 2009). In 1999, three capping treatments were established over the overburden, C35, C50, and C100 (Kelln et al., 2009; Syncrude Canada Ltd., 2000). Each treatment covers about 0.8-0.9 ha of the site (Figure 2.2). This combination of capping treatments was established as a capping trial, and was not replicated elsewhere on Syncrude reclamation sites. In the fall of 1999, each capping treatment was planted with alternating rows of white spruce and trembling aspen. A total of 1600 stems/ha were planted (Syncrude Canada Ltd., 2000). At the same time, this site was seeded with annual barley (*Hordeum vulgare* L.) and fertilized (Syncrude Canada Ltd., 2000). See Table 2.1 for a summary of establishment conditions. In 2018, this site was 19 years post reclamation planting.

2.2-1.2 13-Year Cohort and Site Selection

The 13-year cohort is made up of three sites that were 13 years post-planting in 2018. Data from these sites was complemented by pre-existing data from SBH collected 13 years post-planting in 2012. The three 13-year-old sites were also reclaimed Clearwater overburden, capped in 2003, and planted in 2005 (Syncrude Canada Ltd., 2004; Syncrude Canada Ltd., 2006). Each of these sites received the C100 treatment, which was the standard capping depth required on Syncrude Canada Ltd. sites at the time (Syncrude Canada Ltd., 2013).

Selection criteria focused on finding sites with establishment conditions as similar to SBH as was reasonable. Primarily, each site had similar tree planting densities of aspen and white spruce to SBH (800 +/- 200 stems/ha/species). Only sites that were at least 1 ha in size were considered; in the end, selected sites ranged from 3.3 ha to 97.8 ha. When selecting a sampling location within each site, areas with a noticeable slope were preferred, to be similar with SBH; however, none of the sites had as strong a slope as SBH. Compatibility of sites with the selection criteria was determined using the 2016 Syncrude Conservation and Reclamation geodatabase (Government of Alberta, 2018), viewed with ArcGIS 10.2.1 (ESRI, 2014), and confirmed with site visits in July 2018. Sites were delineated according to the reclamation ID assigned to each site (Government of Alberta, 2018). The selected sites will be referred to as W2, S5A, and S5B (see Figure 2.3 for a map of sites; please refer to Appendix A for a list of alternate site aliases). Table 2.1 summarizes the establishment conditions of each site in the 13-year cohort.

2.2-1.3 9-Year Cohort and Site Selection

The 9-year cohort is made up of five sites that were nine years post-planting in 2018. These were complemented with pre-existing data from SBH collected nine years post-planting in 2008. These five sites were also reclaimed Clearwater overburden, two capped with the C100 treatment, and three capped with the C150 treatment. In 2007, C150 replaced C100 as the standard capping depth required on Syncrude Canada Ltd. sites (Alberta Environment and Parks, 2007; Syncrude Canada Ltd., 2013). As a result, the two C100 treatment sites had their capping implemented before this change, one in 2005 and one in 2006 (Syncrude Canada Ltd., 2006; Syncrude Canada Ltd., 2007). Given the change in regulation, these were the only two sites which otherwise met the selection criteria for C100 sites.

Selection criteria again focused on finding sites with establishment conditions as similar to SBH as was reasonable. Each site must have had similar tree planting densities of aspen and white spruce to SBH (800 +/- 200 stems/ha/species). Only sites that were at least 1 ha in size were considered, and in the end, selected sites ranged from 1.7 ha to 23.0 ha. When selecting a sampling location within each site, areas with a noticeable slope were preferred, to be similar with SBH; however, none of the sites had as strong a slope as SBH, due to regulation requirements (Alberta Environment and Parks, 2007). Compatibility of sites with the selection criteria was established using the 2016 Syncrude Conservation and Reclamation geodatabase (Government of Alberta, 2018), viewed with ArcGIS 10.2.1 (ESRI, 2014), and confirmed with site visits in July 2018. Sites were delineated according to the reclamation ID assigned to each site (Government of Alberta, 2018). The selected sites will be referred to as W1, 575A, 575B, 574, and 570 (see Figure 2.3 for a map of sites; please refer to Appendix A for a list of alternate site aliases). Table 2.1 summarizes the establishment conditions of each site in the 9-year cohort.

2.2-2 Historic Data Collection at SBH

Prior to this study, data were collected on tree height and diameter in 2001, 2002, 2003, 2005, 2008, and 2012 at the SBH site. Depending on the age of the trees, diameter was measured either as root collar diameter (2001-2008), which is diameter measured at ground level, or diameter at 1.3 m (breast) height (dbh, 2008-2012). Height was measured using a tree pole and yardstick, and dbh was measured using calipers or a diameter tape (d-tape); a yardstick and calipers are more practical for younger, smaller trees, whereas a tree height pole and d-tape are more useful as the trees become larger.

Up until 2005, the trees were sampled in plots. From 2005 to 2012, trees were sampled in transects. In 2005, trees were tagged and their GPS locations recorded, allowing the same trees to be re-measured in 2008 and 2012 (and in 2018). In 2008, 640 aspen and 706 spruce were measured at SBH, and in 2012, it was 623 aspen and 700 spruce.

2.2-3 2018 Data Collection

Height and dbh were assessed for 108 aspen and 108 white spruce at each site (and each treatment at SBH) for a total of 216 trees per site (648 trees at SBH). Data were collected in July and August of 2018.

At SBH we measured roughly every other tree (from those tagged in 2005, and in an established transect line) along four transects per treatment. The area within which the tagged trees were selected varied from treatment to treatment, but was roughly 20-30 m by 150 m (treatment plot dimensions are roughly 50 m by 150 m). If a tree was dead, the next tree was measured instead. If this tree was also dead, the previous tree to the first dead tree was measured. Tag numbers of dead trees were recorded.

At the new sites, plots of 30 m by 70 m were established, and each live tree inside the plot was measured, until the total of 216 trees was reached (divided amongst four transects; see Figure 2.4). Plot dimensions were selected based on the assumption that a planting density of ~2000 stems/ha would equate to roughly 420 trees planted within the plot, about double the sampling target. However, if less than 216 trees were found within the plot, trees were measured within a 5 m buffer of the plot, and then a 10 m buffer if necessary, until 216 trees had been measured. Trees shorter than 1.5 m in height were not included, as they were considered too short for an accurate measure of dbh. Numbers of dead and too-short trees encountered were recorded.

Tree height was collected using either a tree pole (for heights under 4 m) or a vertex hypsometer (for heights above 4 m). Dbh was collected using a d-tape for all trees.

2.2-3.1 Soil Sampling

To verify the accuracy of capping treatment implementation, and establish the actual depths of the peat and minerals layers, soil cores were taken at each site, between August and October 2018. Three cores were taken at each site, and their depths averaged to create a mean value for that site. One core was taken at the centre of each site, the second approximately 21 m north of the first (~50 m north at SBH) and the third approximately 21 m south of the first (~50 m south at SBH). From the core the depths of the LFH (litter, fermented, and humic material), peat, and mineral depths were measured. Layers were delineated both visually and by texture. The LFH was denoted by its litter and humic content. The peat layer was distinguished by the noticeable peat content and lesser density relative to the mineral layer. The mineral layer, in turn, was identifiably light brown in colour with hints of pink, and was generally rich in clay. Finally, the overburden was a markedly black and dense material. Coring stopped once the overburden was reached.

2.2-4 Data Analysis

The R Software Environment (R) version 3.5.1 (R Core Team, 2018) was used for statistical analyses and graphical data representation. ArcGIS 10.2.1 (ESRI, 2014) was used to create sitemaps and visualize GPS data. All analyses were done separately for the two tree species.

Historic and 2018 data from SBH were used to create both height and diameter growth trajectories for aspen and white spruce at SBH. This addressed prediction 1, by comparing the species' growth over time for the three treatments present at SBH: C35, C50, and C100. The 2018 data, in particular, was used to compare mean height and diameter values between treatments, to identify if any strong treatment differences can be identified, between the C35, C50, and C100 treatments. Since there was no true replication of the three treatments at SBH no statistical analyses were completed. If means were noticeably different, the lower confidence interval of the better-performing treatment was compared with the mean of the more poorly performing treatment; this established a conservative estimate of the growth differences between the treatments.

To evaluate prediction 2, descriptive statistics derived from the data for the 13-year cohort were used to examine variation in mean height and diameter 13 years post planting among sites for the C100 treatment. This provided information on the range of variation of performance for this treatment and allowed us to determine if growth at SBH was comparable to other sites receiving that treatment, at a similar age (i.e., the sites planted in 2005).

The 9-year cohort of sites allows for comparison between the C100 and C150 treatments for prediction 3. In this case there were replicate sites for each treatment; I used a mixed model ANOVA with random effects to compare height and diameter between the C100 and C150 treatments. This was done in R with the *lmer* function from the *lmerTest* package. Capping treatment was the fixed effect, while site was included as a random effect, to account for the lack of independence among individual trees at each site. The test residuals were used to confirm normality and homogeneity of variance. In the case of aspen height and dbh, and spruce dbh, these assumptions were not met, and the dependent variable was subjected to a square root transformation for aspen, and a log transformation for white spruce.

2.2-4.1 Site Index Calculation

The height data collected in 2018 were used as the basis of a calculation of Site Index (SI) for each site, in order to compare an age-independent site potential among them. SI was calculated using the method described by Huang et al. (2009) in *A growth and yield projection system (GYPSY) for natural and post-harvest stands in Alberta*. Code provided in Appendix 1 was used to create a spreadsheet with the ability to calculate SI at base height age 50 years, given tree species, height, and age. This spreadsheet was provided by Syncrude Canada Ltd. All trees were planted at one year of age, and tree age (as of end of the 2018 growing season) has been adjusted accordingly. Site Index could not be calculated for white spruce for site W1 because the minimum age for white spruce calculation in 9.5 years. For all other sites and for aspen at W1, data for each tree measured in 2018 was input into the spreadsheet, and SI output was averaged for each site. This mean SI value for each site was then compared against natural site class valuations, determined according to the normal variation found in the Central Mixedwood natural region (per Huang, Titus, & Lakusta, 1994). For aspen, SI values >18 are considered good, >14 to ≤18 are considered medium, >10 to ≤14 are considered poor, and ≤10 are considered unproductive. For white spruce, SI values >15.5 are considered good, >10.5 to ≤15.5 are considered medium, >6.0 to ≤10.5 are considered poor, and ≤6.0 are considered unproductive. In addition, SI values were compared with type d ecosite mean values (per

Beckingham & Archibald, 1996). Mean aspen SI for a type d ecosite is 18.2 ± 0.2, and 16.8 ± 0.2 for white spruce.

2.2-4.2 Site Index Model Selection

Site SI means were then used to determine if there was a relationship between capping treatment and site potential; specifically, to test whether the expected peat, mineral, and total capping depths were correlated with the mean site SI. Linear and multiple linear models were tested, and I compared the Akaike Information Criterion (AIC) values for each model. For these models, *cor* and *cor.test* were also applied to test the strength of any relationship. An examination of the test residuals was used to check that assumptions of normality were met, and to confirm linearity. For each species, the model with the lowest AIC value was chosen as the best model.

2.3 Results

Trembling aspen and white spruce displayed different and variable responses to treatments and site conditions. This is apparent based on the data collected after 19 years of growth at South Bison Hills Instrumented Watershed (SBH), in the 13-year cohort, and in the 9-year cohort. Site-to-site variability was high, and may indicate that capping treatment is not the main or only factor influencing tree growth at these reclamation sites.

2.3-1 South Bison Hills Instrumented Watershed

As per my first prediction, I found that aspen and white spruce responded differently to the treatments at SBH. After 19 years of growth, aspen showed marked differences in both height and dbh between the C35, C50, and C100 treatments (Figures 2.5 and 2.6, respectively). In contrast, white spruce did not show strong differences between treatments (Figures 2.7 and 2.8).

Growth differences between treatments for aspen became clear by 2008 and continued to steadily increase in magnitude with age thereafter. These differences were quite prominent in 2012, where C100 aspen had a mean height of 5.9 m and dbh of 5.2 cm, while C50 aspen had a mean height of 5.2 m and dbh of 4.6 cm, and C35 aspen had a mean height of 5.0 m and dbh of 4.5 cm. By 2018, these differences were even greater, and the mean height of aspen in treatment C100 was 9.5 m, whereas C50 aspen had a mean height of 8.5 m, and C35 aspen had a mean height of 7.5 m. This meant that C100 aspen were 22% taller than C35 aspen and 8% taller than C50 aspen. C100 aspen also had a much greater diameter, with a mean of 8.7 cm versus the C50 mean of 7.6 cm and the C35 mean of 6.8 cm. C100 aspen thus have a 22% greater stem diameter than C35 aspen, and a 9% greater stem diameter than C50 aspen.

2.3-2 13-Year Cohort

Despite my second prediction that trees growing in the C100 treatment at different sites 13 years post-reclamation would have similar growth outcomes, I found that there was a fair amount of variation between sites, for both aspen and white spruce (Figures 2.5-2.8). Site S5B had the highest growth outcomes for both species, and for aspen in particular, the growth of these 13-year-old trees was comparable to the 19-year-old trees growing in treatments C35 and C50 at SBH (Figures 2.5-2.6). S5A had second greatest aspen growth, but had similar growth to

SBH treatment C100 for white spruce. SBH treatment C100 had the third greatest aspen growth. The site with the lowest growth outcomes was W2, where height and dbh for both aspen and white spruce fell below even treatment C35 at SBH. This poor performance was especially true for white spruce, where growth of these 13-year-old trees was comparable to the growth of 9-year-old trees (Figures 2.7-2.8).

2.3-3 9-Year Cohort

In support of my third prediction, neither white spruce (height: $p = 0.93150$, $N = 753$; dbh: $p = 0.5921$, $N = 735$) nor aspen (height: $p = 0.7924$, $N = 745$; dbh: $p = 0.5463$, $N = 743$) demonstrated significant differences in growth between the C100 and C150 capping treatments (Figure 2.9).

As with the 13-year cohort, differences among the different capping depth treatments at SBH were less dramatic than differences between all sites of the 9-year cohort, regardless of capping depth (Figures 2.5-2.8). It also appeared that trees on most of the new 9-year cohort had similar or greater growth than trees on SBH at the same age, especially in the case of white spruce (Figures 2.7-2.8). For white spruce, the best site for growth outcomes was 575A (C100), followed by 575B (C150), 574 (C150), 570 (C150), W1 (C100), and finally SBH (C100). For aspen, the trends were slightly different, with 570 (C150) having the best growth, followed by 575A (C100), 575B (C100), SBH (C100) ~ W1 (C100), and finally 574 (C150).

Overall, variability between sites was greater in the 13-year cohort than the 9-year cohort. The mean aspen height range between sites in the 9-year cohort was 1.3 m, and for the 13-year cohort this nearly tripled to 3.8 m. Similarly, for white spruce the mean height range for the 9-year cohort was 0.8 m, versus 2.1 m for the 13-year cohort. The mean aspen dbh range between sites in the 9-year cohort was 1.0 cm, but for the 13-year cohort this had more than tripled to 3.5 cm. For the mean spruce dbh, the 9-year cohort range was 1.4 cm, and 3.4 cm for the 13-year cohort. This indicates that growth differences due to site differences only increase as the trees age.

2.3-4 Soil Sampling and Site Index

At SBH, the three capping treatments were 4-9 cm thinner than prescribed (Figure 2.10). Notable, was the development of an LFH (litter, fermented, and humic material) layer on SBH, which is 19 years post-planting. The sites of the 13-year cohort tended to match the technical specifications for the C100 capping treatment fairly well but still ranged from 7 cm thinner to 11 cm thicker than prescribed. The sites of the 9-year cohort, however, differed substantially from both their specifications. For example, site 574 had 50 cm of extra capping depth, site 575A had 37 cm of extra depth, and site 575B was short 37 cm of capping depth.

Mean SI values for white spruce were considered medium to good according to site class values for Alberta (per Huang, Titus, & Lakusta, 1994), and with the exception of site W2, all site mean SI values were higher than expectations for a type d ecosite (per Beckingham & Archibald, 1996) (Table 2.2). Mean SI values for aspen were considered poor (in the case of 574) to good, and only three sites (SBH C100, S5A, and S5B) were equal to or higher than the expectations for a type d ecosite (Table 2.2). Interestingly, the poorest site for aspen was good for spruce, and the poorest site for white spruce (while still considered medium) was considered medium for aspen.

2.3-6 Site Index and Soil Depth Regression Analysis

According to Akaike's Information Criterion (AIC), the best model for aspen mean SI was the linear mixed model with mean SI as a function of peat depth, total capping depth, and their interaction (AIC = 49.88, Table 2.3). This relationship was not significant however ($p = 0.156$), but is nearly significant with regards to the contributions of the peat layer ($p = 0.074$). However, as all AIC values are relatively similar, it is possible that the differences in model fit are very slight.

For white spruce, the best model according to AIC value was a linear model with mean SI as a function of measured peat depth (AIC = 44.35, Table 2.4). However, this is not a significant correlation ($p = 0.387$). The best quadratic and simple logarithmic models were also for mean SI as a function of measured peat depth (quadratic: AIC = 45.55, simple logarithmic: AIC = 44.89). As with aspen, all AIC values are relatively similar, and it is again possible that the differences in model fit are very slight.

2.4 Discussion

This study demonstrated that, at South Bison Hills Instrumented Watershed (SBH), thicker capping depths are associated with improved trembling aspen growth. In combination with data from newer reclamation sites, the results suggest that capping depths greater than 1 m do not result in further improvement in growth, at least within the nine-year time period we were able to assess. The differences in aspen growth appeared at SBH around nine years post-planting, indicating it is necessary to wait nearly a decade to assess whether growth differences will exist, and likely longer to become confident in their veracity. In contrast, white spruce did not demonstrate different growth outcomes for the different treatment depths, growing consistently across all tested depths. However, it is possible that white spruce will simply take longer to display noticeable growth differences than aspen, due to their slower and more conservative growth.

Differences among sites seem to have important influences on growth, and may overwhelm effects of differing capping depth, necessitating further investigation into their impacts. It is also important to note that SBH, which was planted 19 years before the current sampling, tends to underperform compared to the newer sites at equivalent ages; this may be attributable to the improvements made to reclamation techniques in the years since SBH was established in 1999. However, it is important to note that, for almost all sites, and especially for white spruce, site index at base height age 50 years (SI) is considered normal to high, indicating that these sites are all generally performing well.

2.4-1 Long-term Growth at South Bison Hills

As predicted, aspen growth at SBW over 19 years post-reclamation was greatest on the thickest capping treatment (C100), but white spruce growth did not differ among capping treatments. One main purpose of the capping treatment is to protect against salt ingress from the Clearwater overburden material beneath, with thicker treatments shown to provide greater protection than thinner (Kelln et al., 2009). Previous research into the impacts of soil salinity upon aspen and white spruce have shown that aspen is more sensitive to salinity than white spruce; aspen will not grow in soil with salinity (at a 50-100cm depth) greater than 15 dS/m, yet white spruce will continue to grow in soil with salinity (at a 50-100cm depth) as high as 23 dS/m (Lilles et al., 2012). It is therefore conceivable that the differences in growth between treatments for aspen is due to the thinner treatment having higher salinity, which in turn negatively impacts the growth of the more sensitive aspen compared to white spruce. In addition, aspen growth declines more than expected over time on naturally saline sites (Lilles et al., 2012), which could explain the roughly 10-year delay in the appearance in treatment differences for aspen.

Current levels of salinity in surface soils were not explicitly assessed in this study, but a previous study of this site did examine the salinity of each capping treatment. Kessler et al. (2010) found that the lower 15-20 cm of each treatment had accumulated salts from the overburden, which is roughly half of the depth of treatment C35, and a third of treatment C50. Conductivity of these portions was as high as 6.0 dS/m, and salinity was significantly higher in treatments C35 and C50 than C100 (Kessler et al., 2010). Furthermore, Lazorko (2008) found that the majority of tree roots at SBH were contained in the top 30 cm of the capping treatment, which is consistent with findings from natural boreal forest sites (Strong & La Roi, 1983; Jackson et al., 1996). Taken together, these results indicate that trees growing in treatment C35 are unable to escape saline soils, trees in treatment C50 do have some available rooting area that is not saline, and trees in treatment C100 have ample rooting space without strong saline impacts. This allows us to infer that salinity contributes to the different growth outcomes for aspen and white spruce. Lilles et al. (2010) hypothesized that high availability of soil moisture and nutrient could help mitigate the effects of soil salinity in natural sites, so it is possible that these factors play a role at SBH as well.

The lack of response in white spruce could additionally be a result of the conservative growth strategy in that species; white spruce tends to grow more slowly than aspen (Tremblay, Thiffault, & Pinno, 2019; Xing et al., 2018; Man & Lieffers, 1999). Research on naturally saline sites found a decline in growth over time for white spruce, so it is, again, still possible that the current growth patterns will change with time (Lilles et al., 2012). Thus it is important to continue monitoring SBH.

While SBH is demonstrably valuable as a long-term research site for assessing reclamation outcomes, the lack of proper replication does create limitations for its usefulness. However, SBH was designed to keep many influential factors constant, factors which can vary at other sites from year to year. Thus while factors such as weather, capping material and sources, surrounding environment, and reclamation methods can change year to year and site to site, these are consistent at SBH. As a result, SBH is still useful as a demonstration of differences that can arise to different capping treatments, while holding constant many other influential variables.

2.4-2 Comparing SBH with new sites

In an effort to combat the lack of replication at SBH yet still utilize the wealth of data previously collected, I created the 13-year and 9-year cohorts, sampling additional sites that had the C100 treatment. The new sites were either 13 years post-planting, to complement previous data collected from SBH at that age (the 13-year cohort, three additional C100 sites), or 9-years post planting, to complement previous data collected from SBH at that age (the 9-year cohort, two C100 sites). I expected the new sites with the C100 treatment to display growth outcomes similar to SBH. I had based this expectation on the assumption that the C100 treatment would have the same impact on tree growth – by creating a buffer zone for salt ingress from the underlying Clearwater overburden. Thus if this is the main way in which the capping depth treatment influences growth, these sites should all be performing similarly. Although there was much site to site variation, I found that, in general, the newer C100 sites tended to have better growth than SBH. Or, at the very least, SBH is at the bottom end of average when compared to the 13-year and 9-year cohorts.

The fact that we found such large site-to-site variation suggests that other factors are having an important influence on growth outcomes. One of these factors could be the quality of the capping material used. Previous research has found that stockpiled materials retain less nutrient content than those directly placed, and thus direct placement is preferred (Dhar, Comeau, & Vassov, 2019; Mackenzie, 2012). This is anecdotally contradicted by my results; for example, three of the best-growing sites, 575A, S5A, and S5B all received stockpiled peat, yet the lowest outlier for growth performance, W2, received directly placed peat. However, it speaks to the influence that changes in capping source material could have upon growth outcomes. This is supported by previous research, which found that individual properties of the reclamation material may be more relevant than type of storage (Omari, Gupta, & Pinno, 2018). Another factor explaining high site-to-site variation in growth for the C100 treatment could be the effect of the quality of the planting stock and their genetics, which could vary from site to site and year to year. Unique weather in the year of reclamation or shortly thereafter could also have an impact; notably there was a period of severe drought in the early years (2001 – 2002) of growth at SBH (Wheaton et al., 2008). This could have stunted the growth of trees at SBH, as compared to the newer sites. Studies conducted on white spruce and aspen in natural forests during and after this drought indeed found a marked decline in growth (Hogg et al., 2017; Hogg, Brandt, & Michaelian, 2008). Regardless, the high site-to-site variability indicates that further study would benefit from the addition of more sites, with different planting years and conditions, to help characterize the site-to-site variation, and clarify the impacts of capping treatment.

Given the usefulness of SBH as a long-term reclamation site for comparison of capping depths, but lack of treatment replication, it was important to establish whether or not SBH is representative of reclamation on Clearwater overburden. First, it should be noted that according to site index, white spruce at SBH are exceeding expectations set for the target ecosite, as are aspen in treatment C100. However, as mentioned above, aspen and white spruce in the SBH C100 treatment have shown somewhat poorer growth at comparable ages to the 13 and 9-year cohorts. Severe drought early after establishment of SBH (Wheaton et al., 2008) could be to blame. Alternatively, one could also be optimistic that changes to reclamation technique since SBH was established in 1999 have resulted in improved site quality. Some of these changes include the fact that the newer sites were not fertilized during establishment, or planted with a cover crop of barley (Syncrude Canada Ltd., 2002; 2004; 2006; 2007; 2009; 2010; 2011).

With this is mind, it is possible that treatment C100 at SBH is not perfectly representative of the variety of C100 reclamation outcomes at all sites; however, this site does retain its value as a long-term study site of capping treatments. It may be that high site-to-site variability means that no one site can be representative, so a larger sample size could be useful.

2.4-2.1 C100 and C150

The 9-year cohort additionally consisted of three C150 sites, a capping treatment thicker than C100. Previous research conducted at SBH suggested that the optimal capping depth was 75 cm (COSIA, 2018; Syncrude Canada Ltd., 2013), therefore I predicted that increasing capping depth from 100 cm (C100) to 150 cm (C150) would not result in better tree growth. This prediction was supported by the data for the 9-year cohort, as there was no significant difference between these two capping treatments, for either white spruce or aspen. However, this result should be approached with caution. The high site-to-site variability mentioned above could be masking any treatment differences, and it is possible that growth differences will simply take longer to appear. As mentioned above, capping treatment differences for aspen at SBH did not start to appear until nine years after the site was planted, and these differences continued to develop for another 10 years to the 19 year post-reclamation sampling. It is conceivable that these trends for growth differences among capping treatment could keep developing as time goes on. As a result, continued study of these sites will be necessary to be able to confidently conclude there is no effect on tree growth of increasing capping thickness from 100 cm to 150 cm.

2.4-3 Growth Differences and the Success of Capping Treatment Implementation

The lack of differences in white spruce growth among capping treatments could be interpreted in one of two ways, either white spruce are all growing similarly well, or they are all growing similarly poor. Fortunately, it seems likely that the former is true, as the site indices were high overall. This is true for both species, when compared to the expectations for natural sites in Alberta (per Huang, Titus, & Lakuska, 1994). Previous research has also found SI values for reclamation sites to be comparable to natural forests (Huang et al., 2014). Results were a bit more mixed when compared with mean SI values for white spruce and aspen in type d ecosites (per Beckingham & Archibald, 1996), where more sites had site indices for white spruce or aspen that were lower than these means. One site in particular, W2, had SI values much lower than expected for a type d ecosite; as found in Chapter 3, this site can be classified as a type b ecosite, which has different expectations for tree growth. However, both aspen and white spruce still had lower SI values than expected for this ecosite (per Beckingham & Archibald, 1996).

In general however, the good SI values for the sites assessed speak to the success of reclamation techniques in establishing productive stands of aspen and white spruce. This could indicate that capping treatments provide ample nutrients and water, compared to natural soils. For example, peat is known to have a high water-holding capacity (Van Breemen, 1995), and in general upland forests do not have peat layers (Nichols, 1998; Beckingham & Archibald, 1996), compared to many of the reclamation sites in this study. This trend towards high SI values is particularly true for white spruce, so it is possible that whatever positive impacts of capping treatments exists relative to natural forests, they are better for white spruce.

A notable trend when comparing the two tree species is that sites with higher SI values for aspen are often not the sites with higher SI values for white spruce. In other words, what makes a site favourable for aspen is not necessarily what makes it favourable for white spruce. It is not unexpected for white spruce and aspen to have different needs for site conditions to support growth (Zhang et al., 2013; Lilles et al., 2012; Xing et al., 2018), and balancing these differences is a challenge within reclamation. As such, it could be useful to further study which factors promote growth for aspen versus white spruce on reclamation sites, and how to promote both species on the same site.

2.4-3.1 What Components of the capping treatment can explain growth differences?

The purpose of this study was not to produce explanatory results for the differences in capping treatment, but some inferences are possible. In 2009, the SBH sites were evaluated according to the land capability classification system for forest ecosystems in the oil sands (Alberta Environment and Parks, 2006; Syncrude Canada Inc., 2010). This system considers soil moisture and nutrient regime, while also taking factors like salinity into consideration (Alberta Environment and Parks, 2006). Each of the sites sampled were rated either class 2 (moderate capability) or class 3 (low capability), but these evaluations of capability to support forest species did not correspond well to SI values for either species. As a result it is unlikely that this particular assessment of soil quality will be useful in clarifying the performance of white spruce and aspen on a reclamation site.

Breaking down the capping treatments into the two main components, the peat layer and the mineral layer, was done in an attempt to determine if these two could be playing a greater role in tree growth individually rather than as a whole. The observed variability in depths, despite only four prescribed treatments, allowed for a variety of peat and mineral layer depths to be examined, and for these actual depths to be compared directly with aspen and white spruce SI. This study found no significant results, but it is possible that a study specifically designed to compare specific depths of peat and mineral layers to SI could yield different results. Previous research has shown that relationships between tree growth and depth of peat placement on reclamation sites exist, but the type of relationship differs according to different studies. A study of Jack pine (*Pinus banksiana* Lamb.) in the same reclamation area found that increasing peat capping depth had a positive linear relationship with tree height (Farden et al., 2013). Alternatively, it has also been shown that deep layers of peat retain cold temperatures during the growing season (Nichols, 1998), which can be detrimental to root growth (Wan et al., 1999). A study that includes a greater number of sites, and a greater variation in the depths of peat and mineral layers, could better quantify any relationship of capping treatment with white spruce and aspen growth.

It is worth noting that natural soil profiles are much more complex than any capping treatment in this study (Beckingham & Archibald, 1996), and there is no doubt that soil processes and properties in the constructed soils on reclamation sites differ from natural forests (Rowland et al., 2009). It is possible that increasing the complexity of capping treatments, and encouraging the natural soil processes, such as the delivery of nutrients and water, could also change the growth outcomes for white spruce and aspen. For instance, Kwak et al. (2015) found that the addition of coarse woody debris to reclamation sites increased microbial presence in the soil, which is expected to enhance nutrient cycling.

With all of this in mind, it is clear that there are still many unknowns when it comes to how trees respond to reclamation capping treatments. Further studies could explore in more detail how trees respond to the different components of capping treatment (i.e. varying thicknesses of peat, varying thicknesses of mineral) and complexity of these.

2.4-4 Conclusions

This study established the value of SBH as a long-term reclamation study site, easily enhanced by data collection from additional reclamation sites. Based on the data from this site and the younger sites included in this study, it is my recommendation to continue with improvements to site preparation, such as ending the broadcast fertilizing of young sites. I would also suggest that increasing capping from 100 cm to 150 cm is not necessary, but further time may be needed to confirm that the relevant growth trends do not change. Future research could continue to add sites to the dataset, and a detailed capping study, with specific depths of peat and clay layers could better investigate the contribution of these layers to growth outcomes.

Table 2.1 Summary of the reclamation procedures and site conditions for each of the study sites, including the capping treatment, peat material placement, tree planting density, and whether sites were additionally seeded with a cover crop, fertilized, or planted with shrubs. Source of information: Syncrude Canada Inc. annual reclamation reports (2000, 2004, 2006, 2007, 2009, 2010, and 2011) and the 2016 Syncrude Conservation and Reclamation geodatabase (Government of Alberta, 2018). Tree species codes are as follows: Aw = trembling aspen, Sw = white spruce, Pj = jack pine.

Site	Capping Treatment	Peat Placement	Seeding	Fertilizer	Shrubs (stems/ha)	Trees (stems/ha)
SBH 2.5 ha	1999	Direct	1999	1999	N/A	1999
	C100, C50, & C35		20 kg/ha annual barley	363 kg/ha 10N-30P-15K-4S		800 Aw: 800 Sw
W2 97.8 ha	2003	Direct	N/A	N/A	2005	2005
	C100				37.5 saskatoon, 205.5 green alder	1089.1 Aw: 1027.7 Sw
S5A 3.3 ha	2003	Stockpiled	2003	2003	N/A	2005
	C100		28.1 kg/ha annual barley	327.1 kg/ha 10N-30P-15K-4S		867.4 Aw: 1040.5 Sw
S5B 4.1 ha	2003	Stockpiled	2003	2003	N/A	2005
	C100		28.1 kg/ha annual barley	327.1 kg/ha 10N-30P-15K-4S		867.4 Aw: 1040.5 Sw
W1 16.9 ha	2005	Direct	N/A	N/A	2010	N/A
	C100				1097 Aw: 1134 Sw	

Site	Capping Treatment	Peat Placement	Seeding	Fertilizer	Shrubs (stems/ha)	Trees (stems/ha)
575A 1.7 ha	2006	Stockpiled	N/A	2006	2009	2009
	C100			302 kg/ha 10N-30P-15K-4S	950 Aw: 950 Sw	~230[*]
575B 23.0 ha	2008	Direct	N/A	N/A	2009	2009
	C150				950 Aw: 950 Sw	~230[*]
574 5.9 ha	2008	Direct	N/A	N/A	2009	2009
	C150				1199 Aw: 1200 Sw: 53 Pj	~230[*]
570 7.4 ha	2008	Direct placement	N/A	N/A	2009	2009
	C150				1199 Aw: 1200 Sw: 53 Pj	~230[*]

[*]It is unclear the exact amounts and proportions of shrubs planted, but there were an estimated 230 stems/ha for all 2009 shrub plantings, which included the following species: buffaloberry (*Shepherdia canadensis* (L.) Nutt.), saskatoon (*Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roem.), pin cherry (*Prunus pensylvanica* L. f.), green alder (*Alnus viridis* (Chaix) DC. ssp. *crispa* (Aiton) Turrill), dogwood (*Cornus sericea* L. ssp. *sericea*), blueberry (*Vaccinium myrtilloides* Michx.), dwarf birch (*Betula nana* L.), silverberry (*Elaeagnus commutata* Bernh. ex Rydb.), shrubby cinquefoil (*Dasiphora fruticosa* (L.) Rydb.), fly honeysuckle (*Lonicera villosa* (Michx.) Schult.).

Table 2.2 Summary of mean site index (SI as height at breast height age of 50 years) for each species and site in 2018. Site W1 could not be included in white spruce SI calculation, as the minimum age for white spruce calculation in 9.5 years. Mean SI for each species and site is given a site class, according to accepted SI values for Alberta (per Huang, Titus, & Lakusta, 1994). Site class ranges from unproductive, to poor, to medium, to good. In addition, these values were compared to mean SI values for type d ecosites (per Beckingham & Archibald, 1996). Sites with SI values higher than the mean for type d are indicated with a "+", and values lower than the mean for type d are indicated with a "-"; doubling these symbols indicates either the highest or lowest SI value, respectively, for each species in the study.

Site	Treatment	Tree Age (years)	Aspen SI (mean)	Site Class	Type d ecosite	Spruce SI (mean)	Site Class	Type d ecosite	2009 Soil Quality
SBH	C35	20	15.8	medium	-	19.5	good	+	Class 2
	C50		17.0	medium	-	19.6	good	+	
	C100		18.6	good	+	19.7	good	+	
W2	C100	14	14.1	medium	-	14.2	medium	--	Class 3
S5A	C100	14	18.7	good	+	19.5	good	+	Class 3
S5B	C100	14	21.0	good	++	20.6	good	+	Class 3
W1	C100	9	15.4	medium	-	N/A			Class 2 & 3
575A	C100	10	15.3	medium	-	20.7	good	++	Class 2
575B	C150	10	15.0	medium	-	19.2	good	+	Class 3
574	C150	10	13.8	poor	--	18.9	good	+	Class 2
570	C150	10	15.7	medium	-	18.7	good	+	Class 2

Table 2.3 Summary of the regression model selection results examining the relationship between trembling aspen site index (SI as height at breast height age of 50 years) and the observed depth of the peat and mineral capping layers, and of their combined depth. Three model types were tested: simple linear, multiple linear without interaction, and multiple linear with interaction. Akaike Information Criterion (AIC) values were used to compare the models, with the lowest value in bold. P-values for *cor.test* results are displayed, and additionally the p-value for the contribution of peat to the model with the best AIC value.

Test	AIC	multi R^2	P-value	Estimate (a)	Estimate (b)	Estimate (interaction)
linear						
mean SI ~ peat depth	53.07	0.048	0.519	0.049		
mean SI ~ mineral depth	51.12	0.203	0.165	-0.022		
mean SI ~ combined depth	51.87	0.146	0.245	-0.018		
multiple linear						
mean SI ~ peat depth + mineral depth	51.91	0.286	0.261	0.066	-0.024	
mean SI ~ mineral depth + combined depth	51.91	0.286	0.261	-0.090	0.066	
mean SI ~ peat depth + combined depth	51.91	0.286	0.261	0.090	-0.024	
mean SI ~ peat depth * mineral depth	50.84	0.460	0.205	0.349	0.034	-0.003
mean SI ~ mineral depth * combined depth	53.62	0.304	0.440	-0.058	0.060	-0.0001
mean SI ~ peat depth * combined depth	**49.88**	0.505	0.156 **(peat = 0.074)**	0.445	0.025	-0.003

Table 2.4 Summary of the regression model selection results examining the relationship between white spruce site index (SI as height at breast height age of 50 years) and the observed depth of the peat and mineral capping layers, and of their combined depth. Three model types were tested: simple linear, multiple linear without interaction, and multiple linear with interaction. Akaike Information Criterion (AIC) values were used to compare the models, with the lowest value in bold. P-values for *cor.test* results are displayed.

Test	AIC	multi R^2	P-value	Estimate (a)	Estimate (b)	Estimate (interaction)
linear						
mean SI ~ peat depth	**44.35**	0.095	0.387	0.059		
mean SI ~ mineral depth	45.13	0.021	0.688	-0.006		
mean SI ~ combined depth	45.28	0.006	0.827	-0.003		
multiple linear						
mean SI ~ peat depth + mineral depth	45.84	0.140	0.591	0.067	-0.008	
mean SI ~ mineral depth + combined depth	45.84	0.140	0.591	-0.076	0.067	
mean SI ~ peat depth + combined depth	45.84	0.140	0.591	0.076	-0.008	
mean SI ~ peat depth * mineral depth	47.81	0.142	0.804	0.043	-0.014	0.0003
mean SI ~ mineral depth * combined depth	47.72	0.321	0.476	-0.191	0.107	0.0003
mean SI ~ peat depth * combined depth	45.47	0.15	0.789	0.016	-0.017	0.0005

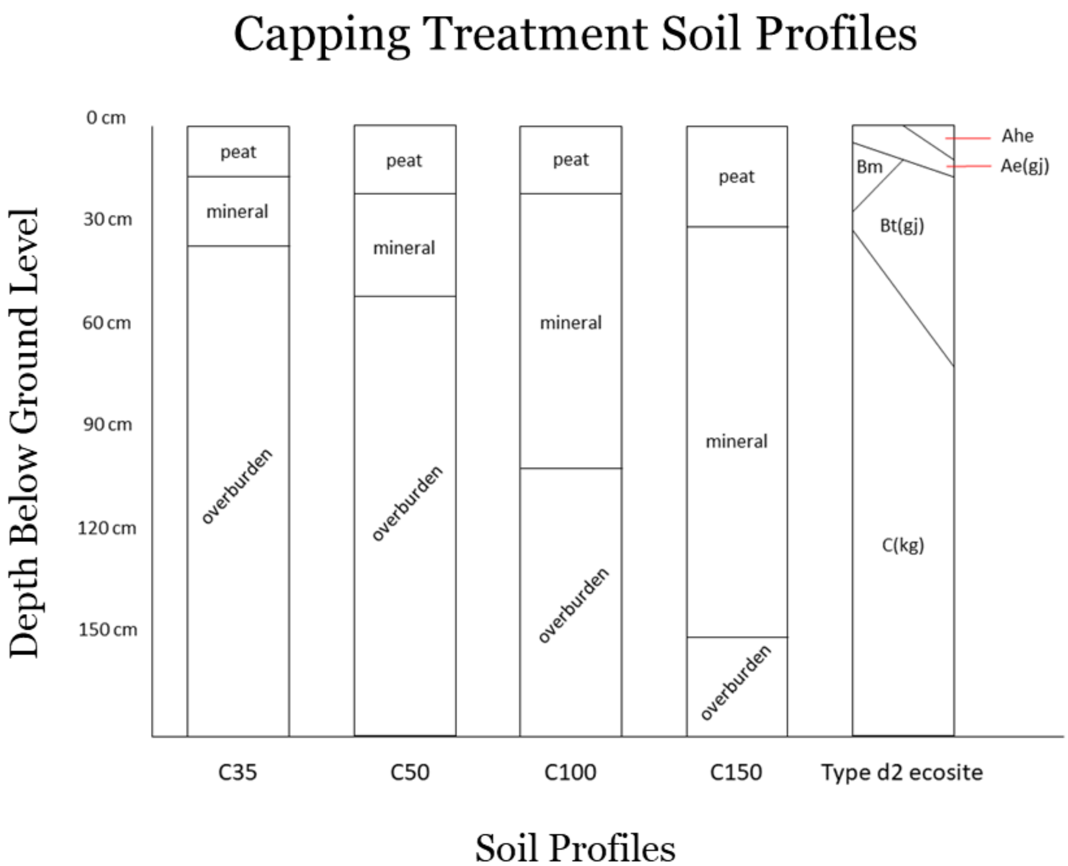

Figure 2.1 Illustration of capping depths present in this study, as compared to the typical soil profile of the targeted natural ecosite (type d ecosite, per Beckingham & Archibald, 1996). There are four capping depths considered in this thesis, C35, C50, C100, and C150. C35 and C50 are only present on South Bison Hills Instrumented Watershed (SBH), whereas C150 is only found in the 9-year cohort (sites capped since 2007). C100 is found on SBH, and in both the 9-year cohort and 13-year cohort (sites capped before 2007). The reference soil profile (far right) is a subset of a type d ecosite, in this case a type d2 ecosite; this was chosen as the canopy cover for this ecosite, a mixture of trembling aspen and white spruce, most resembles the canopy of the reclamation sites. The soil profile for a type d2 ecosite is composed of the following horizons: an A horizon that is roughly 5-15 cm thick, a B horizon that is roughly 25-55 cm thick, and a C horizon that is roughly 30+ cm. An LFH layer (not shown) is usually present, and is roughly 2-10 cm thick for this site type.

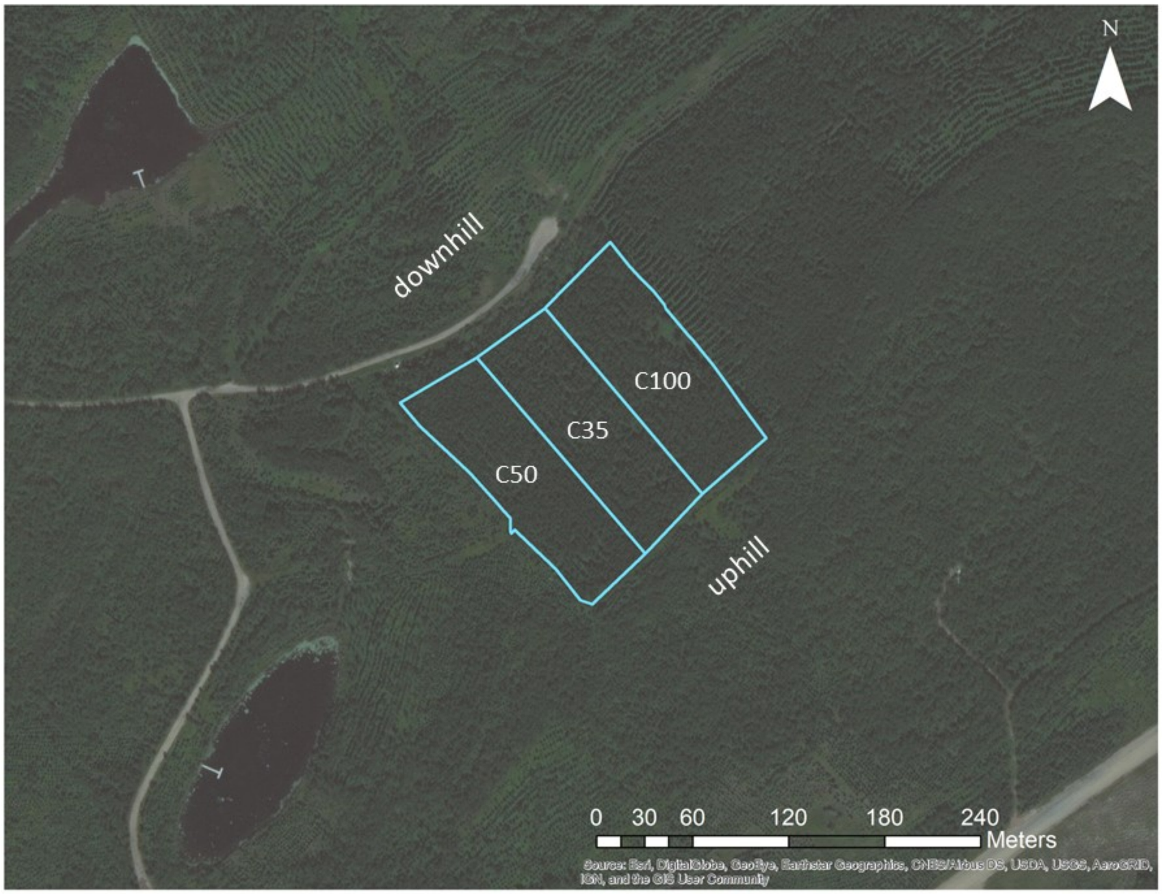

Figure 2.2 Map of South Bison Hills Instrumented Watershed, indicating the three capping treatments, and direction of the hillslope. Overview of South Bison Hills Instrumented Watershed (SBH). Treatment dimensions are roughly 50 m by 150 m, with each treatment covering roughly 0.8-0.9 ha, while total treatment area is about 2.5 ha. The site has a 5:1 slope, from the SE to NW. The underlying image is from ESRI's *World Imagery* basemap, with specific credits to DigitalGlobe (image dated 12 September 2016).

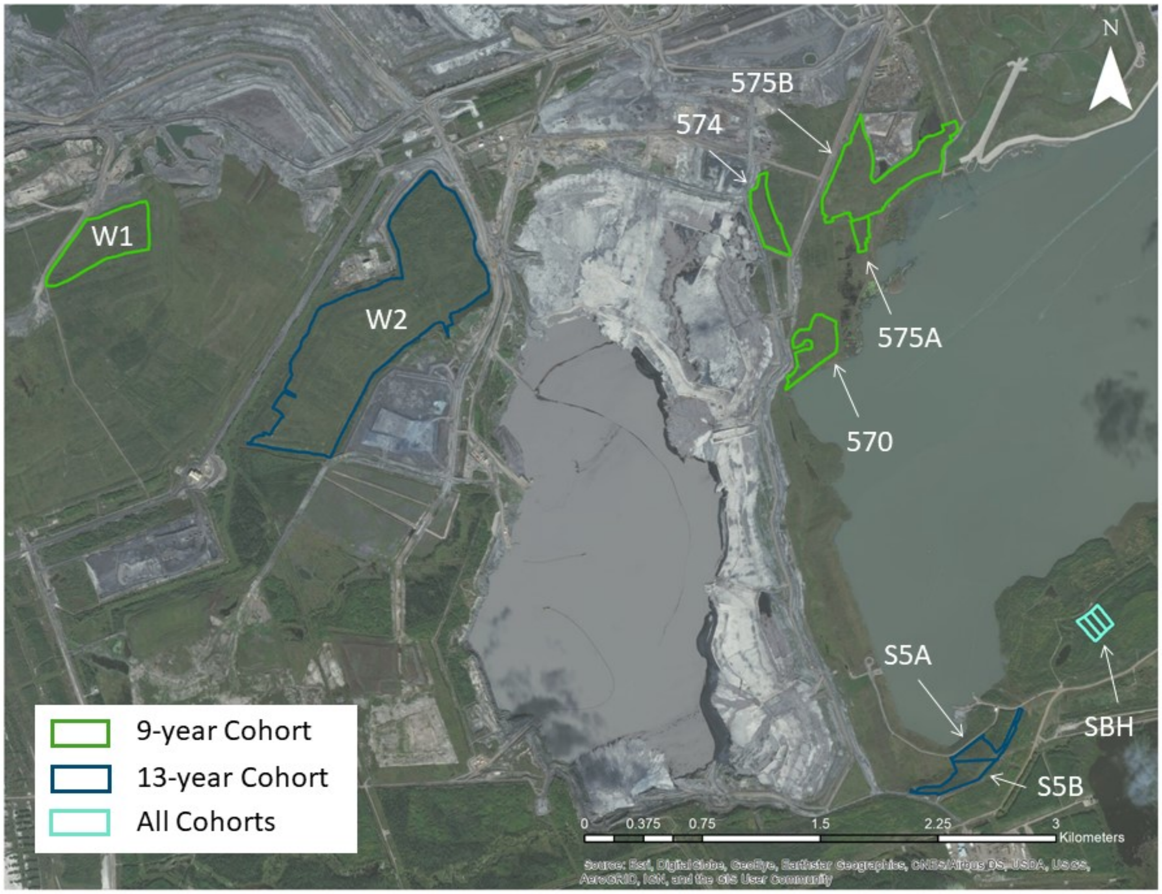

Figure 2.3 Map of the nine sites sampled in 2018, outlining which sites belong to the 13-year versus the 9-year cohorts. Historic data from the South Bison Hills Instrumented Watershed (SBH), collected in 2012 and 2008 was used to complete the 13-year and 9-year cohorts, respectively. Sites W2, S5A, S5B, W1 and 575A had treatment C100, while sites 575B, 574, and 570 had treatment C150. SBH had treatments C35, C50, and C100, with C100 being the far-right plot (Figure 2.2). The underlying image is from ESRI's *World Imagery* basemap, with specific credits to DigitalGlobe (image dated 12 September 2016).

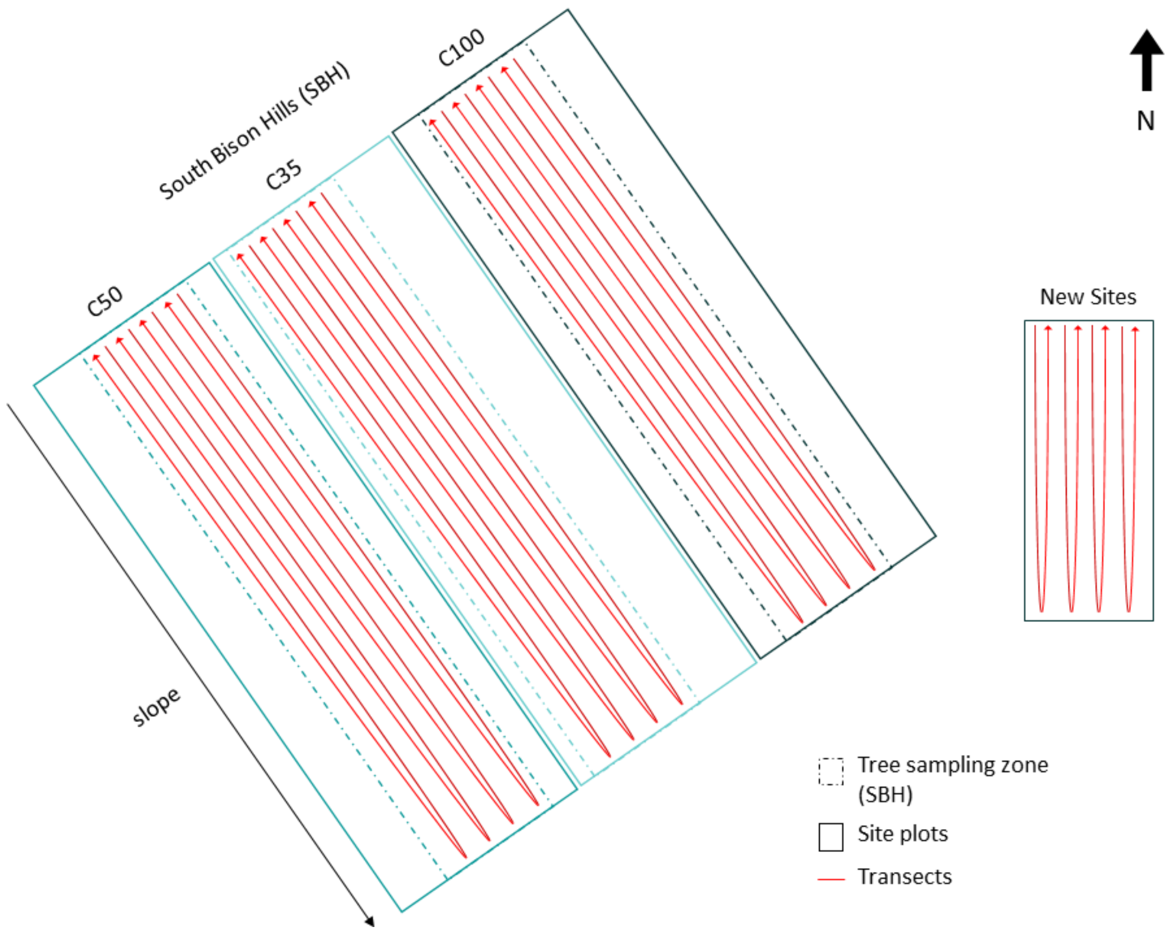

Figure 2.4 Overview of the sampling protocol used in 2018, demonstrating the tree-sampling transects used in 2018. For SBH, transects followed the orientation of the treatment plots. Site plots are roughly to scale. Trees to be sampled were identified by tree ID, as tagged in 2005. As a result, trees that had been sampled in 2005, 2008, and 2012 were able to be resampled. This limited the tree sampling zone to a portion of the actual capping treatment or site plot. At the new sites, trees were sampled from the entirety of the site plot. All sampling was completed in four transects for each site plot.

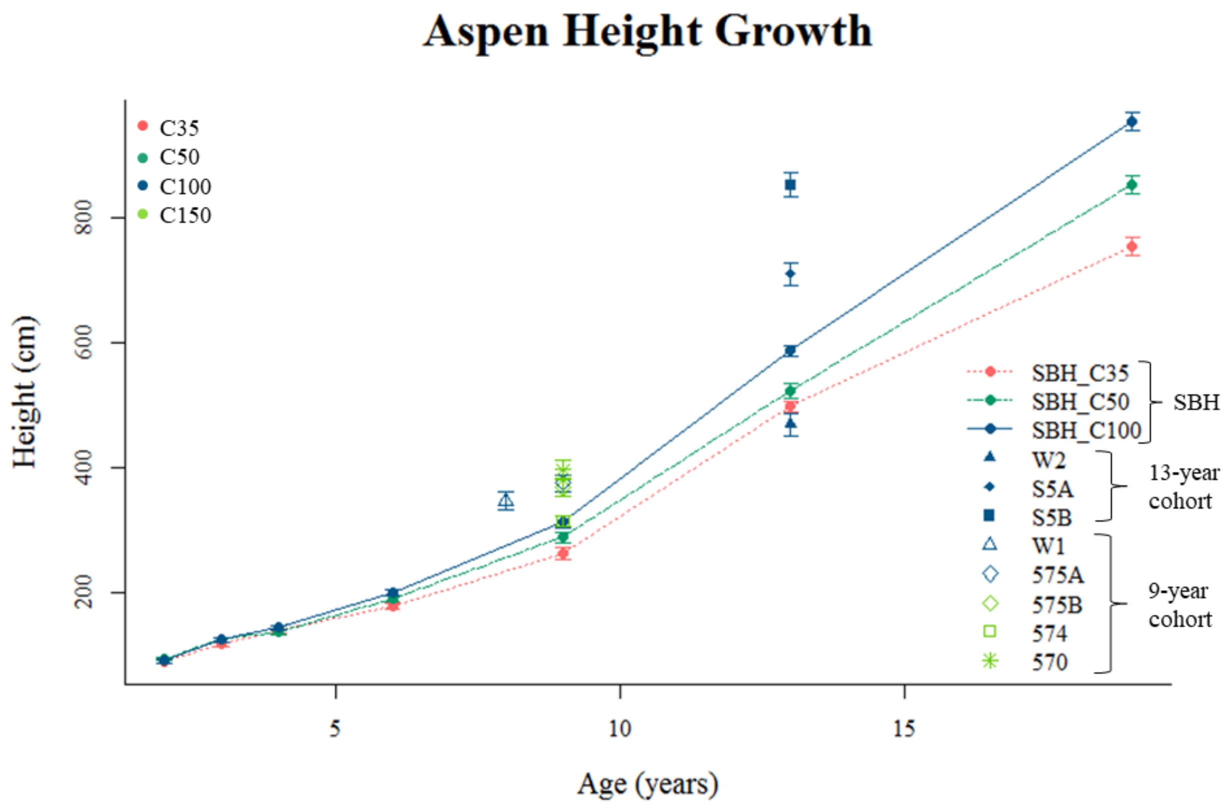

Figure 2.5 Growth trajectory for trembling aspen height, 2-19 years post-planting, at South Bison Hills Instrumented Watershed (SBH), overlain with the 2018 data from the 9- and 13-year cohorts. Error bars represent standard error.

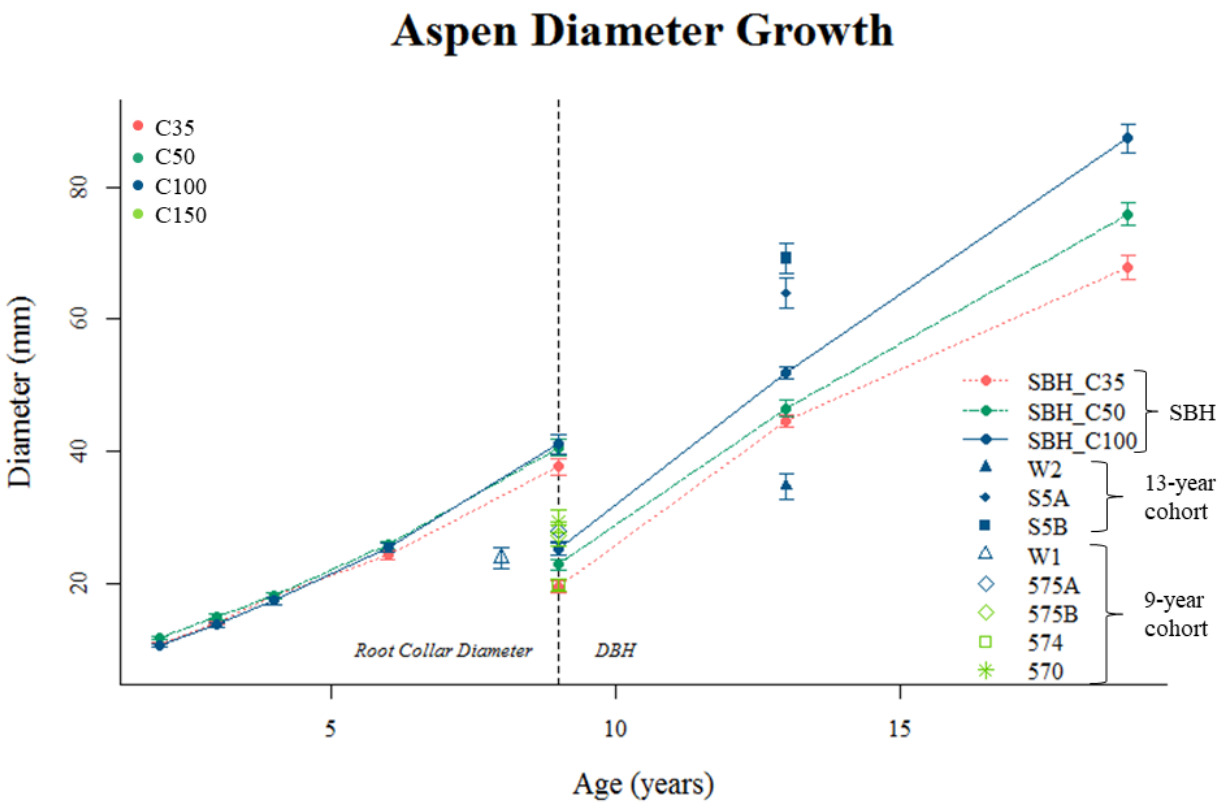

Figure 2.6 Growth trajectory for trembling aspen dbh, 2-19 years post-planting, at South Bison Hills Instrumented Watershed (SBH), overlain with the 2018 data from the 9- and 13-year cohorts. Light blue represents the growth on the 35 cm treatment on SBH (C35), blue the growth on the 50 cm treatment (C50), and dark teal the 100 cm treatment (C100). Sites of the 13-year and 9-year cohorts represented by dark teal also had the 100 cm treatment, and sites represented by green had the 150 cm treatment (C150). The vertical dotted line indicates sampling done in 2005, in which both root collar diameter (measured from 2001 to 2005 inclusive) and dbh (measured 2005 through 2018) were measured on each tree. The double headed arrow highlights site S5B, for which mean dbh at 13 years for the C100 treatment was comparable to mean dbh of treatment C35 at SBH at 19-years post-planting. Error bars represent standard error.

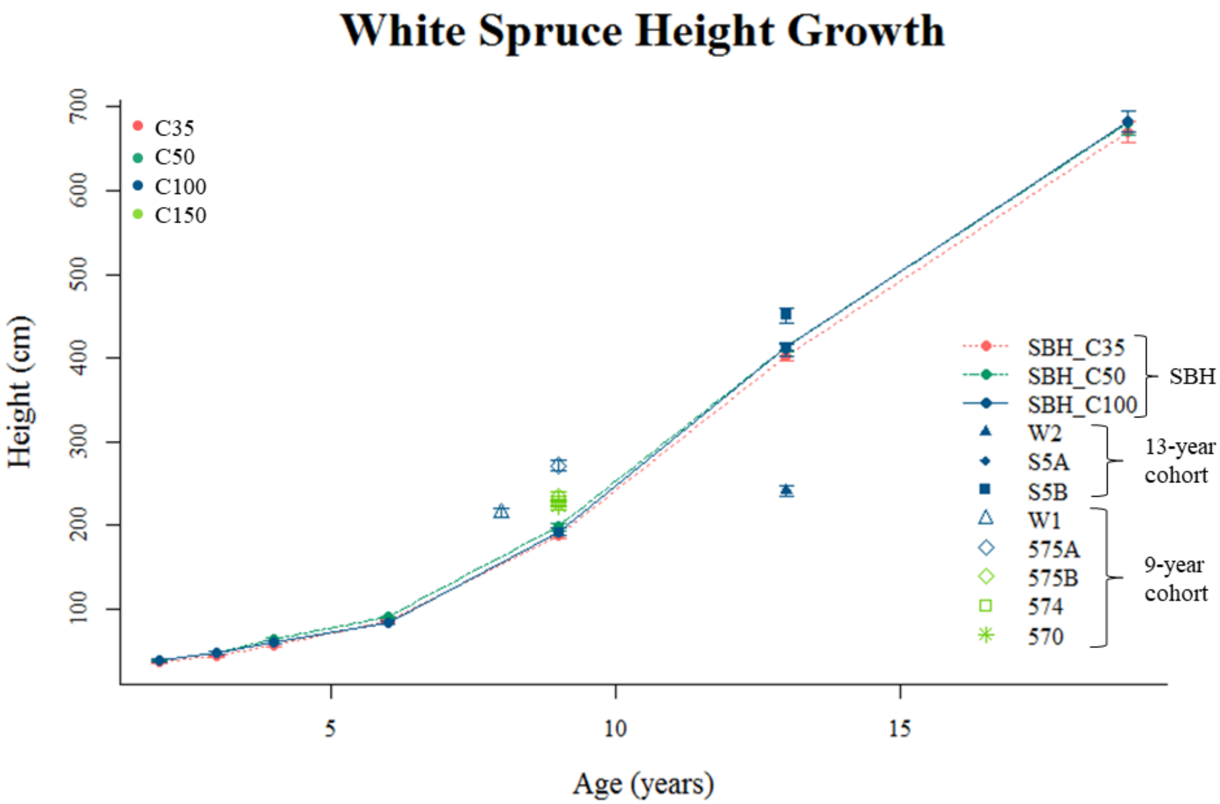

Figure 2.7 Growth trajectory for white spruce height, 2-19 years post-planting, at South Bison Hills Instrumented Watershed (SBH), overlain with the 2018 data from the 9- and 13-year cohorts. Error bars represent standard error.

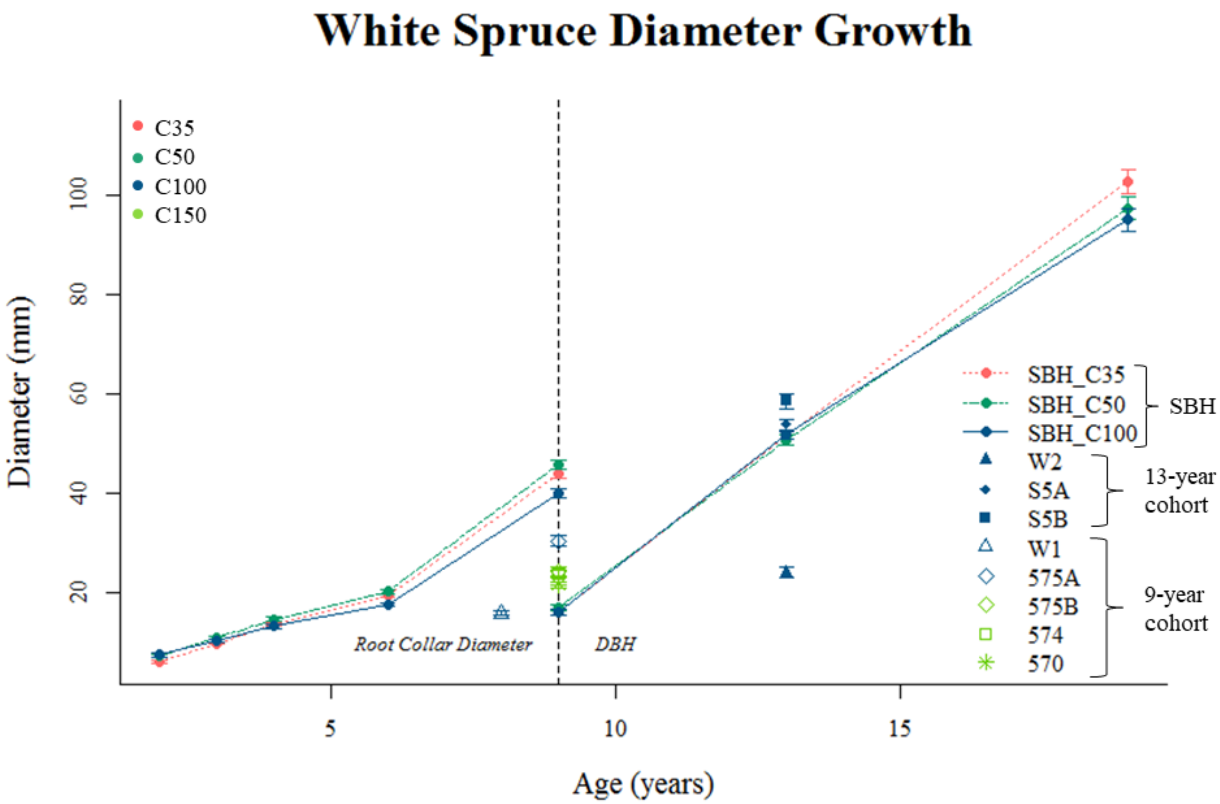

Figure 2.8 Growth trajectory for white spruce dbh, 2-19 years post-planting, at South Bison Hills Instrumented Watershed (SBH), overlain with the 2018 data from the 9- and 13-year cohorts. Light blue represents the growth on the 35 cm treatment on SBH (C35), blue the growth on the 50 cm treatment (C50), and dark teal the 100 cm treatment (C100). Sites of the 13-year and 9-year cohorts represented by dark teal also had the 100 cm treatment, and sites represented by green had the 150 cm treatment (C150). The vertical dotted line indicates sampling done in 2005, in which both root collar diameter (measured from 2001 to 2005 inclusive) and dbh (measured 2005 through 2018) were measured on each tree. The double headed arrow highlights site W2, for which mean dbh at 13 years was comparable to mean dbh of sites 9-years post-planting. Error bars represent standard error.

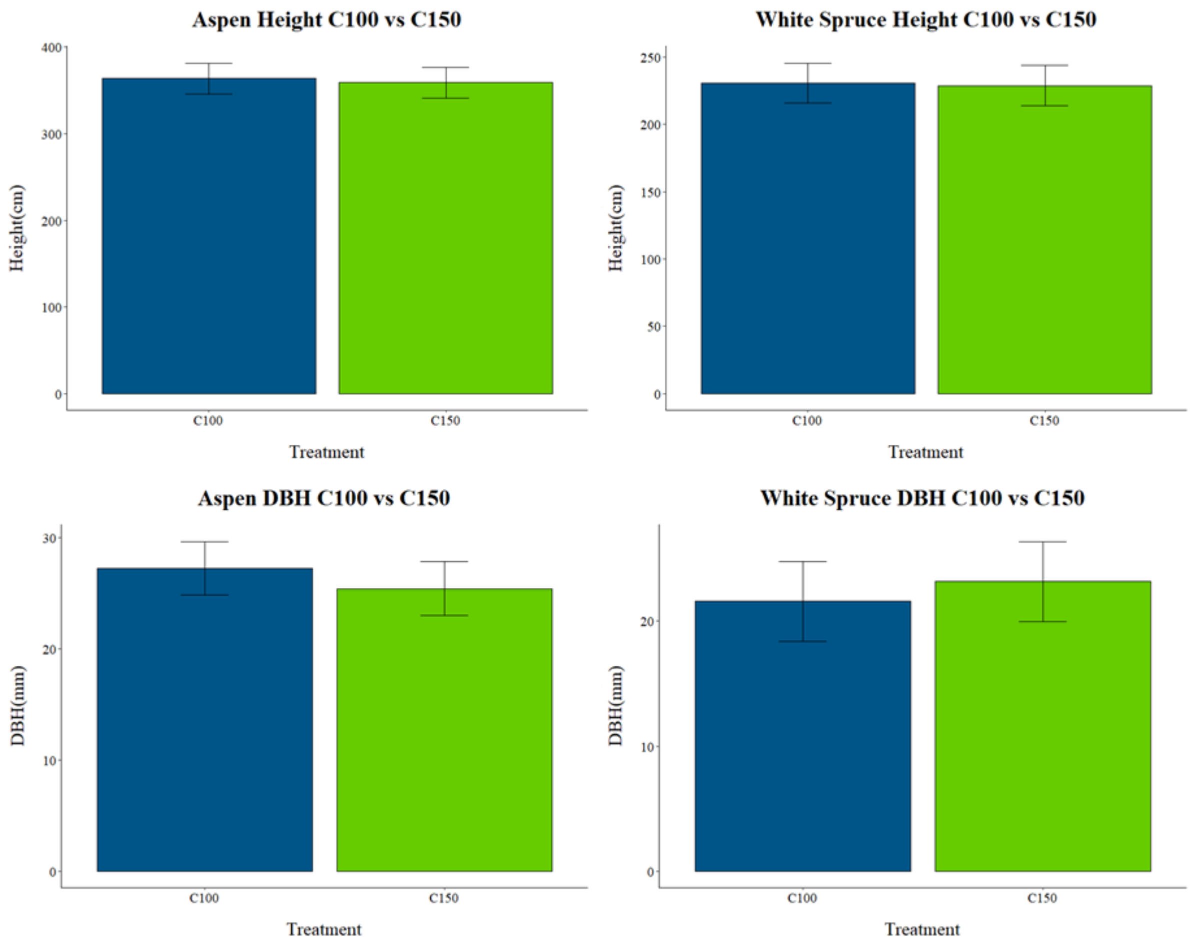

Figure 2.9 Comparison mean height and dbh for the C100 and C150 treatments at eight to nine years post-planting, for the three sites receiving the C100 treatment and the three sites that received the C150 treatment. Data for C150 and most of C100 were collected in 2018, while the data for the C100 treatment at South Bison Hills Instrumented Watershed (SBH) was collected in 2012. Error bars represent standard error. There was no significant difference in growth between the two treatments for either white spruce (based on a mixed model ANOVA with random effects: height: $p = 0.93150$, $n = 6$; dbh: $p = 0.5921$, $n = 6$) and trembling aspen (height: $p = 0.7924$, $n = 6$; dbh: $p = 0.5463$, $n = 6$).

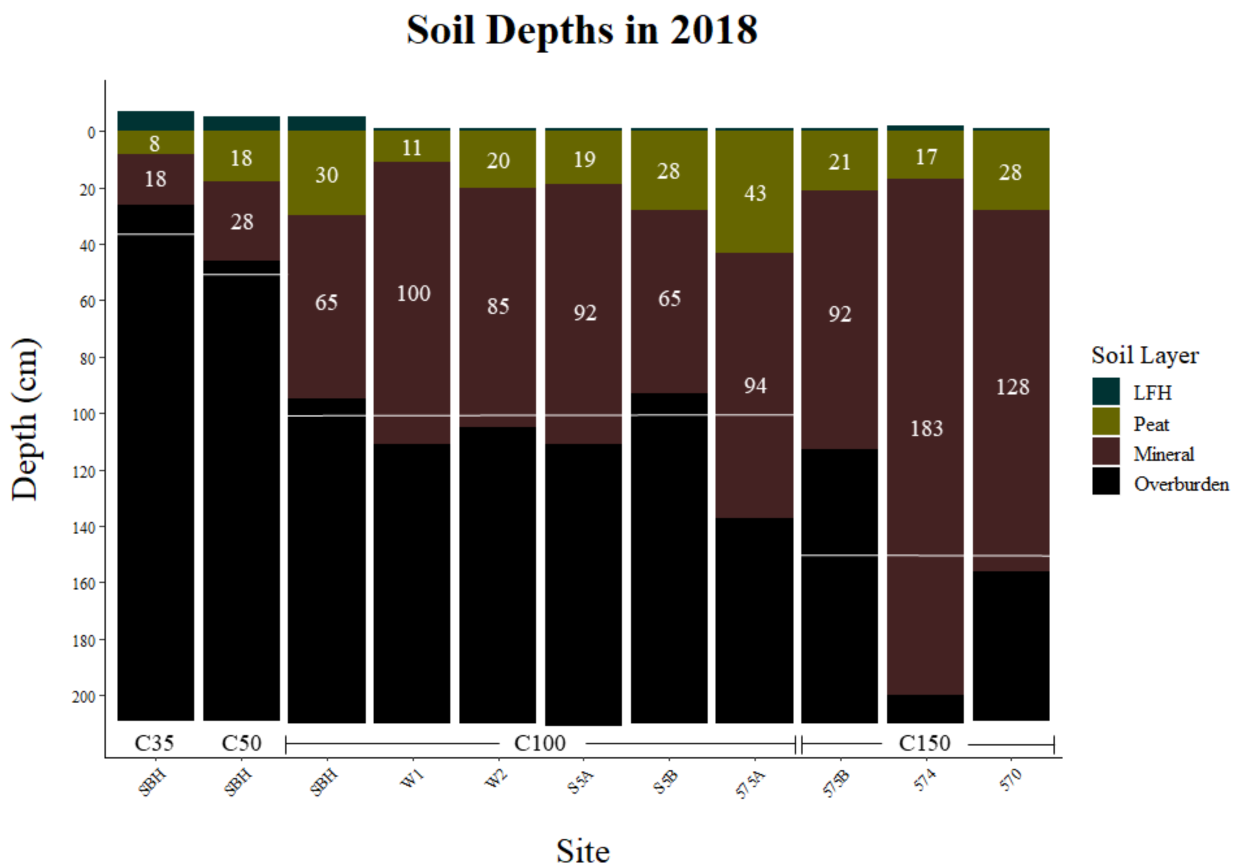

Figure 2.10 Results of soil sampling at each site, as sampled in 2018. Each site was sampled in three locations (except for 575B, which had two sample locations); the mean of these samples was used to create the site soil profile. Treatment C35 is targeted to have 15 cm of peat layer and 20 cm of mineral layer, for a total of 35 cm of capping. Treatment C35 is targeted to have 15 cm of peat layer and 20 cm of mineral layer, for a total of 35 cm of capping. Treatment C50 is targeted to have 20 cm of peat layer and 30 cm of mineral layer, for a total of 50 cm of capping. Treatment C100 is targeted to have 20 cm of peat layer and 80 cm of mineral layer, for a total of 100 cm of capping. Treatment C150 is targeted to have 30 cm of peat layer and 120 cm of mineral layer, for a total of 150 cm of capping. Numbers in white are the mean sampled depth of the peat or mineral layers, as observed in 2018. Horizontal white lines indicate the planned total depth (peat + mineral soil layers) of each treatment, against the observed depths. A summary of site establishment conditions is found in Table 2.1.

Chapter 3: Vegetation community development and differences 3-19 years post-reclamation on Clearwater overburden

3.1 Introduction

Forest reclamation following severe industrial disturbance is a complicated process, and the establishment of a healthy understory vegetation community is often overlooked when determining the success of reclamation. The importance of a healthy understory cannot be understated, however, as it represents the majority of plant biodiversity in forest ecosystems provides a myriad of other ecological benefits, such as providing microsites for tree seedling establishment and driving nutrient cycling (Macdonald et al., 2015a). Understory communities are tied to the health and development of the overstory trees (Barbier, Gosselin, & Balandier, 2008), and in the boreal forest, cycles of natural disturbance lead to expected patterns of recovery and succession that can be used as models for reclamation progress and success (Macdonald, Quideau, & Landhäusser, 2012).

The Athabasca Oil Sands Region (AOSR) is a current and future source of industrial disturbance requiring reclamation. The minable area covers roughly 475 000 ha, much of which will need to be reclaimed to an equivalent capability boreal forest (Rooney, Bayley, & Schindler, 2012; Government of Alberta, 2005; Alberta Environment and Parks, 2010). There are a variety of materials and sites that require reclamation, each with their own challenges. Clearwater overburden, the focus of this study, is a saline material excavated during the mining process (Fung & Macyk, 2000). Salinity in high concentration can prove detrimental to the growth of many boreal forest species, altering the composition of the understory (Lilles et al., 2012). In the case of Clearwater overburden reclamation, a capping treatment is used to minimize salt ingress into the rooting zone, which can help with the establishment of species typical of non-saline sites; a thicker layer of capping is presumed to allow for greater depth of salt-free soil (Kessler et al., 2010). However, thicker capping treatments require more materials, and are thus more expensive; balancing cost and effectiveness are important for the practical applicability of reclamation prescriptions.

Materials used for capping are sourced from areas newly disturbed by mining operations; thus there are limits on total available capping materials. Typically, Clearwater overburden capping treatments contain two main layers, a nutrient-rich upper layer, and a mineral layer (Rowland et al., 2009; Turcotte, Quideau, & Oh, 2009). The upper layer can be composed of peat from excavated peatlands (known as peat-mineral mix, or PMM soils), or forest floor material harvested from upland forests (known as FFM soils) (Macdonald et al., 2015a). Peat is notable for its high water-holding capacity and low nutrient content (Béasse, Quideau, & Oh, 2015; van Breemen, 1995), whereas forest floor material has higher nutrient content, and can provide an upland seed bank (Mackenzie & Naeth 2010; Mackenzie & Quideau, 2012). The seed bank provided by the upper capping layer is sometimes the only “active” seeding of native understory vegetation; site-specific conditions, such as wind-dispersed seed from nearby vegetated sites (disturbed or undisturbed), may fill in the resultant gaps (Macdonald et al., 2015a). The mineral layer material is sourced from the lower horizons of these same areas, and can contain a variety of materials and textures Macdonald, Quideau, & Landhäusser, 2012). As the majority of the minable oil sands area is peatland (Raine, Mackenzie, & Gilchrist, 2002), there is generally greater availability of peat and thus a higher prevalence of use of PMM as a capping material for reclamation. However, it is important to note that the majority of reclamation on Clearwater overburden targets establishment of upland forest (Rooney, Bayley, & Schindler, 2012; Rowland et al., 2009).

In assessing the success of reclamation from the perspective of the understory plant community comparisons can be made to comparable stages in the natural development or succession of forest understory, especially when following natural disturbance (Government of Alberta, 2013; Alberta Environment and Parks, 2010; e.g. Errington & Pinno, 2015). However, in the early post-disturbance time period, colonization by weedy or invasive species can occur on reclaimed sites (Audet, Pinno, & Thiffault, 2014; Errington & Pinno, 2015). There is a general expectation that, as forests progress through a natural succession cycle, and in particular once the canopy has closed, more natural understory conditions will resume, and these weedy species will decrease (Lieffers & Stadt, 1994; Rowland et al., 2009; Macdonald et al., 2015b). This process of succession takes time; natural forest cycles are variable, but some boreal forest fire cycles have been found to last many centuries (Bergeron et al., 2001). Thus, it is important to investigate the long-term recovery of forest understory communities when dealing with forest reclamation, to be able to truly evaluate the success of reclamation prescriptions. Despite an abundance of previous studies on forest reclamation, including in the AOSR, most have considered early vegetation growth (e.g. Hoffman, 2017; Mackenzie & Naeth, 2010; Errington & Pinno, 2015; deBertoli, 2018; Sloan & Jacobs, 2013), while relatively few have examined the understory plant community over the longer term to fully understand patterns of the community development (but see Rowland et al., 2009; Pinno and Hawkes, 2015; Dhar, Comeau, & Vassov, 2019).

The goals of this study were to assess understory vascular plant community development over the longer term for reclamation sites in the AOSR with PMM capping treatments. A 19-year-old site with three different capping treatment depths was assessed at various points in time, from shortly after trees had been planted until shortly after canopy closure. I predicted (1) that the communities on these three treatments would progress through a similar trajectory of succession, and would, by the most recent sampling, be approaching a more natural forest understory, with high litter coverage and a high proportion of native species. Additional several 13-year-old sites that received the same depth of capping with PMM were sampled, and I predicted (2) that at this point, there would be a great variability between sites in terms of understory composition. This is despite the shared capping treatment, due to the lack of active seeding or planting of understory species at these sites. This investigation of understory development will clarify the influence of capping treatment in understory development, and determine whether expectations of a return to a more natural state after canopy closure have been met under the current practices.

3.2 Methods

3.2-1 Research Area and Sites

Research was conducted at reclamation sites in the Mildred Lake Mine Area (57°02'29"N, 111°36'34"W), operated by Syncrude Canada Ltd. This oil sands mining operation is located within the Athabasca Oil Sands Region (AOSR), roughly 40 km north of Fort McMurray, in northeastern Alberta, Canada. This area has a mean annual precipitation of 418.6 mm, and a mean annual air temperature of 1.0 °C (Government of Canada, 2018).

The research area is situated within the Central Mixedwood natural subregion of the Boreal Forest (Natural Regions Committee, 2006). This subregion, which covers approximately one quarter of the province of Alberta, is composed of roughly 60% upland forest and 40% wetland and lowland forest areas (Natural Regions Committee, 2006). The upland areas are commonly dominated by forests in which the canopy is comprised mainly of trembling aspen (*Populus tremuloides* Michx.), white spruce (*Picea glauca* (Moench) Voss), and in certain areas, jack pine (*Pinus banksiana* Lamb.) (Natural Regions Committee, 2006). Typical soils are grey luvisols (Natural Regions Committee, 2006).

This study focused on Clearwater overburden reclamation. For each of the sites included in this research, Clearwater overburden forms the topographical structure of the site, and a capping treatment lies overtop, providing a rooting media for vegetation. There are three capping treatments included in this study, and each consists of different thicknesses of two layers: an upper, organic-rich peat layer composed of a peat/mineral mix, and a lower, clay-rich mineral layer (Figure 3.1). Treatment C35, the thinnest treatment, has 35 cm of capping in total: 15 cm of peat layer over 20 cm of mineral layer. Treatment C50 is 50 cm thick, with 20 cm of peat layer over 30 cm of mineral layer. Treatment C100 is 100 cm thick, with 20 cm of peat layer over 80 cm of mineral layer.

After the landform was created using the overburden material and the capping treatment was applied over the overburden, native trees were planted onto the sites. Reclamation outcomes for the research sites targeted a type d ecosite (M. Yarmuch, personal communication), which is a common forest type in the Central Mixedwood subregion, and is dominated by aspen and/or white spruce (Beckingham & Archibald, 1996). Thus, these sites were planted with roughly equal densities of aspen and white spruce. My research focused on the understory community composition at three different subgroups of sites, whose specific establishment conditions are described below. The main study site was a research site established with three capping treatments, and sampled at various points in time over the 19 years since its reclamation. Additional sites each had only one capping treatment (C100), and were sampled at 13 years post-reclamation. Specific establishment conditions for all of these sites are described below (see also Table 3.1).

3.2-1.1 South Bison Hills Instrumented Watershed

The main study site is located at the SW30 Overburden Research Site, also known as the South Bison Hills Instrumented Watershed (SBH). Here, Clearwater overburden was used to construct a hillslope with a 5:1 slope (Kelln et al., 2009). In 1999, three capping treatments were established over the overburden, C35, C50, and C100 (Kelln et al., 2009; Syncrude Canada Ltd., 2000). Each treatment covers about 0.8-0.9 ha of the site (Figure 2.2). This combination of capping treatments was established as a capping trial, and was not replicated elsewhere on Syncrude reclamation sites. In the fall of 1999, each capping treatment was planted with alternating rows of white spruce and trembling aspen. A total of 1600 stems/ha were planted (Syncrude Canada Ltd., 2000). At the same time, this site was seeded with annual barley (*Hordeum vulgare* L.) and fertilized (Syncrude Canada Ltd., 2000). In 2018, this site was 19 years post-planting.

3.2-1.2 13-Year Cohort and Site Selection

The 13-year cohort is made up of three sites that were 13 years post-planting in 2018. Data from these sites was complemented by pre-existing data from SBH collected 13 years post-planting in 2012. The three 13-year-old sites were also reclaimed Clearwater overburden, capped in 2003, and planted in 2005 (Syncrude Canada Ltd., 2004; Syncrude Canada Ltd., 2006). Each of these sites received the C100 treatment, which was the standard capping depth required on Syncrude Canada Ltd. sites at the time (Syncrude Canada Ltd., 2013).

Selection criteria focused on finding sites with establishment conditions as similar to SBH as was reasonable. Primarily, each site had similar tree planting densities of aspen and white spruce to SBH (800 +/- 200 stems/ha/species). Only sites that were at least 1 ha in size were considered; in the end, selected sites ranged from 3.3 ha to 97.8 ha. When selecting a sampling location within each site, areas with a noticeable slope were preferred, to be similar with SBH; however, none of the sites had as strong a slope as SBH. Compatibility of sites with the selection criteria was determined using the 2016 Syncrude Conservation and Reclamation geodatabase (Government of Alberta, 2018), viewed with ArcGIS 10.2.1 (ESRI, 2014), and confirmed with site visits in July 2018. Sites were delineated according to the reclamation ID assigned to each site (Government of Alberta, 2018). The selected sites will be referred to as W2, S5A, and S5B (see Figure 3.3 for a map of sites; please refer to Appendix A for a list of alternate site aliases). Table 3.1 summarizes the establishment conditions of each site in the 13-year cohort.

3.2-2 Historic Data Collection at SBH

Plant community composition was previously assessed in 2002, 2005, and 2012 at SBH. Percent cover was used in 2002 and 2005, with two quadrats per slope position (top, mid, or bottom) and capping treatment (nine total combinations with two quadrats each giving 18 quadrats total). Quadrats were 2 m by 2 m in dimension. Cover percentage (as a class code) was also used in 2012, again with two quadrats per slope position within capping treatment (18 quadrats). Quadrat location was more specific however; for each quadrat pair, one quadrat was placed in the southeast of the treatment condition (slope versus capping treatment), and the other in the northwest. Quadrats were 10 m by 10 m in dimension, which are the dimensions required for ecosite classification according to Beckingham & Archibald (1996). A summary of cover classes and codes used can be found in Table 3.2. In each year visual estimates of cover were made for each vascular plant species; in 2002 cover was also estimated for bare ground, and in 2005 and 2012, litter and moss cover (total ground cover, not differentiated by species) were also included.

3.2-3 2018 Data Collection

In 2018, I sampled nine vegetation quadrats per site, each 2 m by 2 m. At SBH, each treatment was treated as a separate site, which equated to 27 quadrats total. Quadrats were placed within the tree-sampling region of each site (Chapter 2), in order to have corresponding vegetation community and tree growth data. To address prediction 1, vegetation sampling at SBH in 2018 was used along with the historical data to examine changes over 17 years of vegetation sampling, and up to 19 years post planting. Vegetation sampling at the three new sites, in addition to the historical data at SBH from 2012, was used to address prediction 2, and create a snapshot of vegetation communities 13 years post-planting.

Quadrat layout was designed to be distributed across the entire site (Figure 3.4). Quadrats were distributed throughout the site by establishing three in each of the upper slope, mid slope, and lower slope portions of the site, and within each of those one quadrat was placed in each of the east, centre, and west portions. At the three new sites, plots of 30 m by 70 m were established (per Chapter 2), and the sampling quadrats were laid out within this area using a similar approach as at SBH (Figure 3.4). New site plots were purposely aligned with cardinal directions, meaning that the 30 m borders ran east-west, and the 70 m borders ran north-south. This was done to simplify and standardize the plots, in contrast to the non-cardinal ordination of SBH. Since the new sites had lesser slopes than SBH the quadrats were distributed throughout the site by establishing three in each of the north, centre, and south portions of the site, and within each of those one quadrat was placed in each of the east, centre, and west portions.

At each quadrat, visual estimates of percent cover were made for each vascular plant species (Table 3.2). Pigtails were used to mark quadrat boundaries during the survey. In addition to vegetation, the cover of bare ground, downed woody debris, moss, and litter were also recorded. Unknown species were sampled and pressed, for later identification in the lab. Species suspected to be rare were instead documented with photographs.

3.2-4 Data Analysis

The R Software Environment (R) version 3.5.1 (R Core Team, 2018) was used for statistical analyses and graphical data representation. ArcGIS 10.2.1 (ESRI, 2014) was used to create sitemaps and visualize GPS data.

3.2-4.1 Vegetation Changes at SBH Over Time

To test prediction 1, I used non-metric multidimensional scaling (nMDS) to visualize differences in community composition among the three treatments, C35, C50, and C100 and changes over time (2002, 2005, 2012, and 2018) at SBH. The 2012 cover classes were converted to percentages; species that were found in less than 5% of all sample plots (across all years) were removed to avoid undue influence from uncommon species. I then calculated a mean percent cover (per site) for each remaining species, for each treatment and year. Using *bcdist* from the *ecodist package*, I calculated Bray-Curtis distances, and used nMDS (*nmds* function) to visualize the variation in vegetation communities. The nMDS scores were then used to add successional vectors to the plot, in order to determine if there was a discernable trajectory over time for the vegetation communities. With *enfit* from the *vegan* package, I used the Bray-Curtis distances to identify significant (a = 0.05) plant species that differentiate between the treatments and time points. Significant species were plotted as vectors on the nMDS output.

3.2-4.2 13-Year Cohort

I again used nMDS to test prediction 2 and compare among sites for the 13-year cohort. This included the 2018 data from the three C100 treatment sites and the 2012 data for the C35, C50, and C100 treatments at SBH. Although the main intention was to compare sites with treatment C100, the C50 and C35 treatments at SBH were included in the ordination to be able to understand both the relationships within SBH, and between SBH treatment C100 and the sites of the 13-year cohort. The 2012 cover classes from SBH were converted to percentages, and then I selected to remove species that were found in less than 5% of all sample plots, desiring to keep only "common" species. I calculated a mean percent cover for each remaining species, for each site and treatment. Using *bcdist* from the *ecodist package*, I calculated Bray-Curtis distances, and used nMDS (*nmds* function) to visualize the variation in vegetation communities. With *enfit* from the *vegan* package, I used the Bray-Curtis distances to identify significant (a = 0.05) plant species that differentiate between the sites. Significant species were plotted as vectors on the nMDS output.

3.2-4.3 Vegetation Composition and Ecosite

Vegetation composition was assessed graphically in two ways, to add depth to the nMDS results that addressed predictions 1 and 2. First, species were separated into vegetation (shrub, graminoid, and forb) and other cover categories (bare ground, moss, and leaf litter), and displayed according to the percent cover of each of these categories. Second, the vegetation species were assigned nativeness designations (native, introduced, and noxious weed), and then displayed according to the percent of vegetation cover that fell into these three categories. Native status was based on the Alberta Conservation Information Management System (ACIMS) database for vascular plants, which lists species native to the province of Alberta (ACIMS, 2018). Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016).

As the data from different sampling years was collected by different people and according to different survey systems (Table 3.2), the total mean percent cover for each site was quite variable. To address these discrepancies between different sampling years, the total mean percent cover for each site and treatment was converted to relative percent cover (where total equaled 100%).

Based on the vegetation survey results, ecosite was determined using the key found in the *Field Guide to Ecosites of Northern Alberta* (Beckingham & Archibald, 1996).

3.3 Results

Vegetation communities at the three South Bison Hills Instrumented Watershed (SBH) treatments displayed a clear trajectory for change over time, which all three treatments progressed through together. The variability in vegetation communities among the 13-year cohort sites was greater between sites than between treatments at SBH. In both cases, introduced and/or weedy species were highlighted as significant in the characterization of these differences and trajectories. The summary of vegetation composition as cover by species type mirrored these trends. An evaluation of ecosite type indicated that most of the sites would key out to the targeted ecosite type d, site W2 was more likely a type b ecosite.

3.3-1 South Bison Hills Instrumented Watershed

The nMDS output indicated a clear trajectory of changing composition over time at SBH (concurring with prediction 1) with differences in the overall community composition among each sampling year (Figure 3.5). It can be seen that the differences over time were greater than the differences between the three treatments. Many of the significant species ($p \leq 0.01$) were native species, but a few were introduced and/or considered noxious. In particular, higher abundances of common dandelion (*Taraxacum officinale* F.H. Wigg.), an introduced weed, were associated with the 2018 sampling.

There were also differences among the treatments and changes over time (prediction 1), in terms of which types of vegetation made up the understory community (C35: Figure 3.6, C50: Figure 3.7, C100: Figure 3.8) and in native versus introduced species (C35: Figure 3.9, C50: Figure 3.10, C100: Figure 3.11). Over time the relative percent cover of vegetation decreased while that of leaf litter increased; cover by bare ground declined to near zero by the end of the sampling period. Treatment C100 was a slight exception, as relative percent cover of litter remained roughly the same in 2018 as in 2012. However, this was due to the high litter cover on C100 versus C35 and C50 in 2012; in 2018, all three treatments had similar relative litter percent cover. In addition, the understory was largely composed of forbs and fern allies (about two thirds to three quarters). The proportion of cover by native species decreased relative to cover by introduced and noxious species (which contradicts prediction 1), but it is important to note that the native species list included foxtail barley (*Hordeum jubatum* L.). This species is native to Alberta but is not a typical boreal forest species, and it was quite dominant at the site in 2002. As of 2018, native, vascular vegetation represented only about one half to one third of the relative total understory vegetation cover, while introduced species and noxious weeds make up the majority, particularly for treatments C35 and C100.

In 2018, treatment C35 had much greater moss coverage than the other two treatments, but less shrub and litter cover. Treatment C50 had the highest cover by native species in 2018. Treatment C100 was fairly similar to treatment C50 in 2018, but had slightly greater litter coverage and less cover by native species.

3.3-2 13-Year Cohort

The nMDs output demonstrated that there was great site-to-site variability in understory vegetation communities (Figure 3.12). Similarly, it also indicated that the three treatments at SBH were more similar to each other than they were to any of the other sites sampled (prediction 2). Many of the significant species ($p \leq 0.05$) were introduced and/noxious weeds.

As predicted (prediction 2) there was substantial variation among sites, both in terms of which types of vegetation made up the understory community (Figure 3.13) and in terms of the proportion of native versus introduced species (Figure 3.14). For example, relative percent cover of litter ranged from 16.2% to 74.3%, and forb cover ranged from 14.0% to 53.1%. The treatments at SBH were quite variable compared to each other in this respect as well; for example, the highest relative forb cover was on treatment C35, and the lowest on C100. With regards to nativeness, the proportion of native species cover for all sites ranged from 38.6% to 79.3%, and the proportion of introduced species cover ranged from 10.5% to 54.3%. In this case the treatments at SBH tended to display more similar trends to each other. For instance, the proportionate cover of noxious species was similarly high at all three SBH treatments (12.4-17-5%) versus the rest of the 13-year cohort (4.7-10.2%).

Looking more specifically at the highs and lows for all sites, there are a few worth mentioning. Overall, treatment C100 at SBH had the highest relative litter cover, and W2 had the lowest. Treatment C35 at SBH had the highest relative bare ground and moss cover. Site W2 had the lowest relative noxious species cover, as well as the highest forb cover. Site S5B had the highest proportionate cover of introduced species, and the highest relative graminoid cover. Treatments C35 and C50 at SBH had the highest proportions of noxious weed cover.

3.3-3 Ecosite Evaluation

According to the key used (Beckingham & Archibald, 1996), in 2018, all three treatment sites at SBH would be considered ecosite type d2.7, as would sites S5A and S5B. This site type has a considerable forb layer, minimal shrub cover (<10%), minimal feather moss cover (<20%), and a mixture of trembling aspen and white spruce.

Interestingly, due to the high shrub cover at W2 (>10%), this site would be considered ecosite type b3.2. Bearberry (*Arctostaphylos uva-ursi* (L.) Spreng.) was common at this site, and green alder (*Alnus viridis* (Chaix) DC.), one of the shrubs planted at the site in 2005, was among the dominant shrubs. Taken in combination with the mixture of trembling aspen and white spruce, this led to the designation of a type b3.2 ecosite.

3.4 Discussion

The vegetation communities at South Bison Hills Instrumented Watershed (SBH) displayed a clear temporal trajectory, from shortly after the site was planted with white spruce and trembling aspen, until the present survey was completed after canopy closure of these trees. As predicted, patterns of changing vegetation composition over time were fairly consistent for all three capping treatments, with only minor differences. Also as predicted, there was considerable variation among sites within the 13-year cohort in terms of understory vegetation. Regardless of the site variance, the prevalence of introduced and weedy species in the understory of all sites sampled in 2018, including the closed canopy of SBH, is a concern.

3.4-1 South Bison Hills 19-years Post Planting

Given that SBH was planted with white spruce and aspen in 1999, and tree canopy had closed by the time of the sampling in 2018, I predicted that this would lead to decreased vegetation cover, but favour forest species over weedy species. There was a marked decrease over time in vegetation cover percentage, especially graminoids and forbs, paired with an increase in leaf litter. However, I did not find that native species were more prevalent than weedy species. For all three capping treatments, native species composed roughly one third to one half of the understory vegetation in 2018, weedy species another third, and the final third or quarter was comprised of noxious weeds. Even when considering the nMDS output, while significant species in 2018 included four native species (*Pyrola asarifolia* Michx., *Rubus idaeus* L., *Carex concinna* R. Br., and *Danthonia intermedia* Vasey), two introduced species (*Taraxacum officinale* F.H. Wigg. and *Poa compressa* L.) were also included. Previous research has found that disturbed sites return to a more natural state as canopy closes (Lieffers & Stadt, 1994; Rowland et al., 2009), which contrasts with my results. The high proportion of litter has been noted by other studies as well, and is attributed to decreased microbial activity in reclaimed peat-mineral mix (PMM) soils (Rowland et al., 2009; Kwak et al., 2015).

The peat layer, especially when stockpiled peat was applied, could in itself be contributing to the lack of native upland forest species found. Stockpiled reclamation soils have lost significant portions of their live propagules by the time they are used in reclamation (Koch et al., 1996; Rokich et al., 2000), and in general sites with stockpiled peat tend to have lower cover and species richness (Dhar, Comeau, & Vassov, 2019). Even in the case of directly placed peat with live propagules, the species that are represented in peat material are typically better suited to the lowland sites the peat is sourced from, rather than the targeted upland forest ecosystems (Mackenzie & Naeth, 2010).

The close proximity of the treatments to each other makes it difficult to compare them as independent entities; just as nearby undisturbed sites are a source for native species (Parrotta, Knowles, & Wunderle, 1997), so too could invasive species easily spread from one treatment to the others. However, any differences in understory composition would thus be more clearly attributable to the effects of the specific treatment. For instance, the markedly higher moss cover in treatment C35 can be inferred to be a result of some impact of the treatment.

3.4-2 Comparing the 13-year Cohort

Data collected from SBH at 13-years post-reclamation was compared with additional sites with treatment C100 that were 13-years post-reclamation in 2018. Given that a number of influential factors affecting establishment differed among these sites, I predicted there would be high site-to-site variability, which would outweigh differences between the treatments at SBH. The results supported this prediction, indicating that vegetation community composition is likely more strongly influenced by these factors than treatment prescriptions. There are a variety of factors that could be contributing to this site-to-site variability. As mentioned above, site location and the exposure to neighbouring seed sources could impact which species are establishing on the site (Parrotta, Knowles, & Wunderle, 1997). Some of the sites were planted with shrub species, which in some cases have been found to facilitate the development of the understory community (Gómez-Aparicio, 2009). In addition, direct placement versus stockpiling of peat could influence which propagules are available for establishment, and the application of fertilizer to some sites early in the reclamation process may also have increased the diversity of outcomes between sites. Finally, the growth of the planted white spruce and aspen could play a role, as in some cases these trees are thriving on sites and in others they are not (Chapter 2), and this overstory influences the understory development (Chen et al., 2018).

Ideally, a larger sample size would better enable us to conclude which factors are the most important for understory development and diversity. This larger sample size should draw from a variety of capping material sources and storage methods, years of reclamation, etc. It would also be worthwhile to examine these sites again after canopy closure, to determine if this results in further changes to the understory.

3.4-3 Ecosite

According to established targets for Syncrude Canada Ltd. reclamation, the aim was for the sampled sites to emulate type d ecosites, as described by Beckingham and Archibald (1996) (M. Yarmuch, personal communication). Interestingly, while most of the sites surveyed would conform to this expectation, one of the sampled sites, W2, better resembled a type b ecosite. Type b ecosites generally have less moisture and nutrient content in their soils than type d, which generally slows the growth of trees on this site type (Beckingham & Archibald, 1996). Aspen, for example, have a mean site index at base height age 50 years (SI) of 15.8 on type b sites, but 18.2 on type d sites (Beckingham & Archibald, 1996). It is possible that this slower growth might make these sites seem less successful, but type b ecosites are still a natural ecosite found in the Boreal Mixedwood, and thus creation of such an ecosite would add diversity to the upland reclamation landscape. Landscape diversity is a highly sought-after in forest management (Côté et al., 2010; Felton et al., 2010; Gauthier et al., 2015; Kuuluvainen, 2002), and diverse landscapes are encouraged according to the *Criteria and Indicators Framework for Oil Sands Mine Reclamation Certification* (Government of Alberta, 2013).

3.4-4 Changes in Reclamation Techniques and Recommendations

Forest reclamation in the Athabasca Oil Sands is an evolving field, and the many changes and improvements to techniques pose a challenge for comparing different sites in this study. Some of these differences, such as the use of stockpiled peat versus directly placed peat, have been discussed above. Another difference is that younger sites, such as those in the 9-year cohort discussed in Chapter 2, no longer received fertilizer or seeded barley during establishment. In order to monitor the effects of these changes in reclamation technique, it would be interesting to revisit the 9-year-old sites when they too are 13 years old, for comparison to the 13-year cohort sites included in this chapter.

There also have been changes in the capping treatment. To address the issues with lack of appropriate native seed propagules in the peat layer, recent reclamation has started using forest floor material (FFM) when available in lieu of peat (Macdonald et al., 2015a). This material is harvested from upland forests, and it has been found to harbour significantly more native seed propagules and contribute better to soil nutrients (Mackenzie & Naeth, 2010). As a result, FMM treatments yield higher species diversity and richness (Errington & Pinno, 2015), and have been found to foster a more natural forest understory community than PMM soils

(Hahn & Quideau, 2013; Errington & Pinno, 2015). It should be noted, however, that sites reclaimed using PMM material for capping are not likely to entirely disappear, as the majority of the Athabasca Oil Sands landscape is comprised of lowlands (Raine, Mackenzie, & Gilchrist, 2002), and thus there is greater availability of peat for reclamation purposes than forest floor material. Future studies should compare similar FFM sites to the PMM sites in this study, ideally developing the 13-year cohort further and characterizing any benefits of this newer reclamation approach.

To increase the cover of native species some changes to reclamation practices might be effective, in addition to the greater application of FFM. Native understory species could be seeded, or introduced with natural forest soil plugs or transplants (Norman et al., 2006; Winterhalder, 2004; Jones & Landhäusser, 2018). The addition of woody debris to reclamation sites has also been shown to increase native species cover in the understory (Brown & Naeth, 2014). In addition, steps could be taken to control, or even prevent, colonization by weedy and noxious species, in an effort to help the native species thrive on the reclamation sites. Cover crops have been found to decrease non-native species, although they do not necessarily increase native species cover (Macdonald et al., 2015b). Application of fertilizers has been found to benefit invasive species (Buss, Stratechuk, & Pinno, 2018). A study on the effects of active removal of weedy species did not find a resultant increase in desired native forest species cover (deBertoli, 2018), but the sites sampled were newly reclaimed, and thus may not be representative of weeding responses in older sites such as SBH.

3.4-5 Conclusions

While the understory vegetation communities at SBH displayed a temporal trajectory over time, the continued relatively high abundance of weedy species indicates there might be a need for additional management of older sites and earlier intervention to control weeds so that reclamation sites sufficiently progress towards reclamation targets. Although ecosite variability may result in sites with lower tree productivity, it may increase landscape diversity and promote differing understory communities.

Table 3.1 Summary of the reclamation procedures and site conditions for each of the study sites, including the capping treatment, peat material placement, tree planting density, and whether sites were additionally seeded with a cover crop, fertilized, or planted with shrubs. Source of information: Syncrude Canada Inc. annual reclamation reports (2000, 2004, 2006) and the 2016 Syncrude Conservation and Reclamation geodatabase (Government of Alberta, 2018). Tree species codes are as follows: Aw = trembling aspen, Sw = white spruce, Pj = jack pine.

Site	Capping Treatment	Peat Placement	Seeding	Fertilizer	Shrubs (stems/ha)	Trees (stems/ha)
SBH 2.5 ha	1999	Direct	1999	1999	N/A	1999
	C100, C50, & C35		20 kg/ha annual barley	363 kg/ha 10N-30P-15K-4S		800 Aw: 800 Sw
W2 97.8 ha	2003	Direct	N/A	N/A	2005	2005
	C100				37.5 saskatoon, 205.5 green alder	1089.1 Aw: 1027.7 Sw
S5A 3.3 ha	2003	Stockpiled	2003	2003	N/A	2005
	C100		28.1 kg/ha annual barley	327.1 kg/ha 10N-30P-15K-4S		867.4 Aw: 1040.5 Sw
S5B 4.1 ha	2003	Stockpiled	2003	2003	N/A	2005
	C100		28.1 kg/ha annual barley	327.1 kg/ha 10N-30P-15K-4S		867.4 Aw: 1040.5 Sw

Table 3.2 Summary of cover classes and their respective cover percentages, as used for historic and the 2018 vegetation surveys. All estimates of cover were made in 2 m by 2 m sampling quadrats, except for in the 2012 sampling which used 10 m by 10 m quadrats. Different sampling classes from different sampling years were aligned as best possible. Classes were converted to a mid-point values for analysis.

<table>
<tr><th colspan="2">2002</th><th colspan="2">2005</th><th colspan="2">2012</th><th colspan="2">2018</th></tr>
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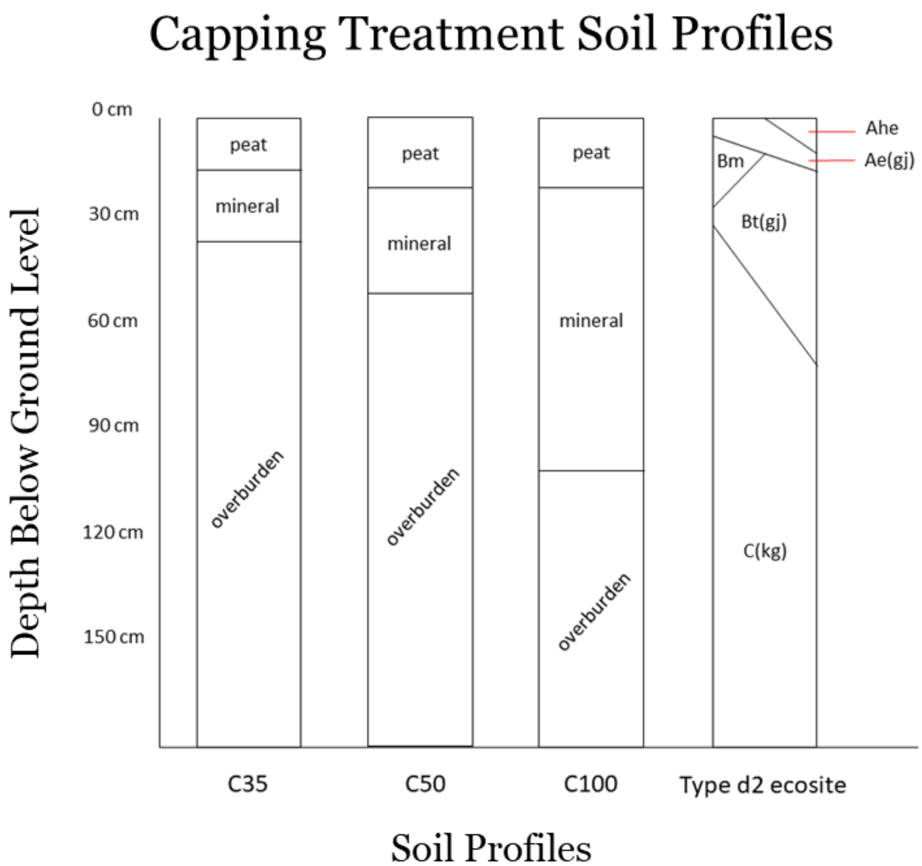

Figure 3.1 Illustration of capping depths present in this study, as compared to the typical soil profile of the targeted natural ecosite (type d ecosite, per Beckingham & Archibald, 1996). There are three capping depths considered in this thesis, C35, C50, and C100. C35 and C50 are only present on South Bison Hills Instrumented Watershed (SBH), whereas C100 is found on SBH and in the 13-year cohort sites that were sampled in 2018. The reference soil profile (far right) is a subset of a type d ecosite, in this case a type d2 ecosite; this was chosen because the canopy cover for this ecosite, a mixture of trembling aspen and white spruce, most resembles the canopy of the reclamation sites. The soil profile for a type d2 ecosite is composed of the following horizons: an A horizon that is roughly 5-15 cm thick, a B horizon that is roughly 25-55 cm thick, and a C horizon that is roughly 30+ cm. An LFH layer (not shown) is usually present, and is roughly 2-10 cm thick for this site type.

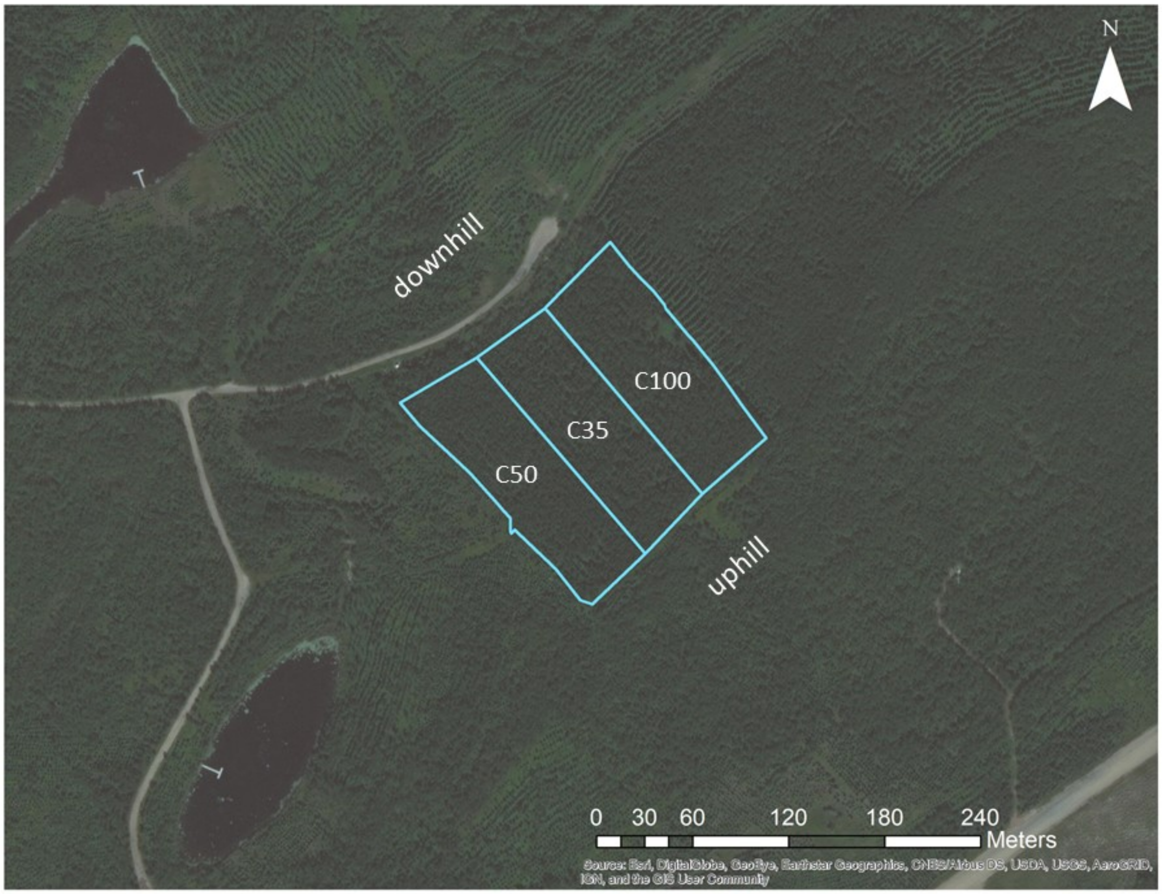

Figure 3.2 Map of the South Bison Hills Instrumented Watershed, indicating the three capping treatments, and direction of the hillslope. The dimensions of the treated areas are roughly 50 m by 150 m, with each treatment covering roughly 0.8-0.9 ha, while total treatment area is about 2.5 ha. The site has a 5:1 slope, from the SE to NW. The underlying image is from ESRI's *World Imagery* basemap, with specific credits to DigitalGlobe (image dated 12 September 2016).

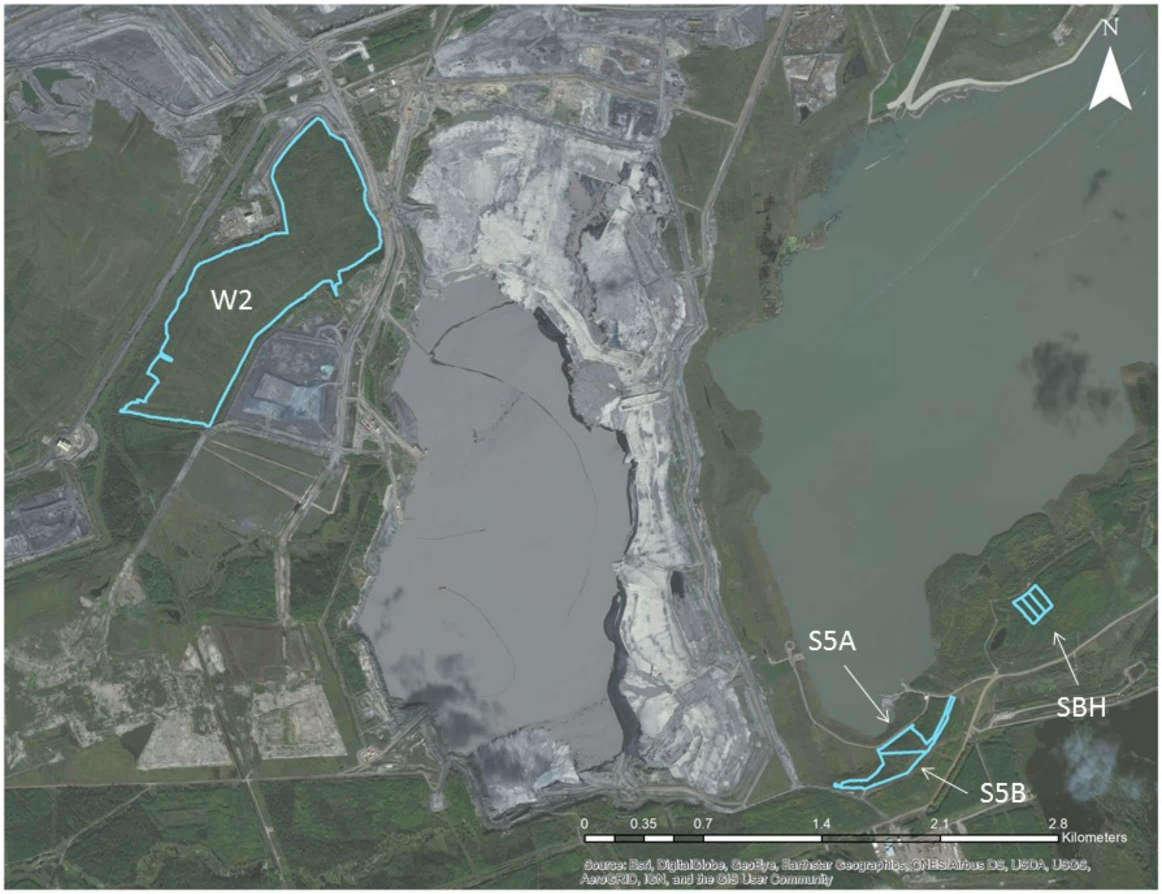

Figure 3.3 Map of the four sites sampled in 2018, including South Bison Hills Instrumented Watershed (SBH) and the three sites of the 13-year cohort. Historic data from SBH, collected in 2008 was used to complete the 13-year cohort. Sites W2, S5A, and S5Bhad treatment C100, while SBH had treatments C35, C50, and C100, with C100 being the far-right plot (Figure 3.2). The underlying image is from ESRI's *World Imagery* basemap, with specific credits to DigitalGlobe (image dated 12 September 2016).

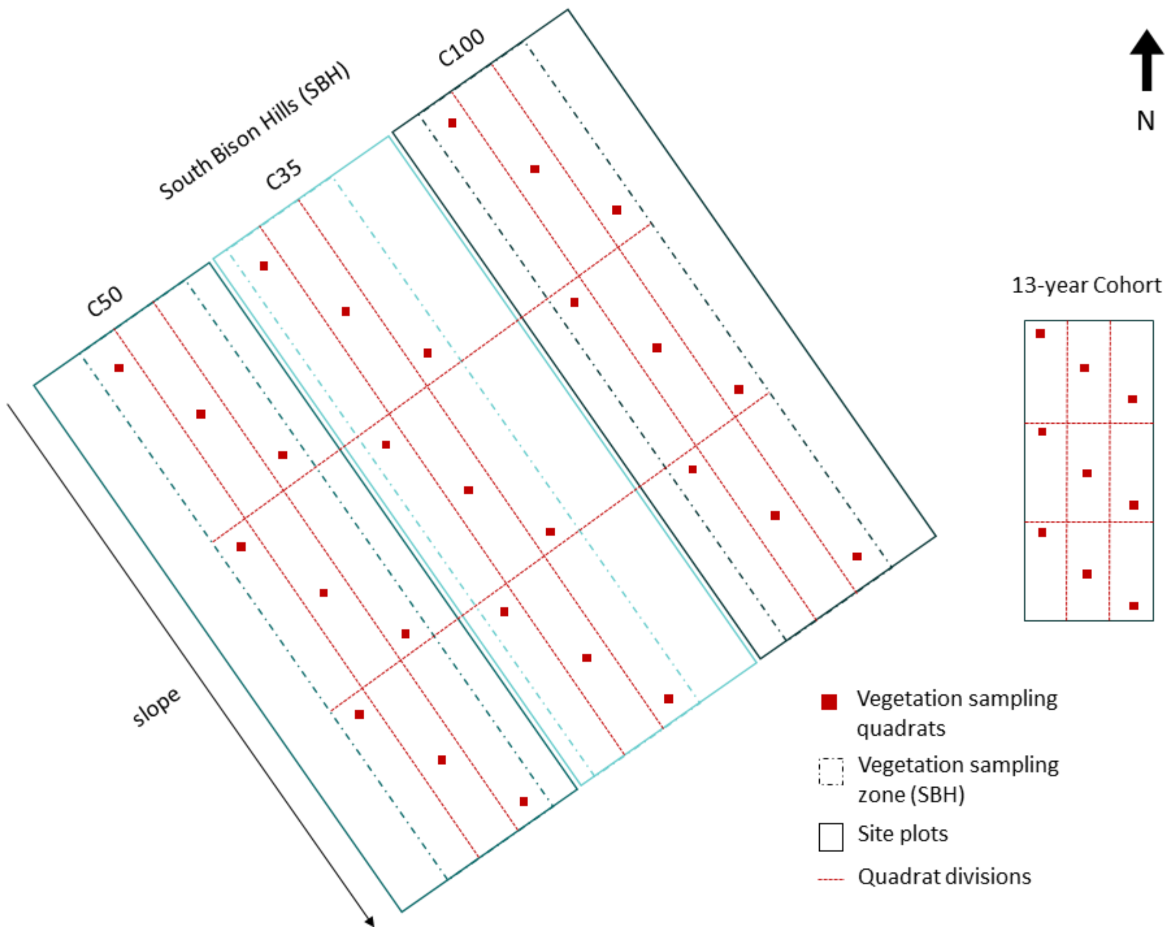

Figure 3.4 Overview of the vegetation sampling quadrat distribution in 2018, showing site plots and sampling quadrats for South Bison Hills Instrumented Watershed (SBH) and the 13-year cohort. For SBH, plots and quadrats followed the orientation of the treatment plots. Plots and quadrats (red squares) are roughly to scale (quadrats are 2 m by 2 m). Quadrat layout was designed to allow the quadrats to be evenly distributed at SBH (per the red, dotted lines), and was replicated at the new sites of the 13-year cohort. At SBH, this meant there were three quadrats in each slope position (top, middle, bottom) and of those three, one each corresponded to the east edge, centre, and west edge of the slope section. At the 13-year cohort, plots followed cardinal directions, which meant that the divisions became north-south orientation (north, centre, south) and east-west orientation (east edge, centre, west edge).

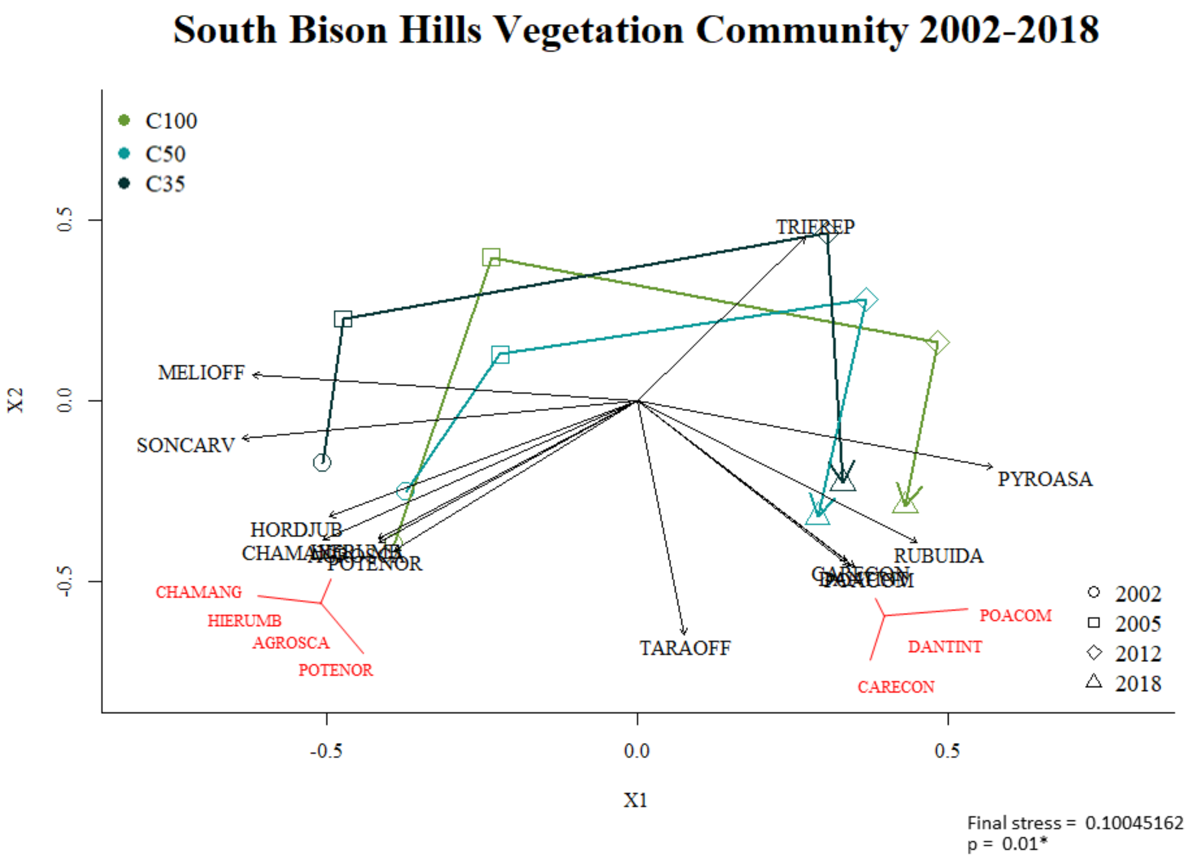

Figure 3.5 An nMDS output for the three treatments (C100, C50, and C35) at South Bison Hills Instrumented Watershed (SBH) over the four vegetation sampling years (2002, 2005, 2012, and 2018), illustrating vegetation community over time at SBH. Significant species (a=0.01) are displayed with their associated vectors and according to seven-letter species codes (Species codes are explained in Appendix B). Where species vectors overlapped, species names have been clarified in red.

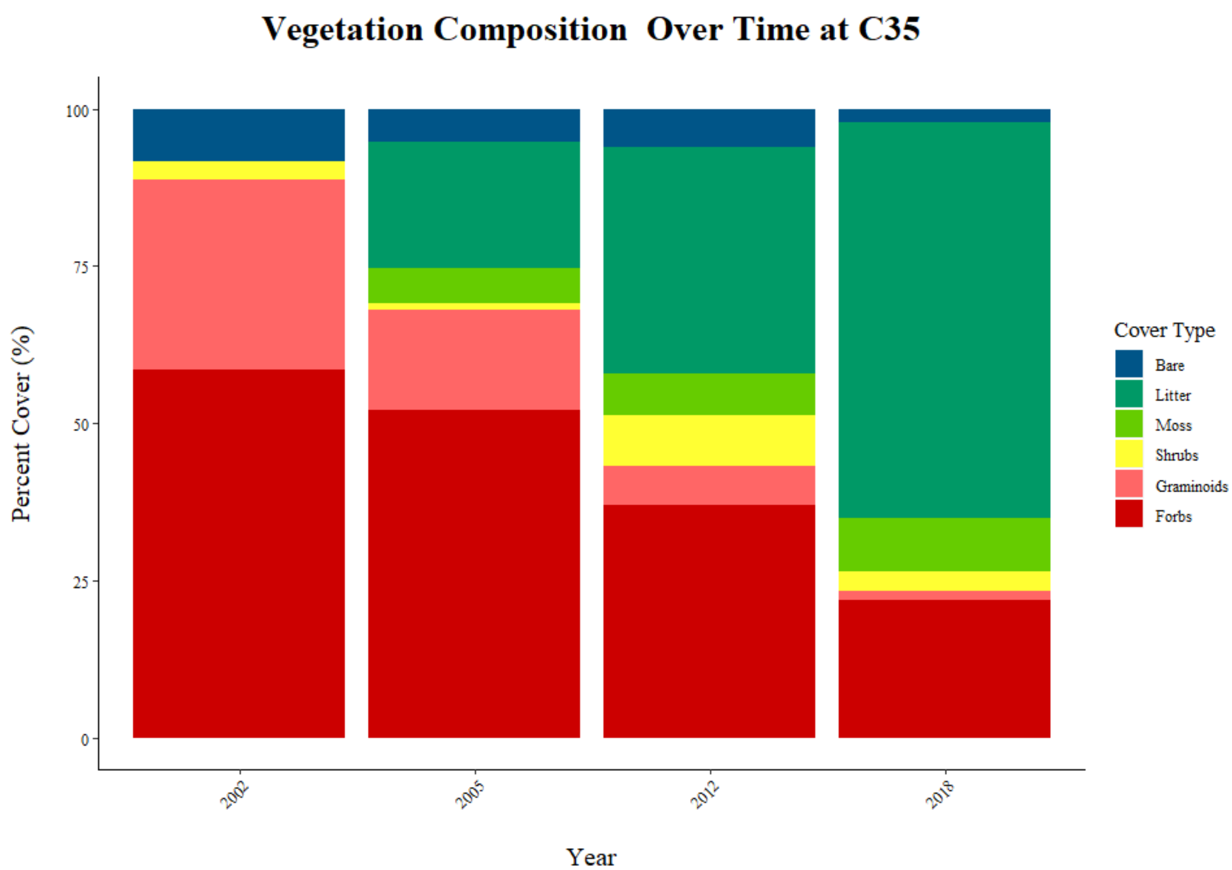

Figure 3.6 Vegetation community composition over time at South Bison Hills Instrumented Watershed (SBH) treatment C35 by vegetation type. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period (post-planting). The vegetation categories are shrubs, graminoids, and forbs (which includes ferns and fern allies). Three additional categories are also included, bare ground (bare), leaf litter (litter), and moss. Note: In 2002, litter and moss were not recorded (but were presumably low).

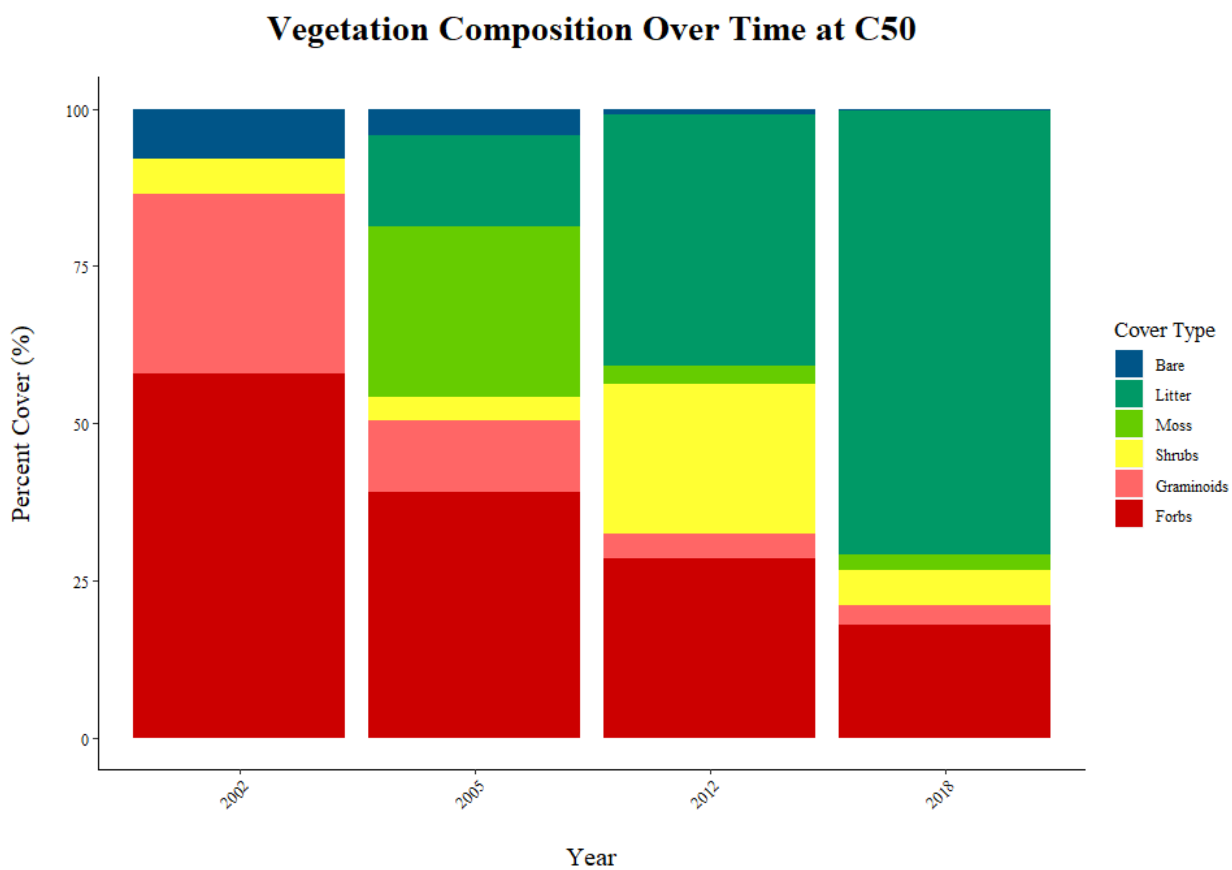

Figure 3.7 Vegetation community composition over time at South Bison Hills Instrumented Watershed (SBH) treatment C50 by vegetation type. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period (post-planting). The vegetation categories are shrubs, graminoids, and forbs (which includes ferns and fern allies). Three additional categories are also included, bare ground (bare), leaf litter (litter), and moss. The spike in moss in the year 2005 is likely due to one sampled quadrat having nearly half of its understory covered by moss. Note: In 2002, litter and moss were not recorded (but were presumably low).

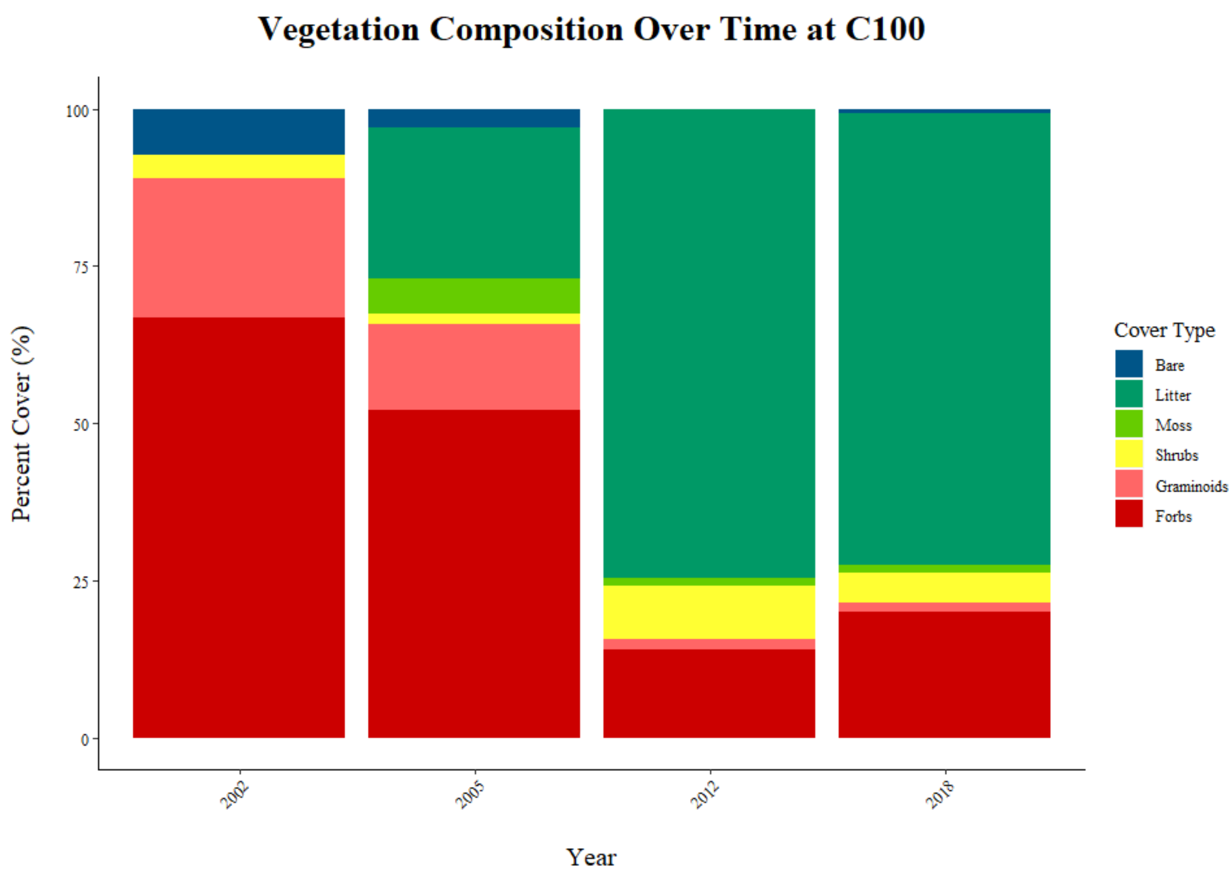

Figure 3.8 Vegetation community composition over time at South Bison Hills Instrumented Watershed (SBH) treatment C100 by vegetation type. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period (post-planting). The vegetation categories are shrubs, graminoids, and forbs (which includes ferns and fern allies). Three additional categories are also included, bare ground (bare), leaf litter (litter), and moss. Note: In 2002, litter and moss were not recorded (but were presumably low).

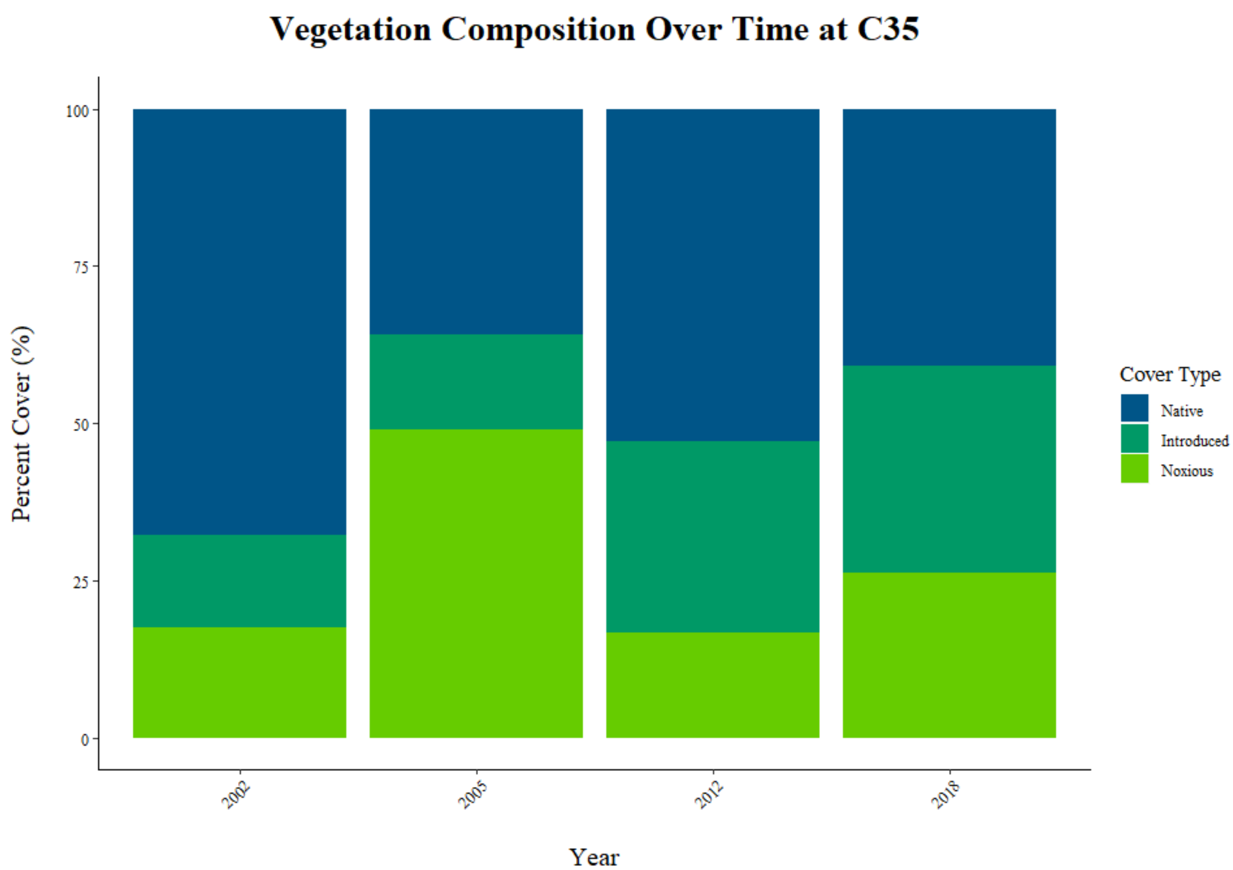

Figure 3.9 Breakdown of vegetation cover (shrubs, gramindoids, and forbs) by nativeness to Alberta, over time at South Bison Hills Instrumented Watershed (SBH) treatment C35. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period. The vegetation categories are native, introduced, and noxious weeds (noxious). Species nativeness was determined using the Alberta Conservation Information Management System (ACIMS) *List of Elements in Alberta -Vascular Plants* (2018). Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016).

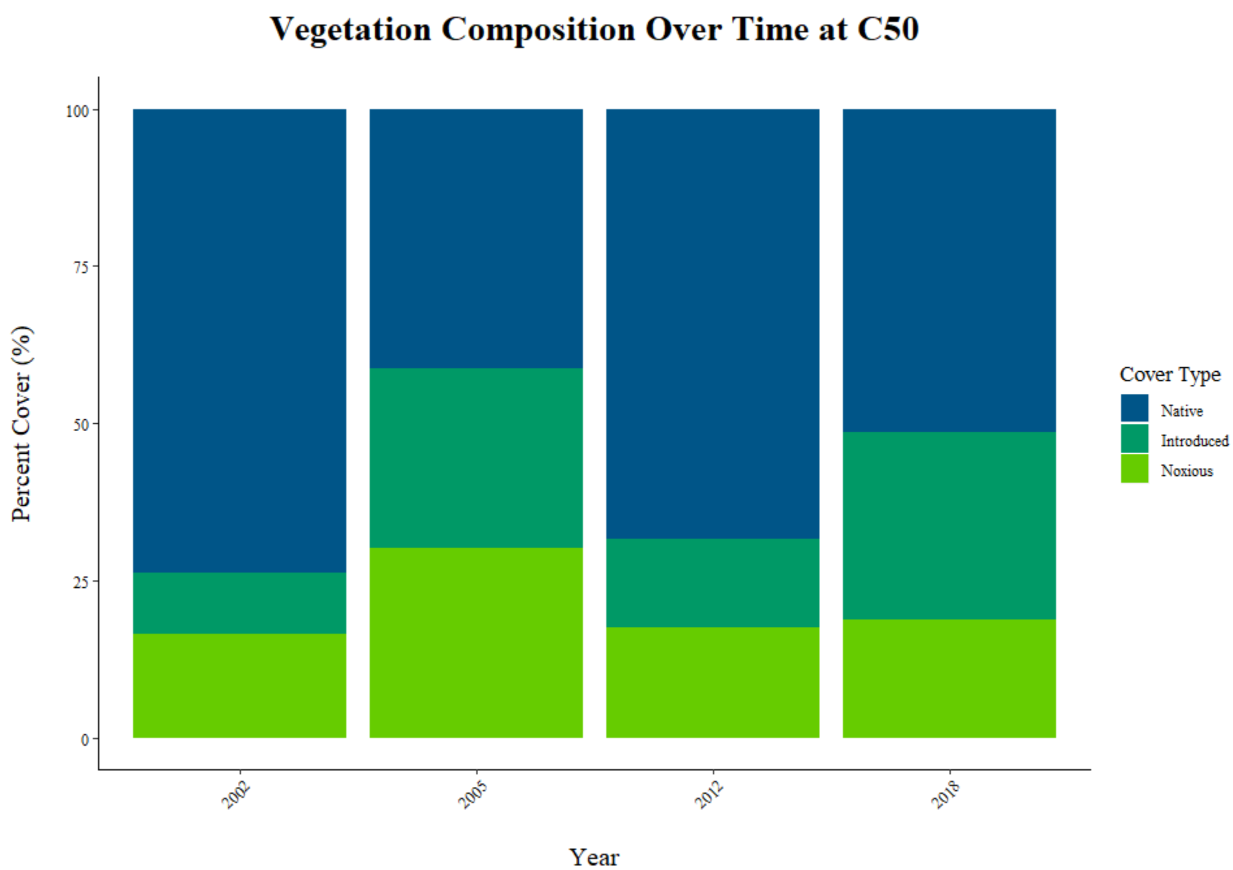

Figure 3.10 Breakdown of vegetation cover (shrubs, gramindoids, and forbs) by nativeness to Alberta, over time at South Bison Hills Instrumented Watershed (SBH) treatment C50. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period. The vegetation categories are native, introduced, and noxious weeds (noxious). Species nativeness was determined using the Alberta Conservation Information Management System (ACIMS) *List of Elements in Alberta -Vascular Plants* (2018). Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016).

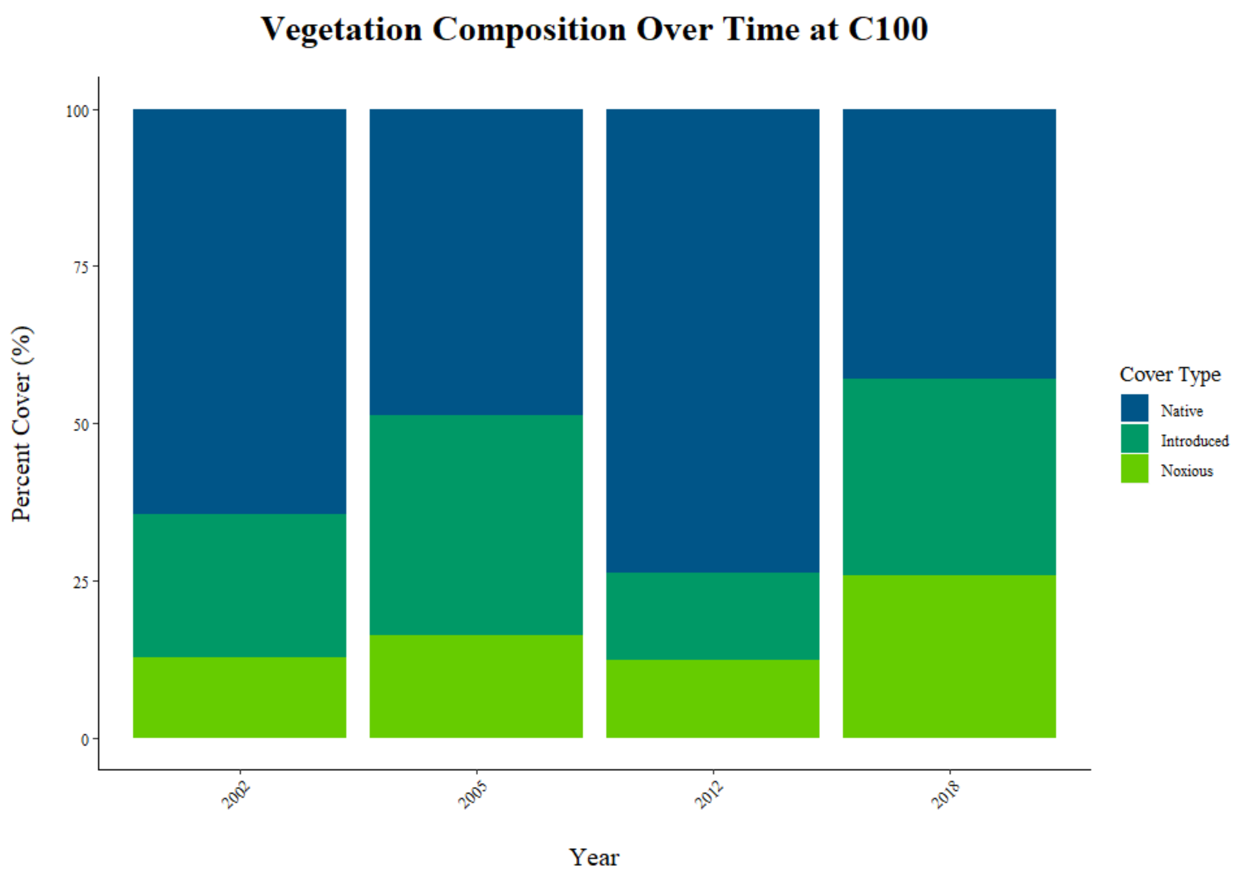

Figure 3.11 Breakdown of vegetation cover (shrubs, gramindoids, and forbs) by nativeness to Alberta, over time at South Bison Hills Instrumented Watershed (SBH) treatment C100. The four different sampling years (2002, 2005, 2012, and 2018) represent a 17-year sampling period. The vegetation categories are native, introduced, and noxious weeds (noxious). Species nativeness was determined using the Alberta Conservation Information Management System (ACIMS) *List of Elements in Alberta -Vascular Plants* (2018). Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016).

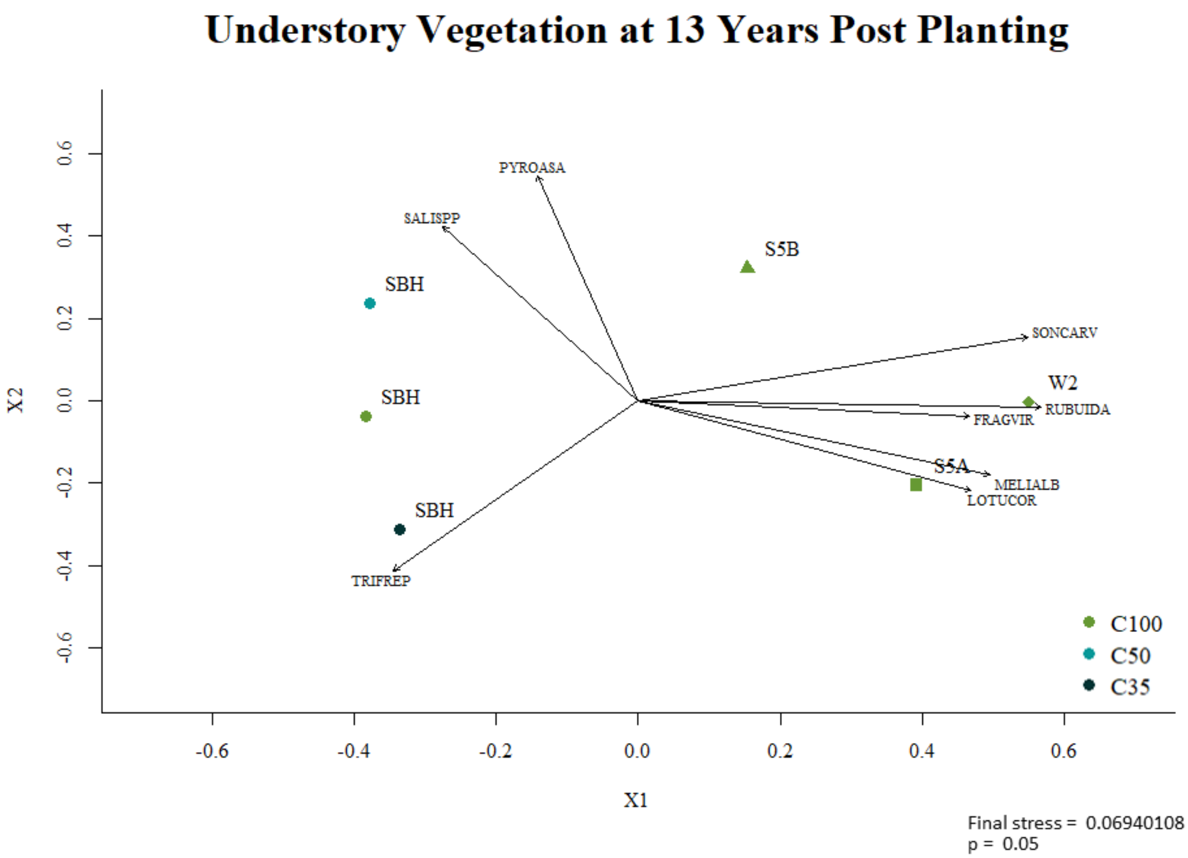

Figure 3.12 An nMDS output for the three treatments (C100, C50, and C35) at South Bison Hills Instrumented Watershed (SBH) and the three additional sites of the 13-year cohort, each with treatment C100. Data collected at SBH in 2012 is used alongside data collected in 2018 at S5A, S5B, and W2. Significant species vectors (a=0.05) are displayed with their associated vectors and according to seven-letter species codes. Data from 2012 were used for South Bison Hills Instrumented Watershed (SBH), and the data collected in 2018 were used for the three new sites. Significant species (a=0.01) are displayed with their associated vectors and according to seven-letter species codes (Species codes are explained in Appendix B).

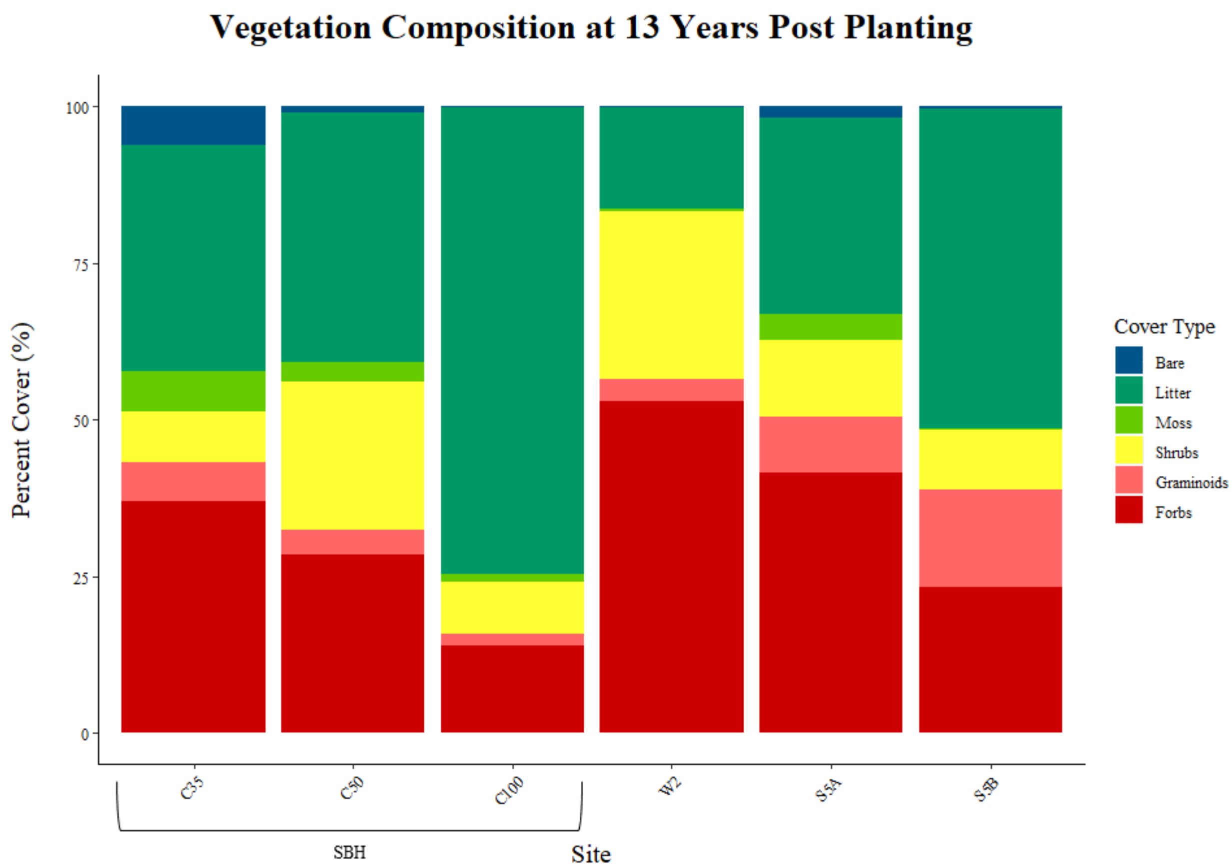

Figure 3.13 Vegetation community composition at South Bison Hills Instrumented Watershed (SBH) and the 13-year cohort by vegetation type. Data from 2012 were used for SBH, and data collected in 2018 were used for the sites W2, S5A, and S5B. The vegetation categories are shrubs, graminoids, and forbs (which includes ferns and fern allies). Three additional categories are also included, bare ground (bare), leaf litter (litter), and moss.

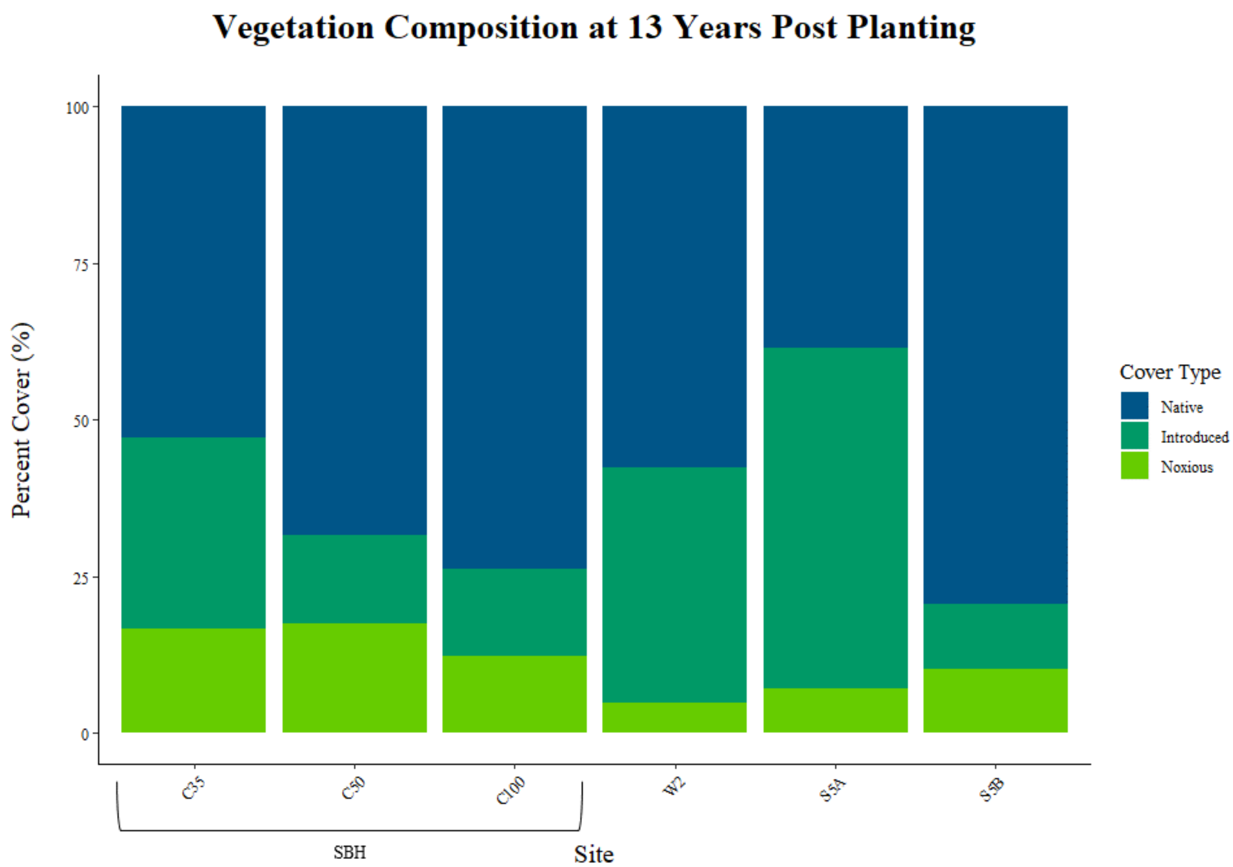

Figure 3.14 Breakdown of vegetation cover (shrubs, gramindoids, and forbs) by nativeness to Alberta, at South Bison Hills Instrumented Watershed (SBH) and the 13-year cohort. Data from 2012 were used for SBH, and data collected in 2018 were used for the sites W2, S5A, and S5B. The vegetation categories are native, introduced, and noxious weeds (noxious). Species nativeness was determined using the Alberta Conservation Information Management System (ACIMS) *List of Elements in Alberta -Vascular Plants* (2018). Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016).

Chapter 4: General Discussion and Conclusions

This study examined tree growth and understory plant community development in the longer-term (nine to 19 years) on reclaimed oil sands mining sites, underlain by Clearwater overburden, in the Athabasca Oil Sands Region (AOSR) of Alberta, Canada. At 19-years post-reclamation on the South Bison Hills (SBH) experimental site, aspen clearly grew better on the thickest capping treatment, C100 (20 cm of peat mineral mix over 80 cm of mineral soil), while spruce showed no differences in growth among the three capping depths (C35 [15 cm peat mineral mix over 20 cm of mineral soil], C50 [20 cm peat mineral mix over 30 cm of mineral soil], and C100). Results from a group of sites assessed at nine years post-reclamation showed no evidence that increasing the capping depth from 100 cm (C100) to 150 cm (C150, 30 cm of peat mineral mix over 120 cm of mineral soil) yielded added benefit to tree growth for either species. However, it is possible the high degree of site-to-site variability may have masked capping depth differences. Overall, site indices of these reclaimed sites were comparable to those expected for natural forests, indicating success of reclamation treatment.

At SBH, the understory plant community composition showed a clear temporal trajectory that was similar for all three of the capping treatments, indicating that the understory transitioned similarly regardless of capping depth. The high level of variation in understory communities among sites in the 13-year cohort indicates that treatment may not be the dominant factor determining understory development. The development of an unintended ecosite type at one of the study sites (ecosite b rather than the target d ecosite) further demonstrates this, while adding diversity to the reclamation landscape. The strong presence of weedy and noxious species is a concern at all sites sampled.

4.1 Impacts of Treatment on Tree Growth versus Understory Development: Competing Needs

This study demonstrated that while peat-mineral mix (PMM) capping treatments do promote successful growth of aspen and white spruce comparable to natural forests, the establishing understory communities are not similarly successful at approximating natural forests, within similar timelines. While forest overstory does influence understory (Chen et al., 2018; Lieffers & Stadt, 1994), more active intervention may be necessary to accelerate development of representative understory vegetation. In the years since these sites were reclaimed, changes to reclamation protocols and techniques have occurred, but implications for the understory development have been mixed. For instance, the use of forest floor material (FFM) within capping treatments instead of PMM has been found to increase the species diversity and number of native forest species in the understory (Mackenzie & Naeth, 2010; Hahn & Quideau, 2013; Errington & Pinno, 2015). FFM treatments have also been found to support higher percent cover of vegetation on reclamation sites as compared to what is achieved on sites capped with PMM material (Hahn & Quideau, 2013). In contrast, tree seedling establishment has been found to be more abundant on PMM soils than FFM, attributed to the higher soil moisture of PMM soils (Pinno & Errington, 2015), and early aspen growth is better on PMM soils (Schott et al., 2016). The use of fertilizer decreases natural tree seedling establishment (Pinno & Errington, 2015) but increases planted tree growth (Schott et al., 2016; Pinno et al., 2012) and increases weedy understory species (Buss, Stratechuk, & Pinno, 2018). In this study, the younger sites which had not received fertilizer tended to have better tree growth, but the trend was not very strong. Cover crops can benefit a natural understory by decreasing the prevalence of weedy species in some instances (Macdonald et al., 2015b), but they can decrease tree growth and survival through early competition (Franklin et al., 2012). Use of cover crops in reclamation in the AOSR was largely discontinued between the 13-year and 9-year cohorts.

4.2 Design Improvements and the Future of the Cohorts for Long-Term Study

The inclusion of additional sites when completing this thesis was used to combat the lack of replication in the long-term research site at SBH. This was a partial solution only, as additional sites existed for one of the treatments (C100) but not the other two (C35 and C50). Establishing multiple sites where all three treatments are present would allow for the three treatments to be properly compared with inferential statistics. It would also better control for site specific conditions that may skew results. Otherwise, it is impossible to truly conclude whether the C35 and C50 treatments actually result in lower aspen growth.

Including a wider range of capping treatments could also be useful. For instance, a 75 cm treatment, found in theory to optimize water resources (Syncrude Canada Ltd., 2013) is a good candidate for inclusion. In addition, treatments that alter the relative depths of peat and mineral layers while keeping the total depth constant could be used to better examine the role of these layers in growth outcomes. For example, this could be a set of C100 treatments with a variety of peat and mineral depths (e.g. 10 cm peat : 90 cm mineral, 20:80, 30:70, 40:60, 50:50).

There have also been changes to reclamation technique and standard procedure which must be considered. For instance, the older sites in this study (SBH and most of the 13-year cohort) were seeded with barley (Syncrude Canada Ltd., 2000, 2004), but the younger sites (9-year cohort) were not (Syncrude Canada Ltd., 2006, 2007, 2009). It would be useful to establish a new set of replicated capping study sites when major changes such as this take place. This would allow these sites to be compared with the older sites at similar ages, and evaluate whether these changes to practice have had a positive impact on growth.

The limitations of the original design of SBH do not, however, negate the value of the data collected from SBH and the various other reclamation sites for this thesis. Indeed, this should be seen as a long-term dataset that can continue to be contributed towards, especially as sites with newer protocols reach the ages of the 9 and 13-year cohorts, for example. Future studies can incorporate data from those sites, such as FFM capping treatments, with the data from the present study. In addition, increasing the sample size in general could help characterize the high site-to-site variation in outcomes noted for both tree growth and understory composition.

4.3 Balancing the Pros and Cons of Increasing or Decreasing Capping Depths

Capping materials are a limited resource on most mine sites, whether they are derived from recently-disturbed sites (Macdonald et al., 2015a; Macdonald, Quideau, & Landhäusser, 2012) or have been stored for long periods as a product of early mine development. Such limited resources must be used effectively and efficiently, and there are tradeoffs and costs which must be understood when choosing reclamation treatments at the site and landscape level. The intended outcomes of reclamation include a diversity of desired ecological states, generally represented by site-level ecosite targets and broader potential land uses (e.g. timber harvest, traditional use, recreation, etc.) (Alberta Environment and Parks, 2010). Choosing thicker capping treatments such as C150 may at first seem like the best solution to increase the chances of reclamation success and reduce the risk of salt exposure, but such thick treatments inherently raise the costs of reclamation, and may bring unintended consequences. For example, the change in regulatory requirement from a 100 cm to 150 cm capping treatment over Clearwater overburden that was required at Syncrude for a number of years has been estimated to cost in the hundreds of millions (B. G. Purdy, personal communication). The results of this thesis did not, at nine years post-reclamation, point to an increase in tree growth as a product of increased capping depth (from 100 cm to 150 cm specifically). Other research has indicated such an increase would dramatically decrease the water available to downstream wetlands as would instead be stored in the soil profile of upland sites (Syncrude Canada Ltd., 2013). The reclaimed landscape should be designed to allow the development of a variety of ecosites, including both upland and lowland ecosites. Considered use of capping depth has great potential to contribute to this diversity. Thus, being derived from limited resources, the results indicate investment in C150 may not be necessary on Cleatwater overburden.

4.4 Reclamation Certification and the Use of Ecosite in Determination of Success

Reclamation certification is the end goal of reclamation activities, yet detailed criteria for this certification are lacking. Existing public documentation describes how to establish reclamation targets, and how, in broad terms, to develop reclamation sites with those targets in mind (Alberta Environment and Parks, 2010). It also establishes targets relative to ecosite type and site index expectations that reclamation sites must meet to be considered reclaimed (Alberta Environment and Parks, 2010). Currently, only one site has received reclamation certification (Atkinson, 2017), and most of the currently available documentation on the subject is still only a guideline. It thus remains unclear where thresholds and required trajectories lie between successful and unsuccessful reclamation. While the objective of reclamation is stated in provincial regulation as a return of the land to equivalent capability (Alberta Environment and Parks, 2010), the link between equivalent capability, reclamation and closure plans, and reclamation outcomes needs to be better quantified.

In general, the concept of equivalent capability does not lend itself to specific quantitative targets, though existing documentation on the subject attempts to do so (Alberta Environment and Parks, 2010). Resemblance to a natural ecosite, assessed quantitatively in terms of the number of characteristic species of a given ecosite identified on a reclamation site, is one such metric. However, the overall usefulness of ecosite types as reclamation targets is also worth addressing. These ecosite categories were developed based on natural forest ecosystems (Beckingham & Archibald, 1996). Yet, reclamation sites, with their even-aged stands of planted trees, more closely resemble plantation forests. For instance, in natural forests, white spruce tends to establish in the understory of mature aspen (Lieffers & Stadt, 1994). However, on reclamation sites such as those in this thesis, aspen and white spruce seedlings of the same age are planted at the same time. In this way, even from establishment, reclaimed systems will have vastly different growth patterns from natural sites. Another concern is that in richer sites in natural forests, higher competition from understory vegetation slows the early growth of trees (Beckingham & Archibald, 1996). While this is expected to balance out with increased tree growth rates later-on and thus high overall SI values (Beckingham & Archibald, 1996), using one of these richer ecosite types as a target could skew expectations for tree growth. Broad site index targets have also been defined by the existing documentation (Alberta Environment and Parks, 2010), but I would also argue that the site index targets are vague and unrealistic. For instance, the minimum SI values for aspen and white spruce (11.6 and 7.1, respectively) are well below the expected mean site indices of the targeted ecosites (Beckingham & Archibald, 1996). Notably, they are also well below the values for the reclamation sites I sampled, especially for white spruce (aspen: 13.8-21.0, white spruce: 14.2-20.7).

Perhaps equivalent capability should not be taken to mean that a site must resemble a type of natural site already found in the boreal forest. Instead, criteria could be established that directly asses capability. Equivalent capability means that a site can support a similar variety of land uses as previously (Alberta Environment and Parks, 2010). Perhaps, an evaluation of the land uses a reclaimed site can offer would be a more direct approach to evaluating reclamation success. For instance, presence at a site of healthy wildlife, especially wildlife that is able to meet its needs for all stages of life on the reclaimed site, could be a useful measure of the integration of reclaimed sites into the broader landscape and ecosystem.

Regardless of the approach taken, it is clear that more relevant and measurable criteria are needed. Long-term studies, such as the research presented in this thesis, can be useful in establishing what a normal progression for reclamation sites might be, and can in turn be used to determine targets for reclamation sites to meet.

4.5 Conclusions

Reclamation success is a delicate balance. Understory and overstory success may be determined by competing factors. Evaluation of success lacks defined criteria, and is thus difficult to accomplish. Expanding and improving upon current research and data collection could improve our ability to make decisive conclusions about which treatment provides the best reclamation value, and help develop the criteria needed to accurately measure reclamation success.

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Appendix A: Sites and Naming

Table A-1 List of sites sampled in 2018, and their names according to this thesis, reclamation ID (Syncrude C&R 2016 geodatabase, Government of Alberta, 2018), and Syncrude Canada Ltd. mining areas. Cohort is also reported, as well as size of the reclamation site, and treatments therein. Note: all cohorts were included in Chapter 2 but the 9-year cohort was not included in Chapter 3.

Site Name	Reclamation ID	Syncrude Area	Cohort	Size (ha)	Capping Treatment
SBH	346	SBH/South Bison Hills Instrumented Watershed	All	2.5	C35, C50, C100
W2	474	W2	13-year	97.8	C100
S5A	467	S5	13-year	3.3	C100
S5B	466	S5	13-year	4.1	C100
W1	603	W1	9-year	16.9	C100
575A	575	4C Island	9-year	1.7	C100
575B	575	46 Dump	9-year	23.0	C150
574	574	46 Dump	9-year	5.9	C150
570	570	4C Littoral	9-year	7.4	C150

Appendix B: Species List from Vegetation Surveys

Table B-1 List of plant species found during the understory vegetation surveys, conducted in 2002, 2005, 2012, and 2018. Species codes were formed using the first four letters of the genus name, combined with the first three letters of the species name. Nomenclature follows Moss (1983) with updated names taken from The Alberta Conservation Information Management System (ACIMS) *2015 List of Elements in Alberta – Vascular Plants*. Provincial status (circa 2015) and common names were also taken from the 2015 ACIMS list. Noxious status was determined according to the Weed Control Act (Government of Alberta, 2016). See Appendix A for a description of the site name codes used to indicate in which sites species were present.

Species Code	**Latin Name (Flora of AB)**	**Latin Name (ACIMS 2015)**	**Common Name**	**Sites Found**	**Provincial Status (ACIMS 2015)**	**Notes**
ACHIMIL	*Achillea millefolium* L.		common yarrow	All sites	S5	
ACHIALP	*Achillea sibirica* Ledeb.	*Achillea alpina* L.	many-flowered yarrow	SBH (C35 & C50)	S5	
AGROSCA	*Agrostis scabra* Willd.		rough hair grass	SBH (all), W2	S5	
ALNUVIR	*Alnus crispa* (Aiton) Pursh	*Alnus viridis* (Chaix) DC.	green alder	W2	S5	
AMELALN	*Amelanchier alnifolia* (Nutt.) Nutt. ex M. Roem.		saskatoon	SBH (C100), S5A	S5	
ARCTUVA	*Arctostaphylos uva-ursi* (L.) Spreng.		common bearberry/ kinnikinnick	SBH (C50 & C100), W2	S5	
ASTRCIC	*Astragalus cicer* L.		cicer milk vetch	SBH (C35)	Exotic	
BETUPAP	*Betula papyrifera* Marshall		white birch	SBH (C35), S5A, S5B	S5	
BOTRCRE	*Botrychium dusenii* auct. non (Christ) Alston	*Botrychium crenulatum* W.H. Wagner	scalloped grapefern	S5B	**S3**	
BROMCIL	*Bromus ciliatus* L.		fringed brome	W2	S5	
BROMINE	*Bromus inermis* Leyss. *ssp. inermis*	*Bromus inermis Leyss.*	smooth brome	SBH (C35 & C50), S5A	Exotic	
BROMPUM	*Bromus inermis Leyss. ssp. pumpellianus Scribn.*	*Bromus pumpellianus*Scribn.	Pumpelly brome	SBH (C50)	S5	
CALACAN	*Calamagrostis canadensis* (Michx.) P. Beauv.		bluejoint	SBH (all), W2, S5A	S5	

Species Code	Latin Name (Flora of AB)	Latin Name (ACIMS 2015)	Common Name	Sites Found	Provincial Status (ACIMS 2015)	Notes
CAREAQU	*Carex aquatilis* Wahlenb.		water sedge	W2, S5B	S5	
CAREATH	*Carex atherodes*Spreng.		awned sedge	S5A	S5	
CAREBRU	*Carex brunnescens* (Pers.) Poir.		brownish sedge	S5A	**S4**	
CARECON	*Carex concinna* R. Br.		beautiful sedge	SBH (all), W2	S5	
CAREPRA	*Carex praticola* Rydb.		meadow sedge	SBH (all)	S5	
CARESAR	*Carex sartwellii* Dewey		Sartwell's sedge	SBH (C100)	**S4**	
CARESIC	*Carex siccata* Dewey		hay sedge	SBH (all)	S5	
CARESPP	*Carex spp.*		unknown sedge	SBH (all)	N/A	
CAREUTR	*Carex utriculata* Boott		small bottle sedge	W2	S5	
CHAMANG	*Epilobium angustifolium* L.	*Chamerion angustifolium* (L.) Holub	common fireweed	All sites	S5	
CIRSARV	*Cirsium arvense* (L.) Scop.		creeping thistle	SBH (all), S5A, S5B	Exotic, noxious	
CORNSTO	*Cornus stolonifera* Michx.		red-osier dogwood	SBH (C50), S5B	S5	
DANTINT	*Danthonis intermedia* Vasey included under *Danthonia californica* Boland	*Danthonia intermedia* Vasey	intermediate oat grass	SBH (all)	S5	*D. intermedia* reported
ELYMREP	*Agropyron repens* (L.) P. Beauv.	*Elymus repens* (L.) Gould	quackgrass	S5A	Exotic	
ELYMTRA	*Agropyron trachycaulum* (Link) Malte ex H.F. Lewis	*Elymus trachycaulus* (Link) Gould ex Shinners	slender wheatgrass	SBH (all), W2, S5B	S5	
EPILPAL	*Epilobium palustre* L.		marsh willowherb	SBH (C50 & C100)	**S4**	
EQUIARV	*Equisetum arvense* L.		common horsetail	All sites	S5	
EQUIPRA	*Equisetum pratense* Ehrh.		meadow horsetail	SBH (C100)	S5	
EURYCON	*Aster conspicuus* Lindl.	*Eurybia conspicua* (Lindl.) G.L. Nesom	showy aster	SBH (C100)	S5	
FRAGVIR	*Fragaria virginiana* Duchesne		wild strawberry	All sites	S5	
GALIBOR	*Galium boreale* L.		northern bedstraw	S5B	S5	

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HIERUMB	*Hieracium umbellatum* L.		narrow-leaved hawkweed	SBH (all), S5B	S5	
HORDJUB	*Hordeum jubatum* L.		foxtail barley	SBH (all), W2	S5	
LATHOCH	*Lathyrus ochroleucus* Hook.		cream-colored vetchling	SBH (C35 & C100), W2	S5	
LOTUCOR	*Lotus corniculatus* L.		bird's-foot trefoil	SBH (C35 & C50), W2, S5A	Exotic	
LUZUSPP	*Luzula spp.*		unknown wood-rush	SBH (C35 & C50)	N/A	
MEDISAT	*Medicago sativa* L.		alfalfa	SBH (all), S5A	Exotic	
MELIALB	*Melilotus alba* Desr.		white sweet-clover	All sites	Exotic	
MELIOFF	*Melilotus officinalis* (L.) Lam.		yellow sweet-clover	All sites	Exotic	
MERTPAN	*Mertensia paniculata* (Aiton) G. Don		tall lungwort	SBH (C50)	S5	
MOEHLAT	*Moehringia lateriflora* (L.) Fenzl		blunt-leaved sandwort	SBH (C50)	S5	
ORTHSEC	*Orthilia secunda* (L.) House		one-sided wintergreen	SBH (all)	S5	
PHLEPRA	*Phleum pratense* L.		timothy	SBH (C100)	Exotic	
PICEGLA	*Picea glauca* (Moench) Voss		white spruce	All sites	S5	
PANTMAR	*Plantago maritima L.*		sea-side plantain	SBH (C100)	**S1**	Found in 2002 & 2005, suspected incorrect ID
POAARI	*Poa arida* Vasey		plains bluegrass	S5A, S5B	**S4**	
POACOM	*Poa compressa* L.		Canada bluegrass	SBH (all), S5A, S5B	Exotic	
POAPAL	*Poa palustris* L.		fowl bluegrass	SBH (all), W2	S5	
POAPRA	*Poa pratensis* L.		Kentucky bluegrass	SBH (all)	S5	
POASPP	*Poa spp.*		unknown bluegrass	SBH (C35 & C50)	N/A	
POPUBAL	*Populus balsamifera* L.		balsam poplar	SBH (all)	S5	
POPUTRE	*Populus tremuloides* Michx.		trembling aspen	All sites	S5	
POTENOR	*Potentilla norvegica* L.		rough cinquefoil	SBH (all)	S5	
PRUNVIR	*Prunus virginiana* L.		choke cherry	SBH (C100)	S5	

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PYROASA	*Pyrola asarifolia Michx. (& Pyrola bracteata* Hook.)	*Pyrola asarifolia Michx.*	common pink wintergreen	SBH (C100), S5B	S5	Only *P. asarifolia* reported
RHINMIN	*Rhinanthus minor* L.		yellow rattle	SBH (C50 & C100)	**S4**	
RIBEOXY	*Ribes oxyacanthoides* L.		northern gooseberry	SBH (C50)	S5	
ROSAACI	*Rosa acicularis* Lindl.		prickly rose	SBH (C35 & C50), W2, S5A, S5B	S5	
RUBUIDA	*Rubus idaeus* L.		wild red raspberry	All sites	S5	
RUBUPUB	*Rubus pubescens* Raf.		dewberry	SBH (C50 & C100)	S5	
SALIBEB	*Salix bebbiana* Sarg.		beaked willow	All sites	S5	
SALIPET	*Salix petiolaris* Sm.		basket willow	S5A	S5	
SALIPLA	*Salix planifolia* Pursh		flat-leaved willow	W2, S5B	S5	
SALISPP	*Salix spp.*		unknown willow	SBH (all)	N/A	
SHEPCAN	*Shepherdia canadensis* (L.) Nutt.		Canada buffaloberry	SBH (C50 & C100), W2, S5A, S5B	S5	
SOLILEP	*Solidago canadensis* L.	*Solidago lepida*DC.	elegant goldenrod	All sites	**S4**	
SONCARV	*Sonchus arvensis* L. & *Sonchus uliginosus* M. Bieb.	*Sonchus arvensis* L.	perennial sow-thistle	All sites	Exotic, noxious	Both *S. arvensis and S. uliginosis reported*
STELLON	*Stellaria longipes* Goldie *(& Stellaria arenicola* Raup)	*Stellaria longipes* Goldie	long-stalked chickweed	SBH (all)	S5	Only *S. longipes* reported
SYMPCIL	*Aster ciliolatus* Lindl.	*Symphyotrichum ciliolatum* (Lindl.) Á. Löve & D. Löve	Lindley's aster	All sites	S5	
SYMPLAE	*Aster laevis* L.	*Symphyotrichum laeve* (L.) Á. Löve & D. Löve	smooth aster	SBH (C35 & C50), S5A, S5B	S5	
TARAOFF	*Taraxacum officinale* F.H. Wigg.		common dandelion	All sites	Exotic	
TRIEBOR	*Trientalis borealis* Raf.		northern starflower	SBH (C50 & C100)	**S4**	
TRIFHYB	*Trifolium hybridum* L.		alsike clover	SBH (all), W2, S5A	Exotic	
TRIFPRA	*Trifolium pratense* L.		red clover	SBH (all), S5A	Exotic	
TRIFREP	*Trifolium repens* L.		white clover	SBH (all)	Exotic	

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VACCMYR	*Vaccinium myrtilloides* Michx.		common blueberry	SBH (C100)	S5	
VACCVIT	*Vaccinium vitis-idaea* L.		bog cranberry	SBH (C50)	S5	
VICIAME	*Vicia americana* Muhl. ex Willd.		wild vetch	All sites	S5	