Competitive Relationships in Forest Restoration: Impact of Cover Crops and Fertilization on Tree and Understory Development

by

Alia Erin Ksenia Snively

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Department of Renewable Resources
University of Alberta

Abstract

Initial reclamation practices affect the early development and future recovery trajectories of tree and understory species on reclamation sites. In this thesis I explored the effects of yellow sweet clover (*Melilotus officinalis*) as a cover crop on tree performance and competition control in forest floor material which was salvaged and directly placed at two different depths (15 cm and 40 cm). Sweet clover supressed the establishment of some competing species; however total vegetation cover was not impacted and total mortality of planted *Populus tremuloides*, *Pinus contorta*, and *Picea glauca* seedlings was greater in sweet clover plots. Tree seedlings had better annual growth and lower mortality in the 40 cm salvage and placement depth treatment, potentially due to a more diluted seed bank which decreased dominance of competitive species. In a second study, I assessed the impact of four fertilizer treatments (control, 250 kg/ha immediately available fertilizer (IAF), 500 kg/ha IAF, and 670 kg/ha controlled release fertilizer) applied to two different capping materials (forest floor material (FFM) and a peat-mineral mix (PMM)) on initial forest vegetation development. The application of fertilizer did not affect average species richness in either capping material; however fertilization promoted increased cover of grasses in the FFM. Over time the number of annuals/biennials as well as non-native species decreased in both capping materials and the number of desirable forest understory species increased in the FFM.

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Table of Contents

List of Tables

List of Figures

Chapter 1: Introduction

1.1 Reclamation

Reclamation has become a common practice throughout the boreal forest of Canada as a result of extensive surface mining and other forms of natural resource extraction, as well as the ever increasing public pressure to restore these areas. Globally, the boreal forest exists as a circumpolar ring across the northern hemisphere with roughly one third of its area existing in Canada. In Alberta, the boreal forest covers 381,000 km^2 of land in the north or roughly 58% of the province (Natural Regions Committee 2006). This has provided the unique opportunity for research of various reclamation techniques as a result of landscape-scale disturbances incurred from surface mining bitumen and coal deposits in Alberta. In the Athabasca oil sands region of Alberta alone, there is 4,800 km^2 (480,000 ha) of surface mineable land and approximately 767 km^2 (76,700 ha) has already been disturbed (Alberta Government 2013). This area and all other areas disturbed by industrial activity are required by government regulation to be reclaimed. For Alberta, this means that subsequent to resource extraction, developers must return land "to the state of natural productivity that existed previous to the start of industrial activity" (Alberta Government 2013). In the boreal, "natural productivity" refers to a self-sustaining indigenous forest ecosystem which is capable of supporting wildlife habitat, timber production, and watershed functions (OSVRC 1998).

The majority of surface mining in Alberta occurs within the Boreal Forest Natural Region (Natural Regions Committee 2006). Within this region there are eight Natural Subregions which share some similar characteristics; however, revegetation goals should be tailored to the specific Natural Subregion where the disturbance occurs (Natural Regions Committee 2006). In general, the Boreal Forest Natural Region has short summers which receive the most precipitation, whereas winters are long and cold. The landscape ranges from level to undulating and is interspersed with wetlands. Common upland vegetation includes deciduous, mixedwood, and coniferous forests with Luvisolic soils. The most common tree species are *Populus tremuloides* Michx., *Populus balsamifera* L., *Picea glauca* (Moench) Voss, *Picea mariana* (Mill.) Britton, Sterns & Poggenb., and *Pinus banksiana* Lamb. Lowland areas are *P. mariana*, shrub, or sedge fens and marshes with Mesisolic soils (Natural Regions Committee 2006).

The process of surface mining poses unique challenges to reclamation because the soil is excavated as opposed to being left intact following a disturbance such as in-situ resource extraction, logging, or fire. Disturbances which leave the soil intact and only involve the removal of vegetation can affect hydrology, nutrient availability, and vegetation establishment, whereas the process of excavating deep soil pits additionally impacts soil properties (Franklin et al. 2012). These include abrupt changes to soil structure, chemistry, and the propagule bank which results in a longer recovery period (Alberta Environment and Water 2011; Franklin et al. 2012; Mackenzie and Naeth 2010). Some of the changes in soil structure and chemistry include elevated nitrogen levels (Rowland et al. 2009), increased alkalinity, higher bulk density from heavy machinery traffic, and reduced soil moisture along with raised soil temperatures due to the absence of a forest canopy (McMillan et al. 2007). Additionally, surface mining affects nutrient cycling and hydrology on a landscape level. The changes in soil properties, along with altered nutrient and water cycling, slow the rate of natural recovery and affect our ability to establish forest species on reclamation sites. Establishing the appropriate vegetation is perhaps the most challenging step and can be influenced by the treatment of soil during and after excavation.

Surface mining coal and bitumen deposits is performed by first removing all vegetation from a site and then stripping the soil in three layers. This includes approximately 30 cm of topsoil (organic layer/A and B horizons), up to 3 m of subsoil (C horizon), and the remaining material (overburden) up to a depth of approximately 75 m (Alberta Environment and Water 2011). Based on the topographic location of the salvaged soil, topsoil can either be mineral soil, peat, or some combination of the two. Topsoil from upland sites, which contains organic forest floor materials (L-F-H horizons) and the underlying A and B mineral horizons, is commonly referred to as forest floor material (FFM). Topsoil which contains peat is often referred to as a peat-mineral mix (PMM) and consists of a mixture of lowland peat and underlying mineral soil or transitional mineral soil adjacent to lowland areas. In comparison to PMM, field studies using FFM show higher species richness, plant abundance, and percent cover (Mackenzie and Naeth 2010; Naeth et al. 2013). In particular, the FFM generally has greater richness and abundance of native shrub species, upland forest understory species, and reduced cover of non-native species (Naeth et al. 2013). Additionally, microbial activity and nutrient reserves, specifically soluble potassium and available phosphorus, are commonly greater in the FFM (McMillan et al. 2007; Pinno et al. 2012). Greater microbial activity has been related to a relatively lower pH in the

FFM and it has been proposed that FFM may act as a better inoculant for mycorrhizae (McMillan et al. 2007). As a result the PMM is usually a less desirable capping material; however, it is more abundant in areas of the oil sands region which can be surface mined. Also, there are benefits to using PMM such as lower bulk density and increased organic matter, nitrogen, total carbon, cation exchange capacity, and water holding capacity (Mackenzie and Naeth 2010).

Once soil has been stripped, there are currently two options for handling the topsoil and subsoil. One option would be transporting soil to a nearby area and dumping it in two separate stockpiles until it is required for reclamation. The other option is that topsoil and subsoil can be transported directly to a reclamation site and placed without the stockpiling phase. The main benefit of the direct transfer technique, compared to stockpiling, is that there is less of a detrimental impact on nutrient availability and the viability of the propagule bank, which leads to higher species establishment on reclamation sites and reduced reclamation costs (Mackenzie and Naeth 2010; Koch et al. 1996; Rokich et al. 2000). Reduced handling helps maintain soil structure and preserve quality because, without the stockpiling phase, nutrient leaching is decreased and organic matter, propagules, and soil biota are conserved (Alberta Environment and Water 2011). The ability to maintain a viable propagule bank through direct soil transfer has been linked to initially higher species establishment on reclamation sites (Mackenzie and Naeth 2010; Fair 2011). This is beneficial because one of the main challenges in reclamation is to establish greater biodiversity, which includes native understory species, without having to perform seeding or planting (OSVRC 1998).

The direct placement of soil has proven to be superior for native understory species establishment; however, the exact depth at which soil is placed to receive optimal results remains unclear. Within the boreal forest of Alberta, experimental soil salvage and placement depths have ranged from 10 cm to 40 cm with the greatest density of establishment occurring at approximately 20 cm (Mackenzie and Naeth 2010; Fair 2011). Due to the use of large scale machinery, it is believed that at shallow salvage depths (approximately 10 cm) the seed bank can be diluted as a result of admixing between the topsoil and subsoil during soil placement (Mackenzie and Naeth 2010). At greater salvage depths (approximately 40 cm), dilution is even greater due to increased mineral soil content and handling (Fair 2011). As a result, first year establishment is influenced by the placement depth of soil with relatively shallower depths typically having greater establishment. While this is often desirable, increased competition from herbaceous vegetation at shallower depths can be detrimental to planted seedlings and competition control may be required.

Direct placement of soil is the most ideal practice for achieving long term reclamation goals; however, it is not always operationally feasible as surface mining often occurs at a faster rate than reclamation, hauling the soil can be time consuming, and there has not always been available space for permanent reclamation (Alberta Environment and Water 2011). As a result, reclamation can also involve "reconstructing" soil by returning the three excavated layers back to their proper order following a period of stockpiling. First, the overburden material is used as a fill to shape the desired contours of the new landscape and then approximately 80 cm of subsoil which is suitable for plant root growth is placed on top. Finally, the area is covered with FFM or PMM topsoil at a depth of 20-50 cm (Rowland et al. 2009; Macdonald et al. 2012). Topsoil is also referred to as "capping material" because it is the top layer which covers a reclamation site and it is the most suitable material for plant growth. Although it is the most suitable material, there are often problems with initial nutrient availability, ground cover, water retention, and the establishment of ruderal species as a result of stockpiling topsoil. Disrupted soil function and the lack of a desirable propagule bank and/or seed source following soil placement can hinder natural restoration on mine sites (Strong 2000). Therefore, in an attempt to accelerate forest restoration, it is common to apply amendments such as cover crops and fertilizer to reclamation sites along with planting trees and, more recently, shrubs.

1.2 Forest initiation and cover crops

Early tree growth and survival on reclamation sites is important because the ability for trees to establish canopy closure has been linked to faster development of a forest plant community (Maundrell and Hawkins 2004; Strong 2000). Natural forest regeneration begins with relatively quick growing tree species which are early successional, such as *Populus tremuloides* and *Pinus contorta*. These trees are usually the first to establish and create a canopy which changes herbaceous understory species composition by excluding species which are not shade tolerant. In contrast, slow growing, long lived, and shade tolerant trees, such as *Picea glauca*, persist in the community until eventually becoming the dominant species. Therefore, in order to "build" a forest on a reclamation site, the successional stages of a forest must be considered when choosing which trees to plant and at what density. Different tree species will respond differently on reclamation sites as a result of their growth patterns, shade tolerance, and general health and stature at the time of planting. These characteristics, among others, affect a tree seedlings' ability to survive and cope with competition.

On reclamation sites, tree planting is necessary for faster development of a forest; however, this is only one step on the way to ecosystem recovery. Once trees are planted there are many environmental factors which can influence their establishment, growth, and survival. These include the initially variable nutrient conditions, pH levels, possible salinity and compaction, competing vegetation, and exposure to harsh weather events (Casselman et al. 2006; Franklin et al. 2012; Pinno et al. 2012). In particular, initial seedling mortality and poor growth is strongly influenced by competition for light, water, and nutrients from surrounding herbaceous vegetation (Franklin et al. 2012). As a result, seedling quality and competition control are often the most important factors for promoting seedling performance and maintaining the trajectory of a recovering ecosystem. In the first year following soil placement there is often relatively little herbaceous cover, if fertilizer is not applied, and fast growing, disturbance adapted, annual species thrive and dominate (Macdonald et al. 2012). While low cover results in less competition for planted seedlings, depending on its structure, early vegetation cover can modify the initially harsh growing conditions on an open and exposed reclamation site to increase seedling survival (Franklin et al. 2012). As a result, cover crops are often planted to encourage the recovery of natural soil processes and create conditions conducive to seedling growth and survival.

Cover crops are typically quick establishing species which provide the benefit of early vegetation cover while also possessing the ability to control weeds and, depending on the species, provide a canopy (shelter) when planted at adequate densities (Hartwig and Ammon 2002). Quick establishment is important for weed control because it limits the amount of space available and therefore reduces the ability of aggressive weedy species to migrate onto a site (Franklin et al. 2012). In addition, the ability to form a canopy is advantageous because a canopy provides shade and many competitive and undesirable species are shade intolerant (Strong 2000). A canopy also provides tree seedlings with shelter from wind which, in combination with ground shading, reduces water loss through evaporation and cools the soil so that conditions are more favorable for the germination and establishment of understory species (Franklin et al. 2012). The main drawback of using cover crops is that they will inevitably compete with planted seedlings for light and nutrients; however, the desired outcome is that the benefits incurred will outweigh the associated costs of initially reduced growth and possible mortality. There is also the concern of non-native cover crops persisting in the community and this has recently been addressed by looking into the use of shade intolerant species as well as native forbs.

1.3 Fertilization

Capping materials used in reclamation generally have poor soil structure, low surface moisture, nutrient limitations, varying organic matter (OM) content, and a range of pH levels (Rowland et al. 2009; Alberta Environment and Water 2011). This occurs when capping materials are stockpiled due to increased soil handling as well as losses of nutrients through leaching and organic matter through volatilization. These initial conditions affect the long term goals of tree and understory species establishment and survival on oil sands reclamation sites. To help alleviate some of these conditions, it is a common practice to broadcast spread an agricultural grade, immediately available fertilizer (IAF) to reclamation sites for up to five years following revegetation (Pinno et al. 2012). Fertilizer is applied with the intention of increasing initial tree seedling growth which helps promote ecosystem function recovery; however, this is not always the case as IAFs release nutrients quickly and their low rate of recovery, particularly in relation to tree seedlings, makes them inefficient (Rowland et al. 2009; Sloan and Jacobs 2013). The low recovery rate or fertilizer use efficiency (FEU) occurs due to uptake by competing vegetation, immobilization in the soil by microorganisms, as well as leaching and possible contamination of surface and ground waters (Hangs et al. 2003; Fisher and Binkley 2000). This occurs in a reclamation setting because fertilizer is often broadcast spread over newly planted seedlings and the immediately available nutrients are either lost or immobilized because they are out of reach of tree roots or because seedlings have low nutrient requirements at the time of application (Sloan and Jacobs 2013). As a result, tree and understory species can face increased competition from the stimulated growth of nitrophilous species.

A more efficient alternative to using IAFs is the use of controlled release fertilizers (CRFs). These fertilizers have a higher FUE because they release nutrients slowly over time (3-18 months) and therefore release rates coincide more readily with tree seedling demand (Sloan and Jacobs 2013). Increased FUE is best achieved when CRFs are buried (either loose or in tea bags) near the rooting zone of seedlings or mixed in with the plugs; however broadcast spreading over a site is also an option (Sloan and Jacobs 2013). The close proximity of CRF's to the roots of tree seedlings, along with the slow release rates, result in higher FUE and thus fewer losses in the form of immobilization, leaching, and uptake by competing vegetation. Therefore the use of CRF has the potential to reduce the amount of nitrophilous species which compete with tree seedlings and understory species while still promoting moderate amounts of herbaceous cover. Moderate vegetation cover is desired for retaining nutrients in the ecosystem and promoting nutrient cycling (Chang and Preston 2000). However there is still the concern of dominance by nitrophilous species, which are not part of the target ecosystem, since any fertilizer application which increases their growth and abundance could hinder ecosystem recovery.

There is another potential option which could eliminate or reduce the amount of fertilizer applied to reclamation sites, reduce the presence of nitrophilous species, and requires no additional application effort. This option is planting tree seedlings which have been nutrient loaded by allowing the luxury consumption of nutrients during seedling growth in the greenhouse. Currently *Picea glauca* and *Populus tremuloides* have both demonstrated the ability to be nutrient loaded with minimal additional inputs during seedling production (Timmer 1997; Schott et al. 2013). The use of nutrient loaded seedlings, which are considered better quality planting stock, could greatly reduce the amount of fertilizer applied in the field while still promoting tree seedling growth and survival. Additionally, due to variation in soil quality and typically higher nitrogen levels in reclamation soils, fertilizer requirements are not uniform across all sites and a standard fertilizer blend may not always be beneficial (Alberta Environment and Water 2011). This is the most efficient use of fertilizer; however, there is limited research on if this will be an effective option for use on reclamation sites.

1.4 Understory development

Historically, revegetation of mine sites has mainly focused on tree establishment and growth, with less regard to understory species development. However, they are connected as understory shading, which occurs as a result of canopy closure, removes shade intolerant ruderal species from the community and promotes the growth of indigenous understory species which are shade tolerant (Strong 2000). Species establishment and availability following a disturbance, such as surface mining, is influenced by the propagules available on a site, propagule dispersal to a site, and site resource availability (Kenkel et al. 1997). Therefore, establishing understory forb species can be achieved if there is a sufficient propagule bank or seed source in the surrounding area; however, it has been found that planting shrubs is usually necessary (Rowland et al. 2009). The understory plant community in boreal forest ecosystems is the greatest contributor to plant species diversity and plays an important role in nutrient cycling, as herbaceous litter often contains more nutrients and decomposes faster than tree litter (Gilliam 2007). Shrubs are also an important part of understory communities as they add structural diversity and have been found to accelerate the development of the F and H layers of a forest floor (Rowland et al. 2009).

When attempting to accelerate forest succession on a reclamation site, it is important to consider the long term effects of management choices and incorporate objectives for tree and understory development. It is also important to consider that the creation of novel ecosystems is inevitable (Burton and Macdonald 2011) so management practices must be flexible and adaptable over time.

1.5 Objectives

Current and impending disturbances in the boreal forest have created a necessity for reclamation techniques which are integrative, efficient, and focus on long term ecosystem recovery goals. The overall objective of this thesis was to evaluate current reclamation practices and determine their effects on forest ecosystem recovery by assessing tree seedling survival and growth as well as initial vegetation development.

Chapter 2 presents the results of a study looking at the impacts of four different cover crops on out-planting performance of three boreal tree species. Additionally, cover crops and tree seedlings were grown in a forest floor material which had been salvaged and directly placed at two depths. Tree seedling performance was assessed in terms of height, mortality, and growth data that was collected over three growing seasons.

In Chapter 3, results are presented from a study looking at early vegetation development in two capping materials which were each treated with four different fertilizer regimes. Vegetation surveys were conducted over three growing seasons and percent cover estimates of each individual species was recorded. Community development was assessed by using this data to group species based on their growth form, life history strategy, native status, and presence in a mature forest understory community. Species richness and seed bank expression was also evaluated under the four fertilizer regimes in the two capping materials.

Chapter 4 summarizes the main findings from the previous two chapters and combines these findings with those of related studies to outline possible management strategies and applications. This chapter also identifies some study limitations and key areas for future research.

Chapter 2: Early tree seedling performance in response to cover crop and forest floor salvage and placement depth on a coal mine reclamation site

2.1 Introduction

Forest reclamation has become a common practice as a result of extensive surface mining throughout the boreal forest of Canada and the increasing need to restore these areas. In order to recreate functioning and self-sustaining boreal forest ecosystems, a canopy of trees along with a diverse understory of shrubs and forbs is required. The quick development of an overstory canopy is of particular importance because it will stabilize the site, initiate nutrient and carbon cycling, as well as facilitate and accelerate the recovery of native understory species while reducing agronomic and ruderal species (Strong 2000; Hart and Chen 2006; Lieffers et al. 1993). Along with a canopy, the retention of a nearby seed source or viable propagule bank is required for natural recovery of forest understory species (Holl 2002). Reclamation studies using direct placement of salvaged forest floor material to maintain propagule banks have shown promising results which include improved diversity and native understory species establishment (Mackenzie and Naeth 2010; Koch et al. 1996; Rokich et al. 2000). Within the boreal forest of Alberta, experimental soil salvage and their corresponding placement depths have ranged from 10 cm to 40 cm with the greatest density of establishment occurring at an approximately 20 cm depth (Mackenzie and Naeth 2012; Fair 2011). Due to the use of large scale machinery, it is believed that at shallow salvage depths (approximately 10 cm) the seed bank can be diluted as a result of admixing between the topsoil and subsoil during soil placement (Mackenzie and Naeth 2010). At greater salvage depths (approximately 40 cm), this dilution effect is further enhanced as a result of increased handling and mineral soil content (Fair 2011). As a result, first year establishment is influenced by the placement depth of soil with relatively shallower depths typically having greater establishment. Despite the benefits of this technique, there is still a great tendency for an initial dominance of competitive species which are early successional and potentially undesirable. These species can prevent or severely limit the development of forest vegetation (Hart and Chen 2006; Lee 2004). Therefore the direct transfer of salvaged forest floor material can be considered an improved method for increasing forest species establishment and diversity; however there is not a clear understanding of the impacts of forest floor salvage and placement depth on the establishment and growth of planted tree seedlings and the coinciding development of forest understories in boreal systems.

Initial tree seedling mortality and poor growth is strongly influenced by competition for light, water, and nutrients from surrounding herbaceous vegetation (Franklin et al. 2012), initial soil conditions (Conrad et al. 2002), as well as tree species selection and the quality of planting stock (Grossnickle 2005; Landhäusser et al. 2012). Competition control is often a necessity in the initial years of seedling establishment on reclamation sites; however the exact impact of competition on tree seedlings varies depending on the composition, height, and density of the competing vegetation. For example, at low levels of ground cover, facilitation can outweigh competition and benefit tree seedlings (DeSteven 1991), whereas high levels of ground cover is often detrimental to tree establishment (Franklin et al. 2012). While better quality planting stock can help seedlings tolerate early conditions, which include competition, tree species selection is also important because responses to competition will vary due to differences in growth patterns and shade tolerance. Consequently, a mixture of tree species is often planted on reclamation sites to increase the odds of success as well as mimic the pre-disturbance forest and improve ecosystem resiliency (Macdonald et al. 2012).

In the first year following soil placement there is often relatively little herbaceous cover and only fast growing, disturbance adapted, annual species thrive and dominate (Lee 2004; Fair 2011). While low cover results in less competition for planted seedlings, depending on its structure, early vegetation cover can modify the initially harsh conditions on an exposed reclamation site to increase tree seedling survival (Franklin et al. 2012). These modifications include; improved soil physical and nutritional processes, reduced soil erosion, increased infiltration rates, incorporation of organic matter, increased water holding capacity, and reduced soil temperature (Macdonald et al. 2012; Franklin et al. 2012). As a result, cover crops are often planted on reclamation sites to encourage the recovery of natural soil processes and create conditions conducive to seedling survival and growth. Cover crops are quick establishing species which provide all the previously mentioned benefits of early vegetation cover while also possessing the ability to reduce weeds and provide a canopy when planted at adequate densities (Hartwig and Ammon 2002). Quick establishment is important for weed control because it limits the amount of space available for germination and therefore reduces the ability of aggressive weedy species to migrate onto a site (Franklin et al. 2012). In addition, a canopy forming cover crop is particularly important because a canopy provides shade and many competitive and undesirable species are shade intolerant (Hart and Chen 2006; Strong 2000). Furthermore a canopy provides tree seedlings with shelter from wind which, in combination with ground shading, reduces water loss through evaporation and cools the soil so that germination conditions are more favorable for desirable understory species (Franklin et al. 2012). All cover crops will compete with planted seedlings for resources; however the desired outcome is that the benefits incurred will outweigh the associated costs of initially reduced growth and possible mortality.

Introduced agricultural grasses and legumes have historically been used as cover crops in agricultural crop production with beneficial results because it is common to terminate the cover crop before planting the succeeding crop (Hartwig and Ammon 2002). In contrast, their use on reclamation sites has shown varying levels of success due to the fact that the cover crop remains part of the recovering ecosystem and can compete with desirable species (Franklin et al. 2012). For example, non-native grasses have the ability to quickly establish, provide erosion control, reduce undesirable ruderal species establishment, retain nutrients, and incorporate organic matter into the soil; however they also compete with tree seedlings and can reduce desirable understory species establishment (Rowland et al. 2009; Waldron et al. 2005; Franklin et al. 2012; Landhäusser et al. 2007). In addition to the same benefits and drawbacks as non-native grasses, legumes also have the ability to fix nitrogen and capture snow (Turkington et al. 1978; Powell and Bork 2004; Blackshaw et al. 2001). *Melilotus officinalis* (L.) Lam. (yellow sweet clover) is an introduced, biennial legume with good potential as a cover crop because of its ability to fix nitrogen, suppress other herbaceous vegetation, quickly establish a diffuse biennial canopy, and lack of shade tolerance which ensures that it will not persist into the forested community (Blackshaw et al. 2001; Dickson et al. 2010; Turkington et al. 1978). While introduced grasses and legumes have the potential to be effective cover crops there is a concern for their persistence in the community, and the resulting effect on long term recovery, because most of the species typically used are not part of a natural forest ecosystem (Holl 2002). In contrast to using introduced species which are unlikely to persist in an understory, growth chamber studies have shown promising results for the use of an early successional native forb (*Chamerion angustifolium* (L.) Holub (fireweed)) as a potential cover crop (Landhäusser et al. 1996); however currently there is a lack of field studies which have tested this. A native forb could be beneficial as a cover crop because it would initially provide adequate cover and then be maintained at low levels in the understory (Broderick 1990). This would also provide increased diversity and thus resilience in the reclaimed ecosystem.

This research aims to compare the effectiveness of three cover crops in promoting early survival and growth of *Populus tremuloides* Michx. (trembling aspen), *Pinus contorta* Douglas ex Loudon (lodgepole pine), and *Picea glauca* (Moench) Voss (white spruce) while maintaining the establishment of native understory species from the forest floor propagule bank. The three cover crops include the traditionally used non-native annual grass *Hordeum vulgare* L. (barley), the non-native biennial legume *Melilotus officinalis* (L.) Lam. (yellow sweet clover), and the native perennial forb *Chamerion angustifolium* (L.) Holub (fireweed). In addition to the cover crop treatment, forest floor material was salvaged and directly placed at two different depths to compare potential effects and interactions on a reclamation site.

2.2 Methods

2.2.1 Research area

Research for this study took place within the Genesee Coal Mine operating lease (53°20'39" N, 114°18'10" W). The mine itself is a strip mine which is located approximately 80 km southwest of Edmonton, Alberta. Operations are a joint venture between Capital Power LP and Prairie Mines & Royalty Limited, with the current lease consisting of 28 sections of land (7, 252 ha) (Capital Power 2012).

This research area is part of the Dry Mixedwood subregion of the Boreal Forest Natural Region (Natural Regions Committee 2006). The research site is located in the southern portion of this subregion, where the topography ranges from level and gently undulating to hummocky uplands. The vegetation is characterized by *Populus tremuloides* Michx. dominated stands with scattered *Picea glauca* (Moench) Voss and interspersed fens in low lying areas. The reference understory plant community for this part of the subregion consists of *Corylus cornuta* Marshall, *Rosa acicularis* Lindl., *Aralia nudicaulis* L., *Lathyrus ochroleucus* Hook., *Lathyrus venosus* Muhl. ex Willd., and *Calamagrostis canadensis* (Michx.) P. Beauv. While these species describe the characteristic vegetation of the subregion, it is worth noting that 40-70% of the central area has been cultivated and planted with barley and forage crops. Aspen harvesting for pulp and paper as well as open pit mining and oil and gas activities are also common throughout this subregion (Natural Regions Committee 2006).

The dominant mineral soils in this subregion possess the characteristic eluvial and Bt horizons of a Gray Luvisol and occur where the mean annual soil temperature is less than 8°C. These soils usually support L, F, and H horizons and may have a degraded Ah or Ahe horizon as well (Natural Regions Committee 2006, Soil Classification Working Group 1998). The two most common subgroups that occur in this area are the Orthic Gray Luvisols and the Dark Gray Luvisols which formed from glacial till or lacustrine parent material. These two subgroups are primarily differentiated by an Ah or Ahe horizon greater than 5 cm thick in the Dark Gray Luvisols (Soil Classification Working Group 1998). Orthic Gleysols are often found on level to undulating landforms and organic soils can be found in low lying areas. In particular, wetland areas typically have Terric Mesisols whereas fens and bogs will have Fibric Mesisols (Natural Regions Committee 2006).

Compared to most of the other subregions within the Boreal Forest Natural Region, the Dry Mixedwood subregion has warmer summers and milder winters (Natural Regions Committee 2006). Average daily temperatures in the area range from a minimum of -11.7°C in January to 16.5°C in July, with an annual average of 3.4°C. Average annual precipitation totals 536 mm with 410 mm in the form of rain (mostly falling during the May to September growing season) and 133.9 mm as snow. The climate data presented was collected by Environment Canada (accessed 2013) at the Stony Plain weather station (53°32'51.006" N, 114°06'30.090" W) and covers a 29 year average ranging from 1971 to 2000 (Environment Canada 2013).

During the 2010 growing season (May to September), which occurred between the initial set up of the research site and tree planting, Stony Plain experienced 403 mm of precipitation and had an average temperature of 13.2°C (Environment Canada 2013). In the dormant period following tree planting (October 2010 to April 2011), 155 mm of precipitation fell, mainly in the form of snow, and the average temperature was -5.3°C. Precipitation during the first growing season for the planted seedlings (May to September 2011) totaled 368 mm and the average temperature during this period was 14.3°C. From October 2011 to April 2012, 153 mm of precipitation fell and the average temperature was -1.5°C. The second growing season (May to September 2012) had 389.8 mm of precipitation and an average temperature of 15.1°C. Precipitation during the period of October 2012 to April 2013 totaled 209 mm and the average temperature was -5.0°C. During the third and final growing season of this experiment (May to

September 2013), 269 mm of precipitation fell and the average temperature was 15.3°C (Environment Canada 2013).

2.2.2 Donor site

In order to perform the direct placement method used on this research site, a 4.6 hectare area was chosen in 2009 to act as the donor site for this study. The site was chosen based on its location within the Genesee Coal Mine lease, close proximity to the reclamation site (approximately 2.5 km away), and because it was scheduled for mining in the near future (Fair 2011). The donor site was an aspen dominated stand that was harvested approximately 10 years prior to selection and left to regenerate via root suckering. Prior to salvaging, tree and understory vegetation assessments were completed and the soil was classified as a Dark Gray Luvisol (Wachowski 2012). Along with the above ground vegetation sampling, soil cores were taken to assess what vascular species were present in the seed bank of the donor site and thus what species could be expected at the reclamation site (for more information see Fair 2011).

Prior to salvaging, soil pits were dug at the donor site to determine the optimum depth of the two forest floor salvage treatments. The deep salvage treatment was chosen to be 40 cm because this was the effective rooting depth of the trees while still remaining in good quality mineral soil (avoiding heavy clay and nutrient poor subsoil horizons). It was also predicted that at this depth the soil would be able to retain moisture better and keep the salvaged aspen roots alive (Wachowski 2012). The shallow salvage treatment was chosen to be 15 cm because this depth contained most of the forest floor and therefore root systems and propagules of the understory plant species present (Fair 2011).

2.2.3 Soil salvage and placement procedure

In late January 2010 the trees at the donor site were sheared off, just above the forest floor, with the straight blade of a D-11 caterpillar. This was done during frozen soil conditions (-16°C) to minimize compaction and disruption of the soil (Wachowski 2012). Shortly after shearing, D-11 caterpillars were used to salvage the soil and root systems at the 40 cm and 15 cm treatment depths. This was achieved by pushing the soil into alternating windrows of each treatment depth (Wachowski 2012). The 15 cm treatment was salvaged with only one pass from the D-11 caterpillar, whereas the 40 cm treatment required at least two passes. This was noted as it resulted in greater handling of the 40 cm treatment (Fair 2011).

Due to the use of industrial sized machinery, there were slight deviations from the target depths during salvaging; however these deviations were adjusted for during soil placement (Fair 2011). Following salvaging, the forest floor material (approximately 14,000 tonnes) was loaded separately (based on salvage depth) into Caterpillar 785C dump trucks and transported to the reclamation site. Each truck carried between 120-140 tonnes for a total of approximately 100 loads (Fair 2011). Once at the reclamation site, the donor material was spread directly on top of good quality overburden material in alternating strips of the two salvage depths using D-10 and D-11 Caterpillar bulldozers (Wachowski 2012). As a result of the size of the site and equipment used, placing the soil at the exact depth was also challenging; however, the relative difference between the 15 cm and 40 cm treatments was maintained. For example, the lower slope of the reclamation site received a slightly deeper placement of both treatment depths than the upper slope (Fair 2011). The entire salvage and placement procedure took 16 days to complete (Fair 2011).

2.2.4 Experimental design and cover crop treatments

This research was performed on an approximately four ha reclamation site which had been previously mined and re-filled with overburden material to create a 5-12% northwest facing slope (Wachowski 2012). The experiment was set up as a blocked, split-plot design with six experimental blocks (100 m wide by 65 m long). Half of the blocks were located on the upper slope position of the site and the other half on the lower slope position. Each block received both salvage depth treatments (15 cm and 40 cm) with a four meter buffer between them, thus creating two salvage depth plots (48 m wide by 65 m long) within each block. A cover crop treatment with four levels of species was superimposed randomly onto each of the two salvage depth treatments. This resulted in eight treatment plots (48 m by 15 m) per block and a total of 48 in the study. Within each treatment plot, four permanent one square meter vegetation assessment plots were established for long term sampling (total of 192) and two permanent stakes were used to center the 50 m^2 tree seedling assessment plots (total of 96) (Fair 2011).

On May 20 2010, one quarter of the treatment plots were seeded with *Melilotus officinalis* (L.) Lam. (yellow sweet clover) using an EarthWay Ev-n-spread 2700-A (EarthWay

Products Inc., Bristol, IN) hand held seeder at a rate of 0.2 kg/ha (Fair 2011). This was achieved by using an opening size of 1.25, mixing the *M. officinalis* seed with 200 g of cornmeal to even out the spread, and making four passes over each plot at a relatively quick walking pace (Fair 2011). In addition to *Melilotus officinalis* (sweet clover), *Hordeum vulgare* L. (barley) and *Chamerion angustifolium* (L.) Holub (fireweed) were each seeded into one quarter of the treatment plots at rates of 9.1 kg/ha and 2.7 kg/ha respectively (Fair 2011). Prior to seeding, the germination rates of all seeds were tested and the seeding rates were chosen based on typical seeding rates for reclamation areas (Fair 2011). The last quarter of the treatment plots was not seeded and acted as a control, allowing vegetation to recover from the original soil seed bank.

2.2.5 Tree seedling stock and planting procedures

One-year-old containerized seedlings of three tree species (*Populus tremuloides*, *Picea glauca*, and *Pinus contorta*) were grown commercially from open pollinated seed sources by Smoky Lake Forest Nursery (Smoky Lake, Alberta, Canada 54°6′ N; 112°28′ W; 598 m a.s.l.). *Populus tremuloides* seed came from the Edmonton area (53.65°N, 113.35°W) and seedlings were grown in 615 styroblocks (Beaver Plastic, Edmonton, AB) with a cavity size 6 cm in diameter and 15 cm deep (336 ml of soil volume). *Pinus contorta* and *Picea glauca* seed was collected in the Drayton Valley area (53.22°N, 114.97°W) and these seedlings were grown in 412A styroblocks (125 ml). The average height of seedlings at the time of plating was 33.7 cm for *Populus tremuloides*, 10.2 cm for *Pinus contorta*, and 24.4 cm for *Picea glauca*.

Tree planting at the research site was completed over a five day period (September 23-27) in early fall of 2010. Trees were planted at 1.3 m spacing using 64 m long planting ropes with pink flagging to mark out every 1.3 m. Planting lines were kept parallel by running the ropes between two evenly spaced survey stakes which were set up along the upper and lower edge of the planting boundary. The resulting planting density based on these methods was 5,917 stems/ha. Tree seedlings were planted as a random mixture with a ratio of 40% *Populus tremuloides*, 40% *Pinus contorta*, and 20% *Picea glauca* or a 2:2:1 mix respectively. While maintaining the desired planting mixture, tree planters also ensured that no more than 3 seedlings of the same species were next to each other in the same row. This species mix was used throughout the treatment area as well as in the buffer areas and plot edges. In each treatment plot, 25% of the area was left unplanted to assess natural vegetation establishment (Fair 2011) and aspen root suckering (Wachowski 2012).

2.2.6 Tree and vegetation measurements

Prior to bud flush in May 2011, initial tree measurements (height and root collar diameter (RCD)) were taken in the 50 m^2 circular seedling assessment plots. Each plot had a central stake which was marked with flagging tape and a 3.99 m forester's rope was fastened onto the stake for delineating the plot. Starting with the rope extended directly upslope (south) and moving in a counter-clockwise direction, all planted tree seedlings occurring between the stake and end of the rope were measured until a complete circle was made.

For all seedlings within the plots, height was measured to the nearest 0.5 cm from ground level to the bottom of the terminal bud and RCD was measured at ground level to the nearest 0.1 mm. Yearly growth was calculated by subtracting the average height of the current year by that of the initial planting height for 2011 and then the height of the previous year for 2012 and 2013. Average mortality was calculated from the proportion of living seedlings in a given plot after each growing season, compared to the initial plot density of each species. Browsing data was calculated after each growing season as the proportion of living trees which showed evidence of recent browsing in each plot. General comments were recorded on tree health, planting quality, mortality, and insect and/or herbivore damage.

For the initial measurements, only half of the seedling assessment plots were measured in order to attain an average initial height and diameter for each species. Following the 2011, 2012, and 2013 growing seasons, data was collected from all of the 96 seedling plots. Seedling measurements for 2011 and 2012 were taken in the spring of the following year because the vegetation cover was too dense in the fall to efficiently locate and measure all the trees. Measurements for 2013 were taken in the fall of the same year because the trees had grown large enough to be efficiently located amidst the other vegetation. The same measuring and recording procedures were used throughout the duration of this experiment.

This research site was established in the spring of 2010 which allowed the vegetation one full growing season to become established before the tree seedlings were planted. At the end of 2010, total average cover was significantly more in the 15 cm salvage and placement depth than the 40 cm depth (31% and 21% respectively) and in the sweet clover plots compared to the control (29% and 24% respectively) (Fair 2011). Further monitoring of vegetation development took place during the periods of August 9-18, 2011 and August 13-22, 2012 in the 192 (1 m^2) permanent vegetation plots. During each assessment, individual species were identified and their percent cover was estimated. Percent cover was measured to the nearest 1% if less than 10% and to the nearest 5% if greater than 10% cover (Fair 2011). Cardboard cut-outs of 1%, 5%, and 10% were used to help gauge accurate percent cover estimates and the same researcher preformed all of the estimations to ensure consistency.

When possible, species were identified in the field during the assessments; however, if exact species identification was not possible in the field, the specimen was collected, pressed, given a descriptive name, and brought back to the lab. Once in the lab, other resources such as identification books (Moss 1994; Royer and Dickinson 2007; Johnson et al. 2009; Bubar et al. 2000) or colleagues with experience identifying plants were referenced. All nomenclature used follows the United States Department of Agriculture Plants Database (USDA 2013).

The most common and abundant species or groups of similar species throughout the 2011 and 2012 growing seasons were examined to determine their effect on tree seedling growth and mortality. Species were determined to be common based on their average percent cover as well as the proportion of plots which they were present in. Depth and cover crop treatments were also considered to determine their effect on species composition and abundance.

2.2.7 Statistical analyses

In the year following seeding of the cover crops (2011), fireweed and barley did not successfully establish and therefore were not included in the subsequent analyses. As a result the rest of this thesis will focus on comparisons between only the sweet clover and control cover crop treatments.

Prior to running any statistical tests, the residuals of all data being used were checked for normality using the Shapiro-Wilk's test and equality of variance using Levene's test. There were only minor deviations from normality which, when visually assessed, did not skew in a particular direction. As a result, the statistical tests were carried out without transforming any data because the PROC MIXED procedure in SAS 9.2 (SAS Institute, Cary, NC, USA) is robust to minor deviations from normality (Littell et al. 2006).

This study was set up as a blocked split-plot design to determine the effect of soil salvage and placement depth (15 cm and 40 cm) and cover crop (sweet clover and control) on growth and mortality of *Populus tremuloides, Pinus contorta*, and *Picea glauca* seedlings. The depth treatment was the main plot (fixed effect), cover crop was the split-plot (fixed effect), and block was included in the random term. Overall tree height and mortality data were analyzed using a 3-way Analysis of Variance (ANOVA) with depth, cover crop and tree species as variables. Additionally, annual tree growth and mortality data was analyzed over the duration of the experiment (2011, 2012, and 2013) using a 4-way repeated measures ANOVA with depth, cover crop, tree species and year as independent variables. Results of the Type III Test of Fixed Effects were examined to see if any of the variables had a significant main effect or if a significant interaction occurred between any variables. Whenever an interaction occurred, comparisons were made using lsmeans and alpha was adjusted manually for the predetermined number of comparisons. Both statistical analyses were performed using the PROC MIXED procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA). Furthermore, annual tree browsing data was analyzed separately in each year for each species using a two-way ANOVA with depth and cover crop as variables. This analysis was also performed using the PROC MIXED procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA).

Vegetation cover over the 2011 and 2012 growing seasons was analyzed using a 3-way repeated measures ANOVA with depth, cover crop and year as variables. The analysis was run separately for total cover (which excluded sweet clover in the cover crop plots), sweet clover cover, and overall cover (which included sweet clover cover). This statistical analysis was performed using the PROC MIXED procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA). Cover data from the four most common species (or group of similar species) as well as the cover crop (*Melilotus officinalis*) was compared to annual tree growth and mortality data throughout the 2011 and 2012 growing seasons using simple linear regressions to determine if any correlations existed. Simple linear regressions were performed using the PROC REG procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA).

2.3 Results

2.3.1 Total mortality and annual mortality rates

Following three growing seasons on this reclamation site, total mortality of *Populus tremuloides* and *Pinus contorta* seedlings was 62% and 59% respectively. This was much higher than in *Picea glauca*, where total mortality over the same period was only 25% (Aw: SEM=5.10; Pl: SEM=4.58; Sw: SEM=6.16; $p<0.001$). Overall, combined mortality of the three tree species in the 15 cm salvage and placement depth treatment was approximately twice as much as in the 40 cm treatment (15 cm: average=64%, SEM=4.95; 40 cm: average=34%, SEM=3.90; $p=0.012$). Furthermore, the sweet clover cover crop treatment increased mortality across all tree species and placement depths (sweet clover: average=53%, SEM=4.97; control: average=44%, SEM=5.20; $p=0.038$).

Similar to total mortality, the annual mortality rate across all three species was higher in the 15 cm treatment than in the 40 cm treatment with the exception of 2013 which had no difference between depth treatments (Figure 2-1). The mortality rate following each growing season (annual mortality rate) was affected by placement depth, however the pattern varied most among the three tree species at a placement depth of 15 cm (depth by year by species interaction, $p<0.001$; Figure 2-2). For example, in the 15 cm depth *P. tremuloides* had its highest mortality (49%) after the second year (2012), while *P. contorta* had its highest mortality (40%) after the first year (2011). Alternatively, *Picea glauca* did not show any statistically significant differences among the three years in the 15 cm placement treatment; however the highest observed mortality occurred after the second year (23%). Within the 40 cm treatment, the mortality rate was similar across *P. tremuloides* and *P. contorta*, whereas *P. glauca* generally had less mortality.

Annual mortality rates were also different in the sweet clover cover crop treatment compared to the control (cover crop by year interaction, $p=0.002$; Figure 2-3); however there were no differences among tree species. Following the 2011 growing season, the mortality rate was about 10% higher in sweet clover plots compared to control plots which only had 15% mortality. However, after the second (2012) and third year (2013) mortality rates did not differ between the sweet clover and control plots.

2.3.2 Total height and annual growth rates

After three growing seasons, the total height of *Populus tremuloides*, *Pinus contorta* and *Picea glauca* seedlings was greater in the 40 cm salvage and placement depth treatment than in the 15 cm (15 cm: average=41 cm, SEM=2.90; 40 cm: average=57 cm, SEM=4.54; p=0.001). Furthermore, the difference in total height of *P. tremuloides* seedlings in response to the depth treatment was proportionally greater than in the other two species (depth by species interaction, $p<0.001$; Figure 2-4). In the 40 cm depth treatment, *P. tremuloides* seedlings reached an average height of 92 cm and were taller than those in the 15 cm depth (66 cm). *P. glauca* seedlings also had greater total height in the 40 cm depth treatments compared to the 15 cm depth (48 cm and 38 cm respectively). *Pinus contorta* grew to an average height of 31 cm at the 40 cm depth although this was not significantly different from the 26 cm seedlings in the 15 cm depth. Overall, total height of tree seedlings was not affected by the cover crop treatment in this experiment (p=0.112; data not shown).

Annual growth rates mirrored the trend of final height as they were also impacted by the soil salvage and placement depth which varied by species (depth by species interaction, p=0.025). *Populus tremuloides* and *P. glauca* seedlings had higher growth rates in the 40 cm depth treatment than in the 15 cm depth treatments, while the growth rate of *P. contorta* was not affected by the depth treatment. The cover crop treatment had an overall neutral impact on the seedling growth rates; however the response varied among years (cover crop by year interaction, p=0.010; Figure 2-5). In 2012, annual growth rates of seedlings were less in the control plots than in the sweet clover plots. Furthermore, the 2012 growth rates in the control plots were less than all other year and cover crop combinations, which demonstrated similar average growth rates. This reduction in growth rate was mainly driven by *P. tremuloides* which had less annual growth in control plots during the 2012 growing season resulting in an almost significant cover crop by species by year interaction (p= 0.058; Figure 2-5). *Pinus contorta* and *P. glauca* seedling also tended to have lower growth rates in the control plots in 2012; however, these decreases were not significant. The reduced growth rates in the control plots during 2012 influenced yearly growth rates such that all tree species experienced the least growth in 2012 compared to 2011 and 2013 (2011: average=10 cm $year^{-1}$, SEM=0.75; 2012: average=7 cm $year^{-1}$, SEM=0.87; 2013: average=11 cm $year^{-1}$, SEM=0.95; $p<0.0001$).

2.3.3 Total cover and competing vegetation

During the initial year of vegetation development (2010), the 15 cm soil salvage and placement depth had greater total cover (which excludes sweet clover seeded as a cover crop) than the 40 cm depth (Fair 2011); however this effect did not last into 2011 or 2012. Furthermore, sweet clover cover and overall cover (which included the seeded sweet clover) were also not influenced by the depth treatment. In 2011, total cover was less in the sweet clover plots as compared to the control plots (p=0.01). This is because the average cover of sweet clover was 58% in the plots where it was seeded, compared to control plots which only had 4% sweet clover (Table 2-1). Although the presence of sweet clover during its mature state in 2011 significantly reduced the cover of other vegetation in the sweet clover plots, it also considerably added to the overall plot cover which was 135% compared to 102% in the controls (Table 2-1). In 2012, when sweet clover was immature, its average cover in sweet clover plots was only 14% and there were no differences in total cover or overall cover (Table 2-1).

Throughout the 2011 and 2012 growing seasons, *Trifolium* sp. (*Trifolium pratense* L., *Trifolium repens* L., and *Trifolium hybridum* L.), grasses (*Calamagrostis canadensis* and all other grasses combined), *Rubus idaeus* L., and *Galeopsis tetrahit* L. were the most dominant species in terms of percent cover and abundance. All of these species are known competitors in boreal reclamation areas and reduce tree seedling performance. Sweet clover was also a dominant species and can compete with tree seedlings, although it was mostly confined to the sweet clover plots with only a small amount occurring in the control plots (<5%).

Over the two growing seasons, soil salvage and placement depth played a larger role in influencing percent cover of the dominant species than the cover crop treatment (Table 2-2). In 2011, *Trifolium* sp. cover was 54% in the 15 cm placement depth, which was greater than in the 40 cm depth (30%), and at both depths cover declined in 2012 (p=0.037). All other species had relatively low average cover in 2011 but showed increased cover in 2012. The increase in cover was influenced by soil placement depth such that *Rubus idaeus* had the greatest cover in the 40 cm placement depth treatment (p<0.001) as opposed to grass species and *Galeopsis tetrahit* which had greater cover in the 15 cm depth treatment (p<0.001 and p=0.008 respectively). Throughout both years, sweet clover plots had reduced cover of *Trifolium* sp. compared to the control plots (p<0.001). On the other hand, *Galeopsis tetrahit* cover was greatest in the sweet clover plots following the 2012 growing season (p=0.008). Percent cover of *Rubus idaeus* and grass sp. was not influenced by the cover crop treatment (Table 2-2).

2.3.4 Annual mortality and growth in relation to the top five competitors

In terms of percent cover, *Trifolium* sp. and *Melilotus officinalis* (sweet clover) were the most dominant species in 2011 and grass sp., *Rubus idaeus*, and *Galeopsis tetrahit* were the most dominant species in 2012. Therefore these species were further examined to determine the influence of competition on annual mortality and growth of tree seedlings. *Pinus contorta* was the only species were annual mortality was negatively influenced by total competitor cover (combined cover of the five competitor species) (Figure 2-6). Furthermore, *P. contorta* had greater mortality in plots with increased *Melilotus officinalis* and *Trifolium* sp. cover (Table 2-3). In contrast, both *Populus tremuloides* and *Picea glauca* faced higher mortality in plots with increased grass sp. (Figure 2-7) and *Galeopsis tetrahit* cover; however they were not influenced by total competitor cover. *Rubus idaeus* cover had an overall neutral effect on annual mortality of the three tree species despite its dominance and known competitive nature. Annual growth of *Pinus contorta* increased in plots with greater *Melilotus officinalis* cover (Figure 2-8) and decreased in plots with greater grass sp. and *Galeopsis tetrahit* cover. *Populus tremuloides* also experienced reduced growth in plots with greater grass sp. cover (Figure 2-9) whereas *P. glauca* growth was slightly increased by *Rubus idaeus* cover. Total competitor cover and *Trifolium* sp. cover both had a neutral influence on annual tree growth over the two growing seasons (Table 2-3).

2.3.5 Browsing

The majority of browsing occurred on *Pinus contorta* seedlings, followed by *Populus tremuloides*, and then *Picea glauca*. More *P. tremuloides* were browsed in 2011 compared to 2012; however depth and cover crop treatment did not affect the amount of browsing in either year (Table 2-4). In 2011, *P. contorta* experienced significantly more browsing in the 15 cm depth plots than the 40 cm and in the control plots as compared to the sweet clover plots. In 2012, 40% of *P. contorta* seedlings were browsed and during this time approximately 30% more browsing occurred in the control plots compared to the sweet clover plots (p=0.005). *P. glauca* had the least browsing with yearly averages of only 3%.

2.4 Discussion

2.4.1 Soil salvage and placement depth and tree performance

After the three growing seasons it was evident that all tree seedlings performed better in terms of survival and growth in the 40 cm salvage and placement depth treatment compared to the 15 cm depth. However with this in mind, there were some differences observed among the three species and throughout the three growing seasons. Total mortality of roughly 60% for *P. tremuloides* and *P. contorta* was over double that of *P. glauca* and the majority of *P. tremuloides* mortality occurred at the 15 cm depth in 2012, as opposed to *P. contorta* which had the most mortality at the 15 cm depth in 2011. Annual mortality of *P. tremuloides* and *P. glauca* was most strongly related to the average plot cover of grass species which, in 2012, was greater in the 15 cm depth. *P. contorta* mortality was most strongly related to total competitor cover and in particular, *Melilotus officinalis* and *Trifolium* sp. cover. Both of these species were more prevalent in the 15 cm depth treatment in 2011. *Melilotus officinalis*, *Trifolium* sp. and grass sp. are competitive species which are known to cause mortality of tree seedlings so their presence at this reclamation site and in particular on the 15 cm depth treatment can help explain the high mortality rates (Torbert and Burger 2000; Lieffers et al. 1993). While these correlations were significant and complemented field observations, it is possible that they are not as strong because the tree and vegetation mensuration plots did not always overlap and the tree plots covered a greater area.

Additionally, the high mortality rates observed in this study could have been influenced by the quality of planting stock which can play a significant role in seedling survival and competitive ability (Grossnickle 2005; Landhäusser et al. 2012). While this may have added to mortality, it seems more likely that the time of planting played a larger role. In other studies on reclamation sites in the boreal where *Populus tremuloides* seedlings were planted in the spring, immediately following site set up, mortality rates were less than 5% after two growing seasons (Rodriguez-Alvarez 2011; Schott 2013). Seedlings in this study were planted in the fall onto a site in which the herbaceous vegetation already had a full growing season to develop, after which total cover was greater at the 15 cm depth due to less dilution of the propagule bank (Fair 2011). Therefore during initial establishment in 2011, tree seedlings were already faced with high levels of competition from previously established vegetation. As a result of seed dispersal and continued germination, the dilution effect did not last beyond 2010 and overall cover during

2011 and 2012 was not significantly different between the two treatment depths. The considerably lower mortality rates seen in other studies can be explained by tree seedlings facing less competition from other vegetation because either plastic mulch was used to control weeds or stockpiled soil was used and this resulted in less vegetation establishment (Rodriguez-Alvarez 2011; Schott 2013). With complete vegetation cover, light becomes limiting and it is the taller and/or shade tolerant species which gain the competitive advantage and have a better chance of survival.

Along with reduced mortality, the total height of tree seedlings planted on the 40 cm soil salvage and placement depth was greater than in the 15 cm depth. This trend was apparent in annual growth rates of *P. tremuloides* and *P. glauca* however the growth rate of *P. contorta* was not affected by depth treatment. The 15 cm treatment depth tended to have greater percent cover of the taller and more competitive species such as *Trifolium* sp., grass sp., and *Melilotus officinalis*; however by the end of the 2011 growing season both depth treatments had complete vegetation cover. A study by Franklin et al. (2012) discussed the possibility that there is not always a direct correlation between total ground cover and tree growth; instead the composition of the cover is more important. This could explain why tree seedling growth was most strongly related to individual species cover. In particular, annual growth declined in *P. contorta* and *P. tremuloides* with increased grass sp. cover whereas annual growth of *P. glauca* slightly benefited from *Rubus idaeus* cover. Furthermore the lack of a growth response to soil depth seen in *P. contorta* could be explained as this species was relatively shorter at the time of planting and is shade intolerant. Therefore it faced higher levels of competition for light even at the 40 cm depth compared to the taller *P. tremuloides* and the shade tolerant *P. glauca*. *P. contorta* typically establishes on open, sandy sites and grows quickly due to reduced competition for light so conditions on this reclamation site were not ideal for growth of this species. In addition, *P. contorta* experienced the most browsing of all three tree seedlings, especially in 2012. Therefore it is also possible that the positive effect of the 40 cm depth treatment on annual growth may have been negated by browsing as new growth is like candy for ungulates.

2.4.2 Cover crop and tree performance

In comparison to the effect of soil salvage and placement depth on initial tree growth and survival, the cover crop treatment was less influential. After three growing seasons, sweet clover had a negative impact on tree mortality and a neutral effect on tree height. Total mortality was higher in sweet clover plots compared to control plots; however this result was driven by increased annual mortality rates in 2011 as mortality was not different in 2012 or 2013. The increased mortality rate in 2011 occurred when sweet clover was mature (tall stage) and had formed a canopy which covered an average of 60% of the plots. When mature, sweet clover is a strong competitor (Dickenson et al. 2010; Blackshaw 2001) and while it has a diffuse canopy, *P. tremuloides* and *P. contorta* are shade intolerant species and can have reduced survival in an understory setting (Landhäusser and Lieffers 2001). This was seen in the positive relationship between annual mortality of *P. contorta* and increased plot cover of sweet clover. As a shade tolerant species, the mortality of *Picea glauca* was relatively unaffected by the cover crop treatment. Therefore it is likely that the combination of competition and shade from sweet clover resulted in greater mortality of *P. contorta* and, to a lesser extent, *P. tremuloides* seedlings in sweet clover plots compared to control plots in 2011. In 2012 sweet clover was less competitive in its immature state (short stage) and in 2013 tree seedlings were more established so the influence on mortality was similar to that in the control plots during these years. Furthermore, the overall impact on tree mortality might have been lower in 2011 if seedlings had been planted in 2010 at the same time that sweet clover was seeded. In this case tree seedlings could have become established earlier and thus would have been taller and better able to compete in 2011 when sweet clover was mature.

In general, sweet clover had a neutral impact on the final height of tree seedlings however it did affect the growth rates during the 2012 growing season. A trend of reduced annual growth for all species occurred in 2012; however during this growing season growth rates were greater in the sweet clover plots compared to the control plots. This was driven by *P. tremuloides* which had the greatest difference in growth rates between the sweet clover and control plots in 2012 compared to *P. contorta* and *P. glauca*. This can either be explained by a generally poor growing season in which the presence of sweet clover as a cover crop was able to benefit tree growth or because control plots had less suitable growing conditions during this year.

If the former is true, then because *P. tremuloides* has indeterminate growth it was possibly better able to take advantage of the open canopy and increased light levels in 2012 when sweet clover was in its immature state. *P. contorta* and *P. glauca* have determinate growth so they would have been less able to take full advantage of the open canopy conditions. Also, *P.*

glauca has shown the ability to maintain similar, if not better, height growth an understory as compared to open areas (Landhäusser and Lieffers 2001) so increased light would not necessarily affect the growth of this species as much. Perhaps in the future as sweet clover continues its biennial cycle of growth and is slowly phased out of the community, tree seedlings would have more pronounced benefits in terms of total height and annual growth rates from this cover crop.

However because sweet clover did not increase growth rates in 2012 - it just maintained growth at levels consistent with other years and the control plots in 2011 and 2013 - then the latter may be true and something was occurring just in the control plots to reduce tree growth. General weather data supports this conclusion as the growing season temperature was similar to other years and there was only an average of 20 mm more precipitation than the previous year so there is no obvious reason why yearly growth rates would have been reduced in 2012. Therefore it is most likely that some abiotic or biotic factor in the control plots was reducing the growth of trees and it was not present in the sweet clover plots. Based on personal observations in the field, the control plots in 2012 were often covered with a mat of dead *Trifolium pratense* stems and leaves from the previous year which was smothering tree seedlings. In comparison, the sweet clover plots did not experience this because the mature plants from the previous year remained standing so there was not a mat of dead vegetation on the ground.

2.4.3 Vegetation response to cover crop

Along with promoting tree growth and survival, the desired function of a cover crop is to reduce undesirable weedy species which are not part of the mature forest community. In this case, sweet clover was able to reduce the average plot cover of *Trifolium* sp. however the average plot cover of *Galeopsis tetrahit* was increased in sweet clover plots during the 2012 growing season when sweet clover was immature. Therefore in terms of competition it seems that *Trifolium* sp. can be suppressed although, as overall cover was not influenced by the cover crop treatment, this means that sweet clover was acting as a replacement competitor. However, this is only the case when sweet clover is mature so in the long run, with its biennial lifecycle, sweet clover could provide less competition to tree seedlings and understory species than a perennial cover crop. The only drawback is that during the immature phase of sweet clover annual species, such as *Galeopsis tetrahit*, which produce many seeds and survive in the seed bank are able to establish. This occurs because the stalks of mature sweet clover remain standing after they die which reduces the amount of decomposing material covering the ground, allows light penetration, and provides bare ground for species migration while still adding some leaf litter to the surface. While this may increase the abundance of annual species it can also provide conditions more favorable for understory species. In a study looking at understory species development on this reclamation site it was found that in 2012, when sweet clover was mature, the cover of forest species was not affected by the cover crop treatment however other (non-forest) species and non-native species cover was less in sweet clover plots compared to control plots (Macdonald et al. unpublished). Furthermore, one added indirect benefit was that mature living and standing dead sweet clover plants were able to act as a physical barrier and reduce browsing of *P. contorta* within the sweet clover plots in 2011 and 2012 compared to the control plots.

2.5 Conclusions

Soil salvage and placement depth had the greatest impact on initial tree growth and mortality; therefore it is recommended that, in similar environments to those found at this reclamation site, soil should be salvaged and directly placed at a greater depth. Furthermore in this same study the deeper depth showed greater aspen suckering and growth as well as increased cover of forest understory species (Wachowski 2012; Macdonald et al. unpublished). The benefits of using yellow sweet clover (*Melilotus officinalis*) as a cover crop were minimal and additional research in seedling density and seeding time might be required. Increased mortality and competition are the main risks of using a cover crop and sweet clover was no exception in this study. Sweet clover plots had increased overall mortality of tree species compared to control plots and overall height was not impacted by the cover crop treatment. Therefore initially it would appear that sweet clover did not provide any benefits in terms of reduced mortality or increased growth; however there were the indirect benefits of reducing *Pinus contorta* browsing and *Trifolium* sp. cover which might have resulted in the reduced annual growth observed in the control plots during the 2012 growing season. Presumably the impact of mortality would have been reduced if trees were planted in the same year that sweet clover was seeded and in this case the benefits of using sweet clover may have outweighed the costs. Either way, this study only follows the impacts of sweet clover on annual tree growth and mortality over three growing seasons so research would benefit from a longer monitoring period to determine what impacts the biennial life cycle of sweet clover could have on long term forest restoration.

Tables

Table 2-1: Average total cover (excluding sweet clover), sweet clover cover, and overall cover (including sweet clover) following the 2011 and 2012 growing seasons. Results are from a Repeated Measures ANOVA for each cover type (n=6). There were no interactions for soil salvage and placement depth treatments so significant differences for main effects are displayed using "a" and "b". The cover crop treatment had a significant interaction with time so "x, y, and z" were used to indicate differences (α=0.05).

	Total Cover		**Sweet clover Cover**		**Overall Cover**	
	2011	2012	2011	2012	2011	2012
Depth						
15 cm	85% a	113% a	34% a	11% a	119% a	124% a
40 cm	90% a	113% a	28% a	5% a	118% a	118% a
Cover Crop						
Sweet clover	77% y	113% x	58% x	14% y	135% x	127% xy
Control	98% x	113% x	4% y	2% y	102% z	115% yz

Table 2-2: Average percent cover of the four most dominant species present throughout the 2011 and 2012 growing seasons. A Repeated Measures ANOVA resulted in significant main effects of depth, cover crop and year as well as interactions between depth and year and cover crop and year (α=0.05). Significant interactions between depth and year combinations for a given species are indicated with the letters "a, b, and c" whereas cover crop by year interactions for a given species are indicated with the letters "x and y". Significant main effects are based on an average of both years and are represented by "*".

	***Trifolium* sp.**		***Rubus idaeus***		**Grass sp.**		***Galeopsis tetrahit***	
	2011	2012	2011	2012	2011	2012	2011	2012
Depth								
15 cm	54% a	12% c	4% b	8% b	4% b	26% a	1% c	19% a
40 cm	30% b	7% c	8% b	19% a	5% b	11% b	3% bc	10% ab
Cover crop								
Sweet clover	29% x*	2% x*	8% x	15% x	5% x	20% x	2% y	20% x
Control	55% x*	17% x*	5% x	12% x	4% x	18% x	2% y	9% xy

Table 2-3: Simple Linear Regression results for annual tree mortality and growth rates for *Populus tremuloides* (Aw), *Pinus contorta* (Pl), and *Picea glauca* (Sw) compared to the combined and individual percent plot cover of the top five competing species throughout the 2011 and 2012 growing seasons. Significant relationships are bolded (α=0.05) and numbers presented are R^2 values.

	Total Comp.	*Melilotus officinalis*	*Trifolium* species	*Rubus idaeus*	Grass species	*Galeopsis tetrahit*
Mortality rate (% year^{-1})						
Aw	0.021	<0.001	0.066	0.002	**0.294**	**0.210**
Pl	**0.253**	**0.170**	**0.101**	0.077	0.023	0.048
Sw	0.015	0.005	0.022	0.010	**0.182**	**0.233**
Growth rate (cm year^{-1})						
Aw	<0.001	0.015	0.007	0.034	**0.168**	0.072
Pl	0.036	**0.172**	0.016	<0.001	**0.189**	**0.176**
Sw	0.007	<0.001	<0.001	**0.095**	0.031	0.011

Table 2-4: Average browsing of *Populus tremuloides* (Aw), *Pinus contorta* (Pl), and *Picea glauca* (Sw) following the 2011 and 2012 growing seasons (n=6). There were no significant interactions and significant differences in means of the main effects are indicated by "a" and "b" for the soil salvage and placement depth treatment and "x" and "y" for the cover crop treatment (α=0.05).

	2011			**2012**		
	Aw	Pl	Sw	Aw	Pl	Sw
Depth						
15 cm	17% a	18% a	4% a	8% a	28% a	2% a
40 cm	10% a	7% b	1% a	6% a	51% a	5% a
Cover crop						
Clover	13% x	6% x	4% x	8% x	25% x	2% x
Control	13% x	19% y	1% x	7% x	54% y	5% x
Total	**13%**	**13%**	**3%**	**7%**	**40%**	**3%**

Figures

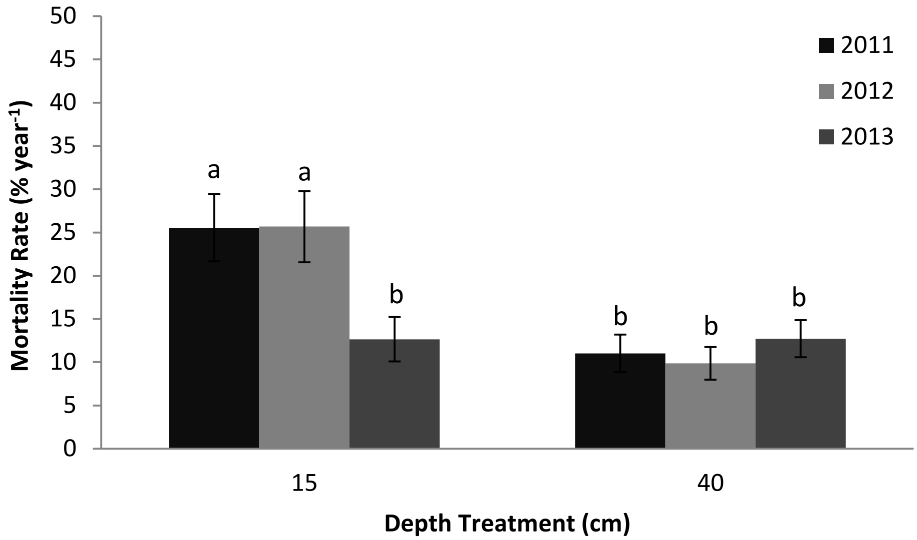

Figure 2-1: Average annual mortality rates in 2011, 2012, and 2013 of tree seedlings grown in 15 cm and 40 cm soil salvage and placement depth treatments. Mortality rates were calculated from the proportion of initial trees that were still living following each growing season. Error bars are standard error of the mean (n=36) and different letters above bars denote significantly different means.

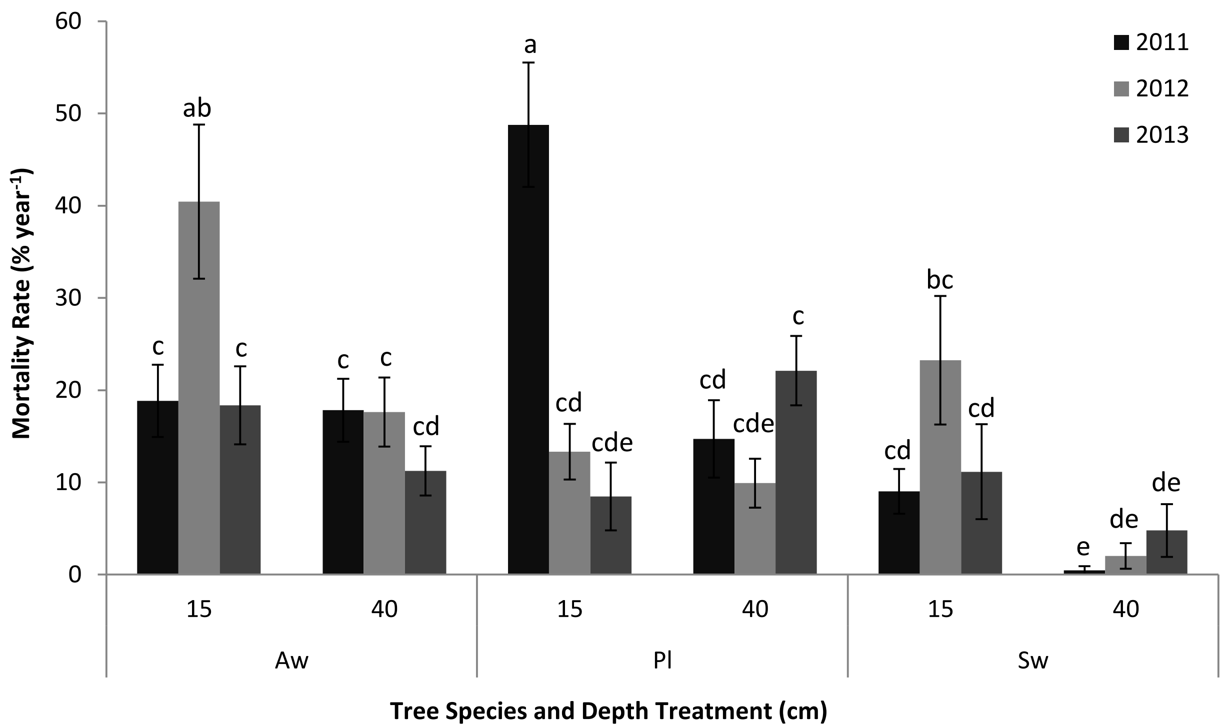

Figure 2-2: Average annual mortality rates in 2011, 2012, and 2013 of *P. tremuloides* (Aw), *P. contorta* (Pl), and *P. glauca* (Sw) grown in 15 cm and 40 cm soil salvage and placement depth treatments. Mortality rates were calculated from the proportion of initial trees that were still living each year. Error bars are standard error of the mean (n=12) and letters above bars denote significantly different means.

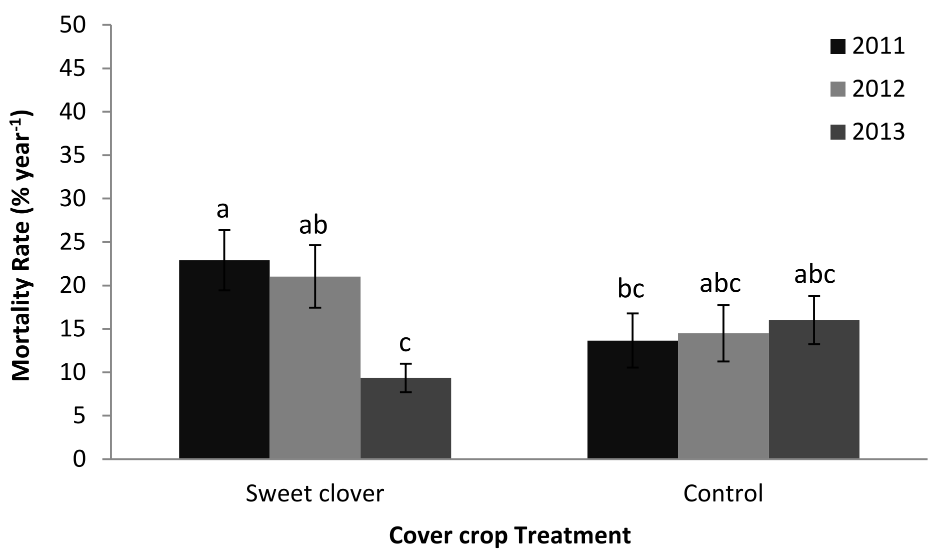

Figure 2-3: Average annual mortality rates in 2011, 2012, and 2013 of tree seedlings grown in sweet clover and control cover crop treatments. Mortality rates were calculated from the proportion of initial trees that were still alive following each year. Error bars are standard error of the mean (n=36) and different letters above bars denote significantly different means.

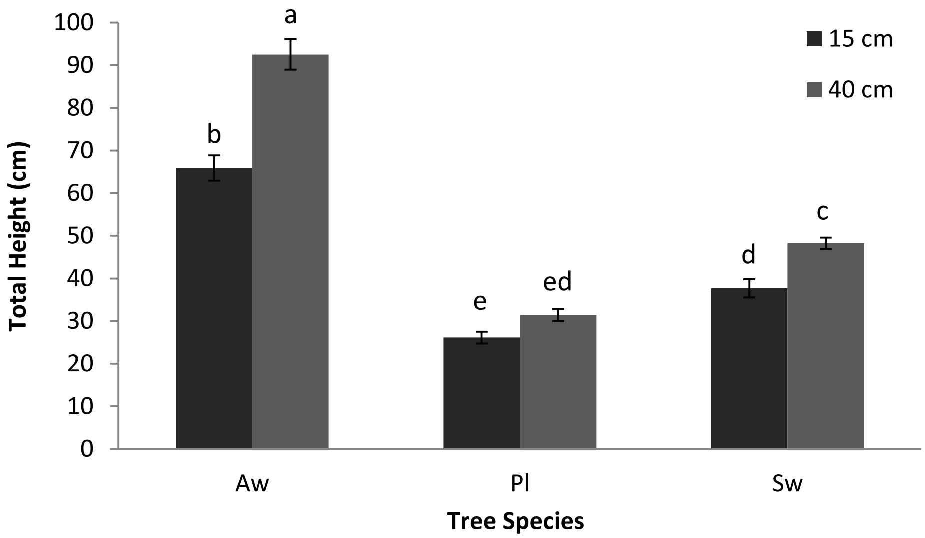

Figure 2-4: Average total height of *P. tremuloides* (Aw), *P. contorta* (Pl), and *P. glauca* (Sw) after three growing seasons in the 15 cm and 40 cm salvage and placement depth treatments. Error bars are standard error of the mean (n=12) and different letters above bars denote significantly different means.

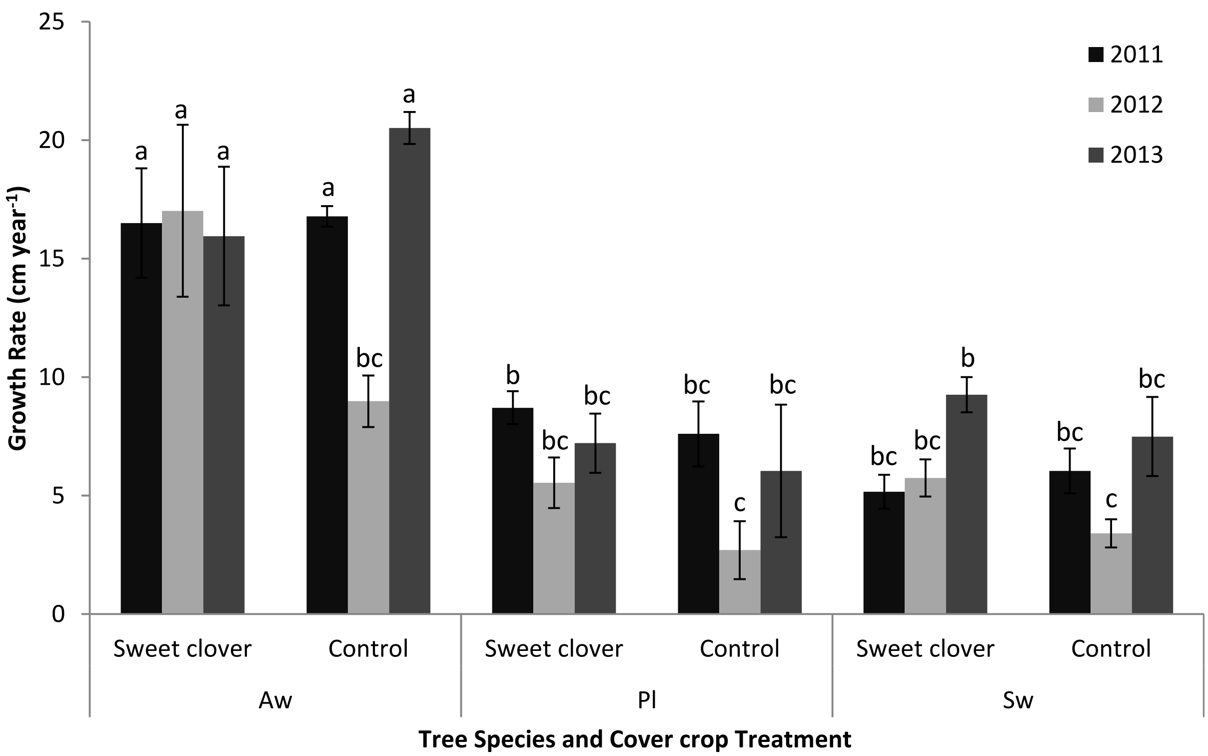

Figure 2-5: Annual growth rates in 2011, 2012, and 2013 of *P. tremuloides* (Aw), *P. contorta* (Pl), and *P. glauca* (Sw) grown in sweet clover and control cover crop treatments. Error bars are standard error of the mean (n=12) and different letters above bars denote significantly different means.

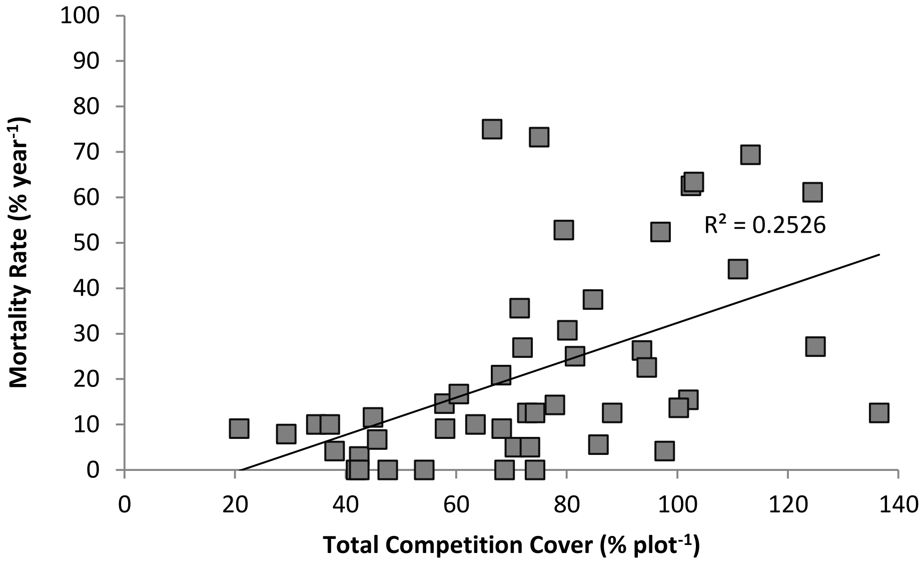

Figure 2-6: Relationship between average plot cover of total competition over the 2011 and 2012 growing seasons and the mortality rates of *Pinus contorta*. The line of best fit and R^2 value is displayed for the significant correlation (α=0.05).

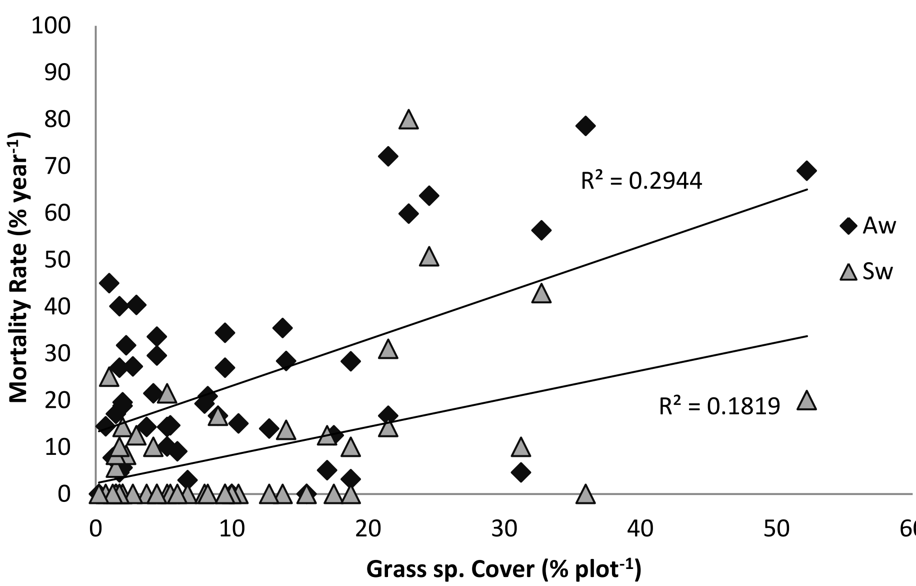

Figure 2-7: Relationship between average plot cover of grass sp. over the 2011 and 2012 growing seasons and the mortality rates of *Populus tremuloides* (Aw) and *Picea glauca* (Sw). Lines of best fit and R^2 values are displayed for significant correlations (α=0.05).

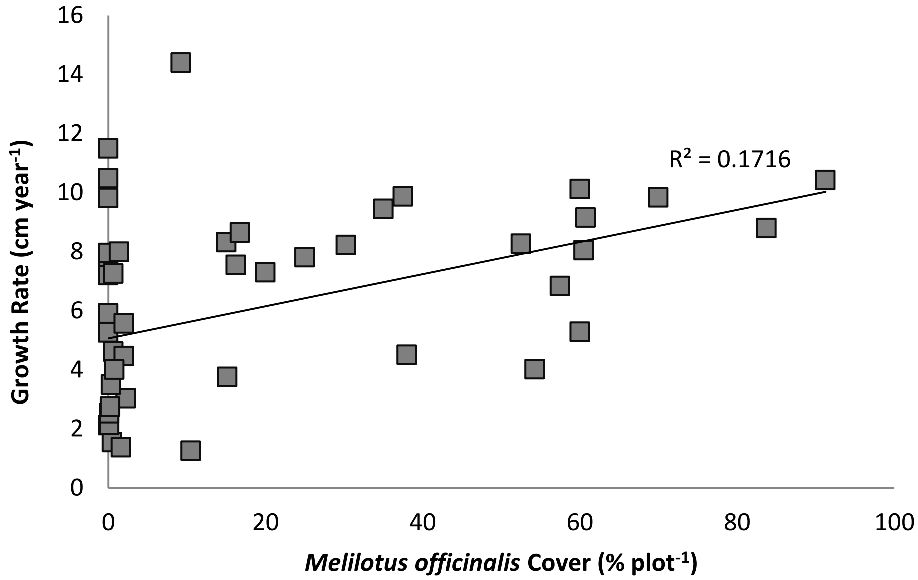

Figure 2-8: Relationship between average plot cover of *Melilotus officinalis* over the 2011 and 2012 growing seasons and growth rates of *Pinus contorta*. The line of best fit and R^2 value is displayed for the significant correlation (α=0.05).

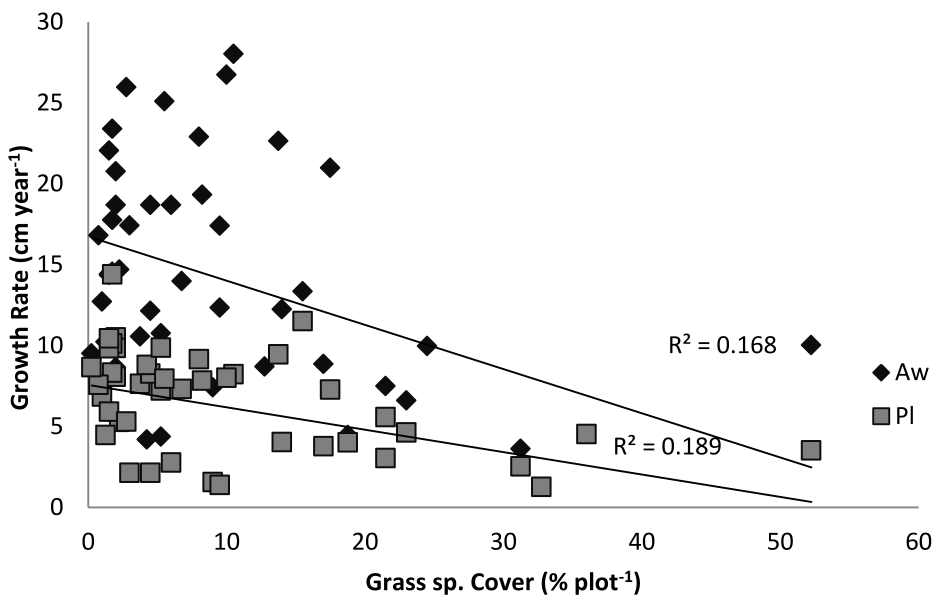

Figure 2-9: Relationship between average plot cover of grass sp. over the 2011 and 2012 growing seasons and growth rates of *Populus tremuloides* (Aw) and *Pinus contorta* (Pl). Lines of best fit and R^2 values are displayed for significant correlations (α=0.05).

Chapter 3: Early vegetation development in response to capping material and fertilization regime in oil sands reclamation

3.1 Introduction

Surface mining occurs throughout the boreal forest region in Canada; however the largest concentration of disturbed land can be found in the oil sands region of northern Alberta. In this region, roughly 71,500 ha of land has or is currently being mined and approximately 7,100 ha is under active reclamation (Alberta Government 2013). To reach the bituminous oil sands, which can be accessed up to a depth of approximately 75 m, the above ground vegetation must first be removed, followed by topsoil, subsoil, and overburden materials. Topsoil and subsoil are stockpiled separately until their use in reclamation is required. The initial reclamation process includes refilling the disturbed areas with mine residue and overburden, recontouring at a landscape level, and capping the area with subsoil and/or topsoil. The stripped topsoil which is used on reclamation sites as a capping material, is either a peat mineral mix (PMM) or a forest floor material (FFM) salvaged from the mine footprint. FFM is a topsoil mixture which contains organic forest floor materials (litter, fibric, and humic (L-F-H horizons)) and the underlying A and B mineral horizons salvaged from upland forest sites. The PMM ranges from 100% peat to a roughly 60:40 mixture of lowland peat and underlying mineral soil or transitional mineral soil salvaged from lowland areas (Alberta Environment and Water 2011).

In general, when compared to natural soils, reclaimed soils have elevated nitrogen levels, are more alkaline, have higher bulk density from heavy machinery traffic, and reduced soil moisture along with raised soil temperatures due to the absence of a forest canopy (Rowland et al. 2009; McMillan et al. 2007). Despite these general similarities, PMM and FFM have differing soil properties which can influence vegetation development on these capping materials. For example the PMM capping material generally has higher nitrogen and total carbon, lower available phosphorus, and is more alkaline than the FFM (Pinno et al. 2012). Furthermore, PMM has a higher water holding capacity whereas FFM has higher microbial activity (McMillan et al. 2007) and a more diverse propagule bank (Mackenzie and Naeth 2010). However, due to the topography of the original mineable area, peat is more abundant and therefore the PMM capping material is more readily available for use in reclamation.

The processes of soil excavation and stockpiling create the most challenges for revegetation because of the harsh soil conditions which are created. These include nutrient limitations, low surface moisture, varying organic matter (OM) content, and a range of pH levels which developing vegetation must contend with (Franklin et al. 2012; Pinno et al. 2012). To help alleviate some of these conditions, it is a common practice to broadcast spread an agricultural grade immediately available fertilizer (IAF) to reclamation sites for up to five years following revegetation (Pinno et al. 2012). Fertilizer application has the potential to increase tree seedling and vegetation establishment, promote soil stability, and enhance nutrient cycling recovery; however this is not always the case as IAFs release nutrients quickly and their low rate of capture, particularly in relation to tree seedlings, makes them inefficient (Rowland et al. 2009; Sloan and Jacobs 2013). The low capture rate or fertilizer use efficiency (FEU) can be the result of immobilization in the soil by microorganisms, uptake by competing vegetation, as well as leaching and possible contamination of surface and/or ground waters (Chang and Preston 2000; Ramsey et al. 2003; Hangs et al. 2003; Fisher and Binkley 2000). Immobilization and leaching occur because fertilizer is often applied out of reach of tree roots and/or when nutrient requirements are low (Sloan and Jacobs 2013). Furthermore, fertilization stimulates the growth of herbaceous, nitrophilous species which can increase competition for trees and decrease the migration of desirable boreal understory species onto a reclamation site (Ramsey et al. 2001; Alberta Environment and Water 2011; Davis et al. 1999).

Understory species are important to boreal forest structure and function as they drive biological processes, provide ecosystem services, and contribute to biodiversity (Nilsson and Wardle 2005). It is common that understory species mediate carbon dynamics, energy flow, and influence nutrient cycling of essential nutrients such as nitrogen, phosphorus, potassium, and magnesium (Gilliam 2007). In addition, understory species add organic matter to soils, can retain nutrients in an ecosystem, and are efficient at recycling nutrients because their foliage contains relatively high concentrations of nutrients and quickly decomposes (Gilliam 2007). This is particularly beneficial on reclamation sites for promoting soil formation, alleviating initially harsh conditions, and encouraging forest recovery. It is therefore important to consider the effects of fertilization on boreal forest understory species. Northern boreal forests are nutrient limited environments and many climax species in these forests are adapted to low nutrient conditions. In these environments, nutrient enrichment could favour nitrophilous grass and forb species which can grow fast, tall, and have large leaf areas (Grainger and Turkington 2013; Hedwall et al. 2013). These species will increase as a result of nutrient additions however sensitive forest species, which are less competitive under high nutrient availability, could disappear. As a result, fertilization of boreal forest ecosystems can lead to a decline in species richness (Grainger and Turkington 2013; Gilliam 2007). While this is often the case in a mature boreal forest it is unknown how fertilization affects the initial development and recovery trajectory of native boreal understory communities on reclamation sites.

In comparison to IAFs, controlled release fertilizers (CRFs), which are more efficient as a result of reduced leaching and uptake by competitors, could potentially be more beneficial on reclamation sites (Sloan and Jacobs 2013). These fertilizers have a higher FUE because they release nutrients slowly over time (3-18 months). Therefore release rates coincide more readily with tree seedling demand and nutrients are held in the environment longer (Sloan and Jacobs 2013; Donald et al. 1991). The use of CRFs on reclamation sites has shown promising results for increasing the FUE of tree seedlings and reducing competition from other vegetation (Hangs et al. 2003; Sloan and Jacobs 2013); however currently no studies have assessed the initial response of boreal forest understory species to the application of CRFs. In terms of tree seedling performance, CRF which is buried in the rooting zone of seedlings is better than broadcast application because this reduces nutrient losses to competing vegetation. However, in comparison, broadcast spreading CRF could be beneficial for understory species development as they also often require some assistance to establish and survive on reclamation sites. Many of these species arise from propagule banks of the capping materials or migrate onto site from the surrounding area; therefore as a result of their patchy distribution it is not feasible to apply CRF to individual plants. In terms of broadcast application, CRF could be more beneficial for boreal understory species than IAF because the delayed release of nutrients could potentially decrease competition from ruderal, nitrophilous species which can dominant a site following the application of fertilizer.

This study was set up to determine the impacts of applying IAF as well as CRF on initial vegetation development in two capping materials used for mine reclamation in northern Alberta. In particular the influence of differing fertilizer regimes on seed bank expression, species richness, vegetative cover, and forest understory community development was assessed over three growing seasons.

3.2 Methods

3.2.1 Research area

Research for this study took place on Mine Dump 8 (MD8) (56° 57' 41" N, 111° 18' 49" W), a saline and/or sodic overburden dump which is located within Suncor Energy Inc.'s Millennium Mine oil sands lease. The mine site is located about 30 km north of Fort McMurray, Alberta and includes approximately 20,000 ha of land disturbed by mining activities since 1967. Of this disturbed land, 1,439 ha are in the process of being reclaimed (Suncor Energy 2012).

This study was conducted within the Boreal Forest Natural Region on a site classified as part of the Central Mixedwood subregion (Natural Regions Committee 2006). The typical landscape of this subregion is rolling hills with uplands dominated by aspen (*Populus tremuloides* Michx.) and white spruce (*Picea glauca* (Moench) Voss), as pure or mixed forest stands. The low lying areas are wetlands dominated by black spruce (*Picea mariana* (Mill.) Britton, Sterns & Poggenb.) bogs. Upland forests contain a diverse understory including *Rosa spp*. L., *Viburnum edule* (Michx.) Raf., *Alnus viridis* (Chaix) DC., *Shepherdia canadensis* (L.) Nutt., *Cornus canadensis* L., *Aralia nudicaulis* L., and *Rubus pubescens* Raf. Wetland areas within the subregion often include *Ledum groenlandicum* Oeder, *Salix* spp. L., *Betula pumila* L., *Carex* spp. L., and a diversity of peat and feather mosses (Natural Regions Committee 2006).

The mineral soils in this subregion are typically Gray Luvisols which possess the characteristic eluvial and Bt horizons and occur where the mean annual soil temperature is less than 8°C. They usually support litter, fibric, and humic (L, F, and H) horizons as well and may have a degraded Ah or Ahe horizon (Natural Regions Committee 2006, Soil Classification Working Group 1998). Lower lying forested areas are often imperfectly drained, resulting in the formation of Gleyed Gray Luvisols or Orthic Gleysols. The Organic soils which occur under fens and bogs are typically Terric Mesisols (Natural Regions Committee 2006). These soils are at an intermediate stage of decomposition and remain saturated with water for extended periods of time (Soil Classification Working Group 1998).

The Central Mixedwood subregion experiences warm summers and long, cold winters. Average daily temperatures in the area range from a minimum of -18.8 °C in January to 16.8 °C in July with an annual average of 0.7°C. Average annual precipitation totals 455.5 mm and includes 342.2 mm of rainfall (mostly falling during the May-September growing season) and 155.8 mm of snow (Environment Canada 2013). The climate data presented was collected by

Environment Canada (accessed 2014) at the Fort McMurray weather station (56°39'0" N, 111°13'0" W) and covers a 29 year average ranging from 1971 to 2000.

Precipitation during the first growing season (May to September 2011) totaled 97.6 mm and the average temperature during this period was 15.9°C. In the dormant season from October 2011 to April 2012, 136.2 mm of precipitation fell and the average temperature was -4.9°C. The second growing season (May to September 2012) had 300.8 mm of precipitation and an average temperature of 16.3°C. Precipitation during the period of October 2012 to April 2013 totaled 169.9 mm and the average temperature was -9.6°C. During the third and final growing season of this experiment (May to September 2013), 320.7 mm of precipitation fell and the average temperature was 16.2°C (Environment Canada 2014).

3.2.2 Capping material and site preparation

Suncor follows soil salvaging techniques and requirements set out in their Environmental Protection Enhancement Act (EPEA) Approval No. 94-02-00 (as amended) for upland and peat mineral mix surface soils and subsoils. Surface soils, which were utilized as capping materials for reclamation at this site, were originally distributed within the footprint of the dump. Prior to mining this area fell within the "d" ecosite classification (Beckingham et al. 1996). This classification is characterized by an early successional *Populus tremuloides*, *Populus balsamifera*, and/or *Betula papyrifera* overstory with submesic to mesic moisture regime and medium nutrient status. After removal of the original forest cover, the site was stripped using excavators, heavy haulers, and crawler dozers between August 2008 and January 2009. Prior to stripping, when the ground was still frozen, D10 and D11 dozers with ripping blades were used to separate upland surface soil from the subsoil. Surface soil was classified as the top 30 cm of forest floor and included the L, F, H, and A horizons, as well as portions of the B horizon. As a result, the surface soil is a mixture of organic forest floor and mineral material and will be called forest floor material (FFM). Once the FFM was removed, another 30 cm was stripped which contained a mixture of the B and C horizons or subsoil. Scrapers were used to separate each lift and the material was hauled by truck 300 m west of the dump to a stockpiling area.

Lowland soils, which contained accumulations of peat over mineral subsoil horizons, as well as transitional soils, were stripped to a depth of approximately 30 cm. Soil was monitored during the salvaging process to ensure a consistent 60:40 peat-mineral mix (PMM) was achieved.

The stripping process occurred from August 2008 to January 2009 and was completed using excavators to windrow the PMM and load haul trucks. FFM, subsoil, and the PMM were placed in separate stockpiles and left for 2 years before their use on this reclamation site.

In 2009, the dump was filled with saline and/or sodic overburden material and sloping (with a goal of 6H:1V ratio) began on the central portion of the site. Prior to placement of the FFM and PMM capping materials, the dump surface was overlain with 1 m of low sodic till to provide a buffer between the capping materials and the overburden material underneath. Surface soil placement on MD8 began in August 2010 and was completed by June 2011.

3.2.3 Experimental design

On a level research site (1.3 ha) the FFM and PMM capping materials were placed in alternating strips of 20 m by 65 m thus creating five 40 by 65 m blocks. For the FFM treatment, the 1 m low sodic till was covered with 30 cm of subsoil followed by 20 cm of FFM. For the PMM treatment, the till was covered with 50 cm of peat-mineral mix. Four different fertilizer treatments were superimposed on the substrate treatments by dividing each strip into four 15 m by 15 m fertilizer treatment plots for a total of 40 plots. Fertilizer treatments were randomly assigned within each strip and plot size was chosen to allow for a 5 m buffer where capping materials changed. This study was set up in a 4 × 2 split block design with five replications to determine the effects of fertilizer application (high application rate of rapid release fertilizer (*high*); low application rate of rapid release fertilizer (*low*); high application rate of controlled release fertilizer (*slow release*); and no fertilizer (*control*)) and two capping materials (FFM and PMM) on early vegetation development at a reclamation site. All plots were planted with one-year-old aspen seedlings on June 9 and 10, 2011 at a spacing of 1.3 m (~6,000 stems per ha).

3.2.4 Fertilizer treatments

Peters General Purpose (20-20-20 N-P-K) fertilizer was used for both the *high* and *low* fertilizer treatments for this experiment. This fertilizer is immediately available for plant uptake and will be referred to as IAF hereafter. Osmocote Plus (15-9-12 N-P-K) with an 8-9 month nutrient release period was used as the *slow release* fertilizer treatment. Since this fertilizer has a controlled release of nutrients it will be referred to as CRF hereafter. Both products are manufactured by The Scotts Company LLC (Marysville, OH) and contain the following chelated micronutrients: Mg, B, Cu, Fe, Mn, Mo, and Zn. Both fertilizers were applied with the EarthWay Ev-n-spread 2700-A (EarthWay Products Inc., Bristol, IN) hand held seeder during the period of June 28-30, 2011. Plots assigned to the *high* treatment were fertilized at a rate of 500 kg/ha or 15 kg per plot (including their buffer area) and *low* fertilizer plots received half of this fertilization (250 kg/ha or 7.5 kg per plot). The *high* and *low* treatments were applied using and opening size of 3 on the spreader and plots were covered twice at a moderate walking speed during the application. The *slow release* treatment plots received 670 kg/ha of Osmocote Plus (equivalent to 20 kg per plot) which was chosen to equal the nitrogen application rate of the high fertilizer treatment. An opening size of 6 was used for the *slow release* treatments and the plots were also walked twice during the application. *Control* plots did not receive any fertilization.

3.2.5 Initial soil analysis and results

Prior to fertilization, two soil samples were collected from the top 20 cm of capping material along a transect at distances of 5 m and 10 m into each 15 m by 15 m plot to be used as a reference for initial soil conditions. Transects were aligned perpendicular to the strips of capping material and soil collection took place from May 10-11, 2011. Samples were analyzed for texture, available nitrate, plant available sulfur, plant available phosphate, plant available potassium, total phosphorous, total Kjehldahl nitrogen, conductivity, and pH. Soil texture measures were calculated from hydrometer readings and percent total clay (<2 µm), silt (2-50µm), and sand (>50 µm) were classified according to particle size (Carter and Gregorich 2008). For nitrate measurements, a dilute calcium chloride solution was used to extract available nitrate and nitrite from the soil. The nitrate was then passed through a copperized cadmium column to quantitatively reduce the nitrate to nitrite. After being diazotized with sulfanilamide and coupled with N-(1-naphthyl) ethylenediamine dihydrochloride, the samples were colorimetrically measured at 520 nm (O'Dell 1993). Plant available sulfur and potassium, were determined by ICP-AES on soil which was extracted with a weak calcium chloride solution (Carter and Gregorich 2008). Total Kjeldahl nitrogen (TKN) and phosphorous (TKP) were determined by using a SmartChem Discrete Wet Chemistry Analyzer (Westco Scientific Instruments Inc, Danbury, Ct) set to 660 nm to analyze samples that had been washed with sulfuric acid (Rutherford et al. 2008). A slurry of 1 part dry soil mixed with 2 parts de-ionized water (by volume) was allowed to stand with occasional stirring for 30-60 minutes. The pH of the resulting slurry was measured with a pH meter, and a conductivity meter was used to measure the conductivity of the filtered extract (Carter and Gregorich 2008). When data was normally distributed, t-tests were performed using the univariate procedure to compare soil sample results of the two capping materials (α=0.05). When data was non-normal, the non-parametric Wilcoxon Mann-Whitney test was used (α=0.05). Of the measured variables, the FFM had greater sand content, available phosphate-P, and available potassium-K, whereas the PMM had greater available sulfate-S (Table 3-1). All other measured variables did not differ between capping materials.

3.2.6 Seed bank study

To assess the seed bank potential of the capping materials, two soil samples were collected along the same transect as the initial soil samples and at the same distances (5 m and 10 m) into each of the 40 plots (80 samples). Collection occurred prior to fertilization and took place from May 10-11, 2011. Samples were obtained by applying even pressure and twisting a soil corer with a diameter of 14.5 cm in a clockwise-counter clockwise fashion until a depth of 10 cm was reached (~1,651.3 cm^3 of soil). The soil corer was then tilted to a 45° angle and a shovel was pushed into the ground alongside the bottom of the soil corer. This was done to hold the soil in the bottom of the corer and/or scrape off excess soil. The shovel and corer were then lifted simultaneously and the soil was inserted into a large Ziploc bag. At the reclamation site, bags of soil were labelled and put into a cooler until they could be transported to a refrigerator.

Soil samples were kept at 4°C in a refrigerator for one week until the seed bank study was initiated in a greenhouse at the Crop Diversification Center North (CDC North) in Edmonton, Alberta, Canada (53°38′ N; 113°21′ W). For each soil sample a rectangular potting tray with dimensions of 52 cm by 25 cm by 6 cm was filled with a 3 cm layer of Sunshine All-Purpose Planting Mix (a blend of sphagnum peat moss and perlite produced by Sun Gro Horticulture Canada Ltd., Seba Beach, AB) to help with moisture retention. The soil sample was spread evenly as a thin layer (1-1.5 cm) on top. Soil samples which did not readily spread as a result of high clay content were pushed through a screen with 0.6 cm by 0.6 cm square holes to break up the clay. Trays were misted with an irrigation system every day for 15 minutes at 9:00, 13:00, and 17:00. Misting nozzles were cleaned with vinegar as needed to ensure optimal and even misting. Furthermore, trays were rotated every week to compensate for spatial variability in the greenhouse.

Throughout the duration of the seed bank study, pictures were taken approximately every week and species were identified and counted. Individuals were identified to the species or, if not possible, the genus level using identification books (Moss 1994; Royer and Dickinson 2007; Johnson et al. 2009; Bubar et al. 2000). Genus level classification was used for aster, sedge, and willow species and grasses could only be identified to the family level. Nomenclature used throughout this experiment followed the United States Department of Agriculture database. After about a month in the greenhouse (June 16, 2011), species which had multiple individuals were thinned to only two individuals per tray to reduce competition and allow room for new species to emerge. Also, around this time, soil in the trays was re-disturbed through slight raking to promote germination of any buried seeds. After about two and a half months no new seedling emergence was observed and the study was terminated.

3.2.7 Field vegetation assessments

Initial vegetation development was assessed in one square meter sampling plots which were set up in both the southwest and northeast corners of a larger (8 m by 8 m) plot which was centered in each fertilizer treatment plot. Sampling of the 80 vegetation plots took place August 22-23, 2011, July 27-28, 2012, and July 30-31, 2013. During each assessment, individual species were identified and their percent cover was estimated. Percent cover was measured to the nearest 1% if less than 10% and to the nearest 5% if greater than 10% cover (Fair 2011). Cardboard cut-outs of 1%, 5%, and 10% were used to help gauge accurate percent cover estimates and the same researcher preformed all of the estimations to ensure consistency.

Species richness (total species per plot), percent cover of individual species (averaged based on two sub-plots), and cover by functional group (sum of individual species cover) were used to compare between the three years of growth. Along with total vegetation cover, three functional groups were used to categorize species encountered in this study: 1) life-form (forb, shrub, graminoid, or tree); 2) origin (native or non-native to Alberta (as per USDA 2014)); and 3) forest species (species considered characteristic of mature boreal forest understories or non-forest species). When possible, species were identified in the field during the assessments, however if exact species identification was not possible in the field, the specimen was collected, pressed, given a descriptive name, and brought back to the lab. Once in the lab, other resources such as identification books (Moss 1994; Royer and Dickinson 2007; Johnson et al. 2009; Bubar et al. 2000) or colleagues with experience identifying plants were used. Nomenclature followed the United States Department of Agriculture database. When counting species richness in 2012 and 2013, all genus level categories identified in 2011 were used even if a species level identification was possible. This was done to avoid exaggerating the addition of species between years however a complete list of all species identified can be found in Appendix VIII.

3.2.8 Plant root simulator probes

During the first two years of this experiment, Plant Root Simulator (PRS) probes (Western Ag Innovations Inc., Saskatoon, Canada) were used to measure bioavailable nutrients in the soil. Individual PRS probes consist of a 10 cm^2 ion exchange resin membrane surrounded by a plastic frame and handle. PRS probes were used in pairs consisting of one anion probe and one cation probe whose membranes adsorb the associated soil nutrients. In both 2011 (June 9-July 26 & July 26-September 10) and 2012 (June 8-July 26 & July 26-September 6) two burial periods (~45 days each) were used. This was done to avoid probe saturation and minimize any degradation of the accumulated nutrients by soil microbes. Each burial consisted of four pairs of PRS probes being used per subplot; two pairs in both the southwest and northeast corners of the permanent tree measurement plots. In order to capture both vertical and horizontal nutrient flow, each probe was inserted into the soil by hand at a 30-45° angle. The entire membrane was buried with only part the handle remaining visible above-ground. After burial, the soil around the probes was pressed to ensure contact between the soil and membrane. In 2011 two samples were analyzed per subplot for a total of 80 samples/burial period while in 2012 (when available nutrient levels had decreased without additional fertilizer input) all probes in each subplot were analyzed together for a total of 40 samples/burial period.

Upon removal, excess soil was knocked off and probes were placed in Ziploc bags and stored in a cooler for less than 4 hours before being moved to a refrigerator. Probes were stored in a refrigerator (for a maximum of 4 days) until cleaning. The membranes and plastic handles of the probes were thoroughly cleaned with de-ionized water and a toothbrush before being sealed in clean, Ziploc bags and being sent in an insulated box to Western Ag Innovations for analysis.

At Western Ag Innovations, the samples were processed according to their established methods. First, the ions were desorbed off the ion-exchange membrane using 0.5 M HCL. The resulting eluate was colorimetricaly analyzed for NO_3^-, NH_4^+, and P. Ca, Mg, Fe, Zn, Mn, Cu, Al, Pb and Cd were measured by inductively-coupled plasma spectrometry while K content results were obtained via flame emission. Results from early-season and late-season burials were pooled for each growing season and the results of PRS probe analysis are expressed in 10 cm^{-2} 94 $days^{-1}$ for 2011 and 10 cm^{-2} 91^{-1} days for 2012.

3.2.9 Statistical analyses

This study was set up as a blocked split-plot design with a capping material treatment (FFM and PMM) as the main plot (fixed effect), four levels of fertilizer treatments (*control*, *low*, *high*, and *slow release*) as the split-plot (fixed effect), and block was included in the random term. Due to well documented differences between the two capping materials, each capping material was analyzed separately to determine the influence of fertilizer treatments and year on species richness, total vegetation cover, and proportional cover by functional group. This was carried out using a two-way repeated measures ANOVA with fertilizer treatments and year as independent variables. Results of the Type III Test of Fixed Effects were examined to see if fertilizer treatments and/or year had a significant main effect or if a significant interaction occurred between the two variables. Whenever an interaction occurred, comparisons were made using lsmeans and alpha was adjusted manually for the number of comparisons being made. Statistical analyses were performed using the PROC MIXED procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA). Additionally, total percent cover was related to total available nitrogen for both capping materials in 2011 using simple linear regressions. Simple linear regressions were performed using the PROC REG procedure in SAS 9.2 and an alpha of 0.05 (SAS Institute, Cary, NC, USA). Prior to running any statistical tests, the residuals of all data being used were visually assessed and then checked for normality using the Shapiro-Wilk's test and/or equality of variance using Levene's test if deemed necessary.

An indicator species analysis was used to determine which species were significant indicators of the FFM and PMM capping materials (α=0.05) in the seed bank study as well as for each growing season at the reclamation site. Furthermore, a non-metric multidimensional scaling (NMDS) ordination was run using species presence-absence data for the seed bank study and all three growing seasons at the reclamation site. The ordination was run using the metaMDS procedure from the vegan package (Oksanen et al. 2013) with a random starting configuration, a stability criterion of 0.0005, and the Bray-Curtis distance measure. The final graph was reached using a 2D solution. Treatment ellipses and centroids were produced using the ordiellipse procedure from the ggplot2 package with a 95% confidence interval (Wickham 2009). All multivariate analyses were carried out using R 3.1.0, 64-bit (R Core Team 2014).

3.3 Results

3.3.1 Soil nutrient availability in 2011 and 2012

In 2011, total soil N supply (NO_3^+ and NH_4^- as determined by PRS probes) was greater in the PMM compared to the FFM ($p=0.0001$; Table 3-2); however both capping materials had the most soil N supply in the *high* plots followed by the *low*, *slow release*, and then the *control* plots (fertilizer effect $p<0.001$; Table 3-3A). In 2012 there was less total soil N supply than in 2011 (year effect $p<0.001$), although there were no differences between fertilizer treatments, capping material, or their interaction. Average soil phosphorus supply was influenced by fertilizer treatment in each year such that the *high* and *low* plots had greater soil P supply than the *control* and *slow release* plots in 2011 and in 2012 the *high* plots alone had the greatest soil P supply (fertilizer effect both $p<0.001$; Table 3-3A and 3-3B). Capping material and year had no impact on average soil P supply ($p=0.522$ and $p=0.210$ respectively). In 2011 and 2012 the FFM had greater average soil potassium supply than the PMM ($p<0.001$; Table 3-2). Average soil K levels within the two capping materials remained similar across fertilizer treatments in 2011; however in 2012 soil K supply was greatest in the *high* FFM plots which led to a fertilizer by year interaction ($p=0.027$; Table 3-3B).

3.3.2 Original seed bank of FFM and PMM capping materials

The FFM and PMM capping materials had relatively distinct species composition and only shared six species in common (two graminoids and four forbs) (Figure 3-8; Appendix I). The original seed bank of the FFM consisted of a total of 28 species and the majority were early successional forbs; however seven species could be considered characteristic of a mature boreal forest understory community (hereafter referred to as forest species) (Appendix I). The indicator species analysis suggested that there were eight indicator species describing the original FFM seed bank. Half of the indicator species were early successional native forbs (*Potentilla norvegica*, *Geranium bicknellii*, *Chenopodium album*, and *Chenopodium capitatum*) and the rest included an early successional native shrub (*Rubus idaeus*), a non-native forb (*Polygonum persicaria*), at least one grass species (Poaceae), and at least one sedge species (*Carex* sp.) (Appendix II). The original seed bank of the PMM contained 13 species. Most of these species were early successional forbs, with the exception of two native shrubs (*Salix* sp. and *Vaccinium oxycoccos*) and two forest species (*Achillea millefolium and Ledum groenlandicum*). Both of the forest species present in the PMM were considered rare as only one individual of each species germinated (Appendix I). The indicator species analysis for the original seed bank of the PMM suggested two early successional native forbs as indicators (*Epilobium palustre* and *Equisetum arvense*) (Appendix II).

3.3.3 Species richness and seed bank expression on the reclamation site

On the FFM at the reclamation site, total species richness was 33 species in 2011. Of the 33 species, 26 were forbs, two were shrubs, three were graminoids, and two were trees. In 2012, 10 new species were added to the FFM and two were lost resulting in an increase of total species richness to 41. The newly added species included nine forbs (three annuals and six perennials) and one perennial shrub (*Salix* sp.). By 2013, 17 more species were found (five annuals and 12 perennials); however species richness began to level off as 12 former species disappeared (six annuals), thus leaving total richness at 46 species. Of the 27 species added throughout the 2012 and 2013 growing seasons, 20 were not previously encountered on the reclamation site or in the original seed bank study and thus were presumed to have migrated in from the surrounding area. Of the seven other newly added species, four were previously found in the PMM and three were found in the original FFM seed bank (Appendix III). Furthermore, 10 of the new species were considered forest species (seven forbs and three shrubs).

Average species richness (per plot) of the FFM was greater in 2012 with 15 species compared to the average of only 13 species in 2013 (year effect $p=0.005$). Additionally, average richness was not affected by the different fertilizer treatments (fertilizer effect $p=0.498$) and there was no interaction between year and fertilizer ($p=0.237$). However, when total richness was compared for each treatment, the *low* plots had the greatest total species richness followed by the *slow release*, *control*, and then *high* plots (Figure 3-1). Furthermore, of the seed bank species expressed on the reclamation site in 2011, there was a loss of 31-45% of species observed in the original seed bank (Figure 3-1; Appendix IV). Throughout the 2012 and 2013 growing seasons, the loss of original seed bank species tended to continue. Between 2011 and 2013, a total of seven species from the original seed bank were lost from the reclamation site. Six of these species were early successional forbs (four annual and two perennial) and only one was a forest species (*Galium boreale*) (Appendix IV).

Of the 22 species found at the reclamation site on the PMM in 2011, 14 were forbs, four were shrubs, two were graminoids, and two were trees. Total species richness increased to 32 species in 2012 with the addition of 12 new species and the loss of two former species. Of the new species, 11 were forbs (three perennials and eight annuals) and one was an annual graminoid (*Juncus bufonius*). In 2013, total species richness declined in the PMM substrate as seven species were lost (four annuals and three perennials) and only five were added (one annual and four perennials). Furthermore, throughout the 2012 and 2013 growing seasons, three of the newly added species were found in the original seed bank, seven were previously found in the FFM, and six were not previously encountered on the reclamation site or in the original seed bank study (Appendix V). Additionally, 4 of the new species were forest species (*Vicia americana*, *Symphyotrichum ciliolatum*, *Mitella nuda*, and *Achillea millefolium*) and one was a native boreal tree species (*Picea glauca*) (Appendix V).

Average species richness (per plot) of the PMM was 12 species in 2012 and 2013 compared to only six species in 2011 (year effect $p<0.0001$). Over the three growing seasons, fertilizer did not affect average richness (fertilizer effect $p=0.98$) and there was no interaction between year and fertilizer ($p=0.4947$). Total species richness for each fertilizer treatment increased the most in 2012, particularly in the *control* and *slow release* plots (Figure 3-2). The seed bank species expressed in 2011 on the reclamation site showed a loss of 39-46% of the species observed in the original seed bank (Figure 3-2; Appendix VI). Furthermore, regardless of fertilizer treatment, the species representation from the original seed bank in the PMM remained relatively constant over the three growing seasons (Figure 3-2; Appendix VI). The only original seed bank species that was present at the reclamation site in 2011 and then disappeared was the native shrub *Ledum groenlandicum*.

3.3.4 Total cover of species and cover of functional groups in the FFM

In the FFM, total plant cover across all fertilizer treatments was greater in 2012 and 2013 compared to 2011 (year effect p=0.0020). In 2011, total plant cover increased with increasing total available nitrogen in the FFM plots (p=0.051; Figure 3-3) such that total cover was greater in the *high* fertilizer plots compared to the *control* plots (fertilizer effect p=0.012); however in 2012 this relationship was not detectable (see also *3.3.1*). In 2012 and 2013 there were not differences in total cover between the fertilizer treatments and this resulted in a significant year by fertilizer interaction term (p=0.0232; Figure 3-4; Appendix VII). Within the FFM, approximately 90% of the total cover was made up by forbs in 2011; however forb cover decreased with each growing season to only about 30% of the total cover in 2013 (year effect p<0.001). In 2011 and 2012, the portion of forbs relative to total cover was not different between any of the fertilizer treatments (Figure 3-5). It was not until 2013 that fertilizer had an effect and there was a greater portion of forbs in the *control* plots compared to the *low, high*, and *slow release* plots which resulted in a significant year by fertilizer interaction (p=0.0053; Appendix VII).

The proportion of graminoids relative to the total cover increased from less than 10% of the total cover in 2011 to roughly 60% in 2013 (year effect p<0.0001). This was influenced by the addition of fertilizer, as the *control* plots had less graminoid cover developing than the other three treatment plots which received fertilizer (fertilizer effect p=0.0066). Between 2011 and 2012 the portion of graminoids was not affected by the different fertilizer regimes which resulted in a significant year by fertilizer interaction term (p=0.0173; Figure 3-5; Appendix VII). Shrubs and trees made up only a small portion of the total plant cover in the FFM and while their cover increased with time (year effect p<0.0294), the different fertilizer treatments did not have an effect (fertilizer effect p>0.40; Appendix VII). The proportion of forest species, non-native species, and annuals/biennials to total cover also changed over the three year period (year effect all p<0.003). The portion of forest species was similar between 2011 and 2012 but increased in 2013. The portions of non-native species as well as annual/biennial species were greatest in 2011 and then decreased with each following growing season.

3.3.5 Total cover of species and cover of functional groups in the PMM

In the PMM, total plant cover was greater in 2012 and 2013 than 2011 (year effect $p<0.0001$); however, in 2011, total cover was found to increase with increasing total available nitrogen ($p=0.023$; Figure 3-3). Although this relationship was short lived and had disappeared by 2012 (see also *3.3.1*). In 2012 *control* plots had less total cover than *slow release* plots (fertilizer effect $p=0.0320$); however total cover was not different for any of the fertilizer treatments in 2011 or 2013, resulting in a significant year by fertilizer interaction term ($p=0.0105$; Figure 3-6; Appendix VII).

Forbs made up the largest portion of total cover in the PMM; however neither year nor fertilizer treatment impacted forb cover. The portion of graminoid cover changed with time and was greatest in 2013 (year effect $p=0.0031$). Shrub cover relative to total cover was not affected by year or fertilizer treatment; however the portion of tree cover was greater in 2013 than in 2012 (year effect $p=0.0238$; Appendix VII; Figure 3-7). The cover of forest species relative to total cover remained low in the PMM throughout the duration of this study and there were no effects of year or fertilizer. Both the proportional cover of non-native species as well as annual/biennial species to total cover changed over time (year effect all $p<0.0130$). Non-native species decreased between 2011 and 2012 whereas annuals/biennials decreased between 2012 and 2013.

3.3.6 Reclamation site community dynamics

The NMDS ordination for the seed bank study and the three growing seasons combined shows distinct community composition for the two capping materials, where FFM plots load towards the left side of the ordination and PMM plots load towards the right side (Figure 3-8). For both capping materials, the seed bank plots loaded towards the bottom of the ordination and had the most overlap of ellipses with plots from the first growing season (2011) at the reclamation site. In 2011, there was no overlap between the two capping materials at the reclamation site (Figure 3-8). Species composition in the 2012 growing season was similar to that in 2011 for both capping materials; however the FFM and PMM plots shifted towards each other and their ellipses began to overlap in this year (Figure 3-8). Following the 2013 growing season, species composition continued to shift upwards on the ordination and the two capping materials maintained some similarity in species composition through the continual overlap in ellipses. Also, in this year, only the FFM plots maintained some overlap with their original seed bank community. Throughout the duration of this study, the FFM plots tended to have greater similarity between plots in each year. Furthermore, with each growing season, the FFM experienced smaller shifts in species composition than the PMM (Figure 3-8).

The indicator species analysis suggests that in 2011 there were 15 indicator species in the FFM and nine of them were annuals/biennials (Appendix II). Furthermore, six of these species were early successional native forbs (*Chenopodium album*, *Chenopodium capitatum*, *Corydalis aurea*, *Dracocephalum parviflorum*, *Geranium bicknellii*, and *Potentilla norgevica*), three were graminoids (*Carex* sp., grass sp., and *Juncus bufonius*), one was an early successional native shrub (*Rubus idaeus*), and three were considered forest species (*Vicia americana*, *Lathyrus ochroleucus, and Aster sp.*). Only two of the indicator species were non-native forbs (*Polygonum persicaria and Kochia scoparia*). In 2012 the indicator analyses suggested 10 indicator species in the FFM; however, nine of them were already identified as indicator species in 2011. The only new indicator species in this year was the non-native, perennial forb *Sonchus arvensis*. The number of indicator species in 2013 remained at 10, however only 5 of these species had been previously been identified as indicators. The species which were no longer indicators of the FFM included two early successional native forbs (*Potentilla norvegica* and *Dracocephalum parviflorum*), an early successional native shrub (*Rubus idaeus*), a native graminoid (*Juncus bufonius*), and a non-native forb (*Polygonum persicaria*). On the other hand, three additional forest species (*Achillea millefolium, Fragaria virginiana*, and *Symphyotrichum ciliolatum*) became indicators in this year (for a total of five) as well as a graminoid (*Agrostis scabra*) and a non-native forb (*Polygonum aviculare*).

The three indicator species for the PMM capping material in 2011 were all native perennials and included a forb (*Equisetum arvense*), a tree (*Populus balsamifera*), and a shrub (*Salix* sp.) (Appendix II). By 2012 the number of indicator species had increased to seven and included the three from 2011, as well as one native tree species (*Populus tremuloides*), two early successional native forbs (*Epilobium angustifolium* and *Chenopodium album*), and one non-native forb (*Senecio vulgaris*). Following the 2013 growing season, five indicator species were identified and three of them were from 2012. The species which were no longer identified as indicators were *Salix* sp., *Populus tremuloides*, *Chenopodium album*, and *Senecio vulgaris*. The two new indicator species were both early successional native forbs (*Erigeron canadensis* and *Equisetum sylvaticum*).

3.4 Discussion

Varying the amount (250 kg/ha, 500 kg/ha, and 670 kg/ha) and type of fertilizer (IAF and CRF) did not impact plant community establishment as measured by average richness, species composition, and seed bank expression on either FFM or PMM capping materials. However fertilization did impact total cover on both capping materials as well as the proportion of forbs and graminoids in the FFM. The cover of all other functional groups explored in this study (trees, shrubs, non-native, annuals/biennials, and forest species) changed only in time or, in the case of shrubs and forest species in the PMM, were not affected at all. Despite some similarities in vegetation development related to the functional groups measured, total richness, species composition, and community development were quite different between the two capping materials. This indicates that capping material selection, which includes the propagule bank contained within, appears to play a much larger role in early vegetation establishment than fertilization.

During the first growing season (2011), total cover in the FFM plots increased with total available nitrogen such that the *high* fertilizer plots had greater total cover than the *control* plots. This resulted from the application of a large amount of immediately available nitrogen which stimulated the growth of nitrophilous forb species. Forb cover in the *high* plots consisted mainly of two early successional native forbs (*Geranium bicknellii* and *Potentilla norvegica*) as well as two non-native forbs (*Kochia scoparia* and *Polygonum persicaria*) which were dominant in the *high* plots but also in the *low* and *slow release* plots. These four species are ruderal, nitrophilous forbs that are able to thrive and dominate on disturbed sites, especially when nutrients are abundant. Additionally, these species contributed to the high portion of forbs (~90% of total cover) observed at the reclamation site in 2011. However, over the three growing seasons, fertilized plots (*low*, *high*, and *slow release*) in the FFM displayed a steady decrease in the portion of forbs and a corresponding increase in the portion of graminoids. By 2013, graminoids made up approximately 70% of total cover in all of the fertilized plots while only contributing to 37% of total cover in the *control* plots. Other studies have found concurring results in which dominant grasses compete strongly with forbs and can limit their establishment (Sluis 2002;

Dickson and Busby 2009). The most dominant graminoid in the fertilized plots was the native, perennial bunch grass, *Agrostis scabra*. This species does well on disturbed sites and, in particular, favours high nitrogen conditions (Tilman 1984). As a perennial, this species likely became established in 2011 when nutrient availability was greatest and then progressively became more dominant over time. The influence of fertilizer on total cover in the FFM was short-lived and did not last into the 2012 or 2013 growing seasons. This can be attributed to initial losses of the highly soluble nutrients due to leaching in combination with increases in total cover which tied up nutrients.

While fertilizer mainly affected total cover, it is interesting to note that in general the *low* fertilizer plots of the FFM tended to have roughly 30-40% greater total richness than the *high* plots, despite these plots maintaining similar cover each year. This can be attributed to the *high* fertilizer plots maintaining somewhat fewer species which had higher individual cover as a result of the greater initial input of nutrients. In fact, species composition was quite similar between these plots as 92-95% of the species found in the *high* plots were also found in the *low* treatment plots each year. Therefore the *low* plots, which received 50% less nutrients, were possibly better suited to promoting germination and migration of a larger number of species because there was greater heterogeneity in cover due to reduced competition from less and potentially smaller individuals. Additionally, 10 of the 21 rare species found throughout this study were in the *low* plots of the FFM and six of these were forest species. However, it is difficult to determine if this was instead due to chance because other studies have shown that richness can be strongly influenced by spatial heterogeneity as a result of differential seed dispersal as well as patchiness in soil moisture and fertility (Armesto et al. 1991).

Similarly to the FFM, total cover in the PMM plots increased with increasing total available nitrogen; however due to the low average total cover (5-16%) in the first growing season, significant differences between fertilizer treatments were not detectable. Low total cover in 2011 could be attributed to the relatively small seed bank of the PMM capping material (13 species in the PMM compared to 28 species in the FFM) which in turn resulted in low initial establishment at the reclamation site. This concurs with a study by Mackenzie and Naeth (2010) which also found lower initial establishment when using PMM as a capping material. In 2012, all of the plots which received fertilizer (*low*, *high*, and *slow release*) had significant increases in total cover. This likely occurred because these plots were able to retain some of the nutrients from the fertilizer application due to the initially low total cover on the PMM and since peat has a high cation exchange capacity which can help reduce nutrient leaching. *Slow release* plots had the greatest total cover in 2012 and this was significantly more than in the *control* plots. This is explained by the application of a CRF in the *slow release* plots which allowed for the continued release of available nutrients into the second growing season. Furthermore, this is supported by the presence of nitrophilous species, such as grasses and *Chenopodium album*, which dominated the *slow release* plots and *Epilobium palustre* which was prevalent in all of the fertilized plots in this year. By 2013, total cover was similar across all fertilizer treatments as a result of the reduction in available nutrients from leaching and the continual uptake by growing vegetation.

Forbs made up the largest portion of total cover in the PMM; however, unlike in the FFM, forbs maintained their dominance throughout this study and were unaffected by fertilizer treatment and year. The proportion of graminoids increased with time but was never more than 27% of the total cover. The low graminoid cover in the PMM compared to the FFM is possibly because fewer graminoids were found in the original PMM seed bank and the PMM was a less suitable substrate for graminoid species which might have migrated onto the site. Shrubs and forest species were not measurably affected by any of the treatments in the PMM; however this is not surprising as both of these groups had low cover throughout the duration of this study. Over similar periods of time, other reclamation studies have also found low shrub establishment and recommend planting as a solution (Rowland et al. 2009).

Species establishment on the reclamation site was lowest in 2011 and predominantly influenced by the respective propagule banks of the two capping materials. Low initial establishment and a lack of forest species in the seed bank can be linked to the use of stockpiled capping materials. Materials used in this study were stockpiled for approximately two years and studies have shown that stockpiling FFM for more than 8 months significantly reduces propagule viability through seed decomposition and germination while still in the stockpile (Naeth et al. 2013). Additionally, Macdonald et al. (unpublished) found that 21 forest understory species had established in the first growing season when FFM was salvaged and directly placed at a reclamation site, compared to only eight forest species in this study. When stockpiled capping materials are used, the initial vegetation which develops is typically disturbance adapted species which have long lived seed banks and the ability to rapidly mature, thus continually refreshing their seed banks (Lee 2004). This has a strong influence on initial vegetation establishment and can hinder ecosystem recovery by reducing the amount of desirable understory species in the propagule bank and slowing the rate at which they can establish on a reclamation site.

Total richness increased the most in the second growing season, primarily as a result of the addition of early successional forb species from the surrounding area. This occurred because early successional species can tolerate a range of soil conditions, mature quickly, and typically produce an abundance of seed which is wind dispersed; therefore allowing these species to establish on both capping materials at the reclamation site. This explains why species composition of the two capping materials became more similar in 2012 as they shifted towards each other in the ordination. Additionally, some early successional annual species, which were previously found in the FFM, established on the PMM. It is conceivable that this occurred because the low initial cover in the PMM plots provided available growing space for ruderal species to establish. Often the establishment of new species can be limited by the competition from already established species (Sluis 2002; Huddleston and Young 2004).

Throughout this study the FFM gained more new species relative to the PMM. Presumably these "new" species originated from the surrounding area as they were not present in the previous years at the reclamation site or contained in the original seed banks. Furthermore, the nearest seed source was a residual forest and almost half of the new species were forest understory species which indicates that the FFM was more conducive to the establishment of these species. This concurs with another study which attributed some of the higher species establishment in the FFM to the mesic soil conditions which are shared by FFM and natural upland forests (Mackenzie and Naeth 2010). The number of forest species found in the FFM increased with time and the most became established in the third growing season (2013). This occurred because some forest species, such as *Ribes triste* and *Rosa acicularis*, require animals to disperse their seeds and many animals prefer visiting areas with at least moderate vegetation cover. Other forest species, such as *Mertensia paniculata* and *Petasites frigidus* var. *sagittatus*, have wind dispersed seeds; however their establishment can also take some time as many factors such as plant height, wind speed, and wind direction affect their dispersal. In contrast, peatlands are typically cold and wet environments which favour hydrophilic species so, despite receiving a similar seed rain as the FFM, conditions on the PMM were not conducive for the establishment of all the same species.

Total species richness remained relatively constant between the 2012 and 2013 growing seasons; however community composition in both capping materials continued to change as the initially high portions of annuals/biennials and non-native species decreased. Seven out of 14 species from the FFM which disappeared over time were early successional annual species and only three were desirable forest species. In contrast, 17of the 27 species in the FFM which were gained throughout the duration of the study were native perennials and of those 11 were forest species. Furthermore, in 2011, nine out of 15 indicator species in the FFM were annuals and two were forest species; however by 2013, only two of 10 indicator species were annuals and five were forest species. The reduction in annuals and non-native species along with the trend towards increasing forest species suggests that the FFM is recovering along a desirable trajectory. Although in order to maintain the current forest understory species and promote the establishment of additional forest species, a tree canopy which provides shade is necessary.

For the PMM, five of the nine species which disappeared over time were early successional annuals and three were forest species. In 2012, nine of the 12 species gained by the PMM were annuals; however by 2013 only one of the five new species was an annual. In contrast to the FFM, none of the indicator species in the PMM were forest species. However, the native boreal tree, *Populus balsamifera*, was an indicator species in all three years and *Populus tremuloides* was an indicator species in 2012. Based on these results, the PMM appears to be on a different and potentially slower recovery trajectory than the FFM. This is presumably due to the low richness and species composition of the original seed bank as well as soil conditions which initially appear to be less conducive for the establishment of forest species. Despite this, the PMM appears to provide better conditions for the establishment of natural (non-planted) trees. This could be due to the increased organic carbon content and water holding capacity of the PMM relative to the FFM. Seeds of both *P. tremuloides* and *P. balsamifera* require a moist substrate following dispersal to ensure quick germination and establishment (Perala and Russell 1983). Additionally, it has been found that *P. balsamifera* can establish on a wide variety of soil conditions whereas *P. tremuloides* prefers soil substrates that have moderate moisture and contain organic carbon (Wolken et al. 2010; Pinno et al. 2012). Furthermore, a study looking at outplanting performance of *Populus tremuloides* seedlings on the same site found that planted seedlings performed better on the PMM capping material compared to the FFM (Schott 2013). While tree establishment is promising on the PMM, the lack of understory forest species present on the PMM will likely have a prolonged effect on the transition into an upland forest community and will require further attention.

3.5 Conclusions

In summary, the fertilizer applications tested in this study did not impact average richness or the expression of the seed bank during initial development of plant communities on either FFM or PMM capping materials. Instead, initial community composition was driven by the respective propagule banks of the two capping materials and therefore the treatment of topsoil (e.g. stockpiling) can have a critical impact on the speed and trajectory of vegetation development. Over time total richness increased and the two capping materials became more similar as a result of migration from surrounding areas and between plots. Species losses also occurred at the reclamation site; however this generally led to a reduction in the cover of annuals as well as non-native species in relation to total cover. The FFM had a greater portion of forest species establishing from distant sources likely due to more suitable soil conditions, whereas the PMM showed a greater propensity for natural tree establishment. To allow for the migration of new and desirable species onto reclamation sites, factors such as the surrounding vegetation, suitability of capping material, and the availability of open growing space must be considered. All of these factors play a large role in helping determine initial vegetation development and community composition.

Although the application of fertilizer in this study had little influence on the initiation of vegetation on this reclamation site, it will likely have a long term negative effect in the fertilized plots of the FFM. These plots experienced rapidly increasing graminoid cover throughout this study which could have prolonged negative effects on the recovery of a forest canopy and understory community. Additionally, while fertilizer application on the PMM did not appear to have any negative effects on early vegetation development, there were no observable benefits over the control treatment.

Currently FFM and PMM capping materials are looked at separately; however this study supports the notion that FFM could help increase species richness on PMM through an island effect. While many annuals migrated from the FFM to the PMM, three forest species, which had previously only been found in the FFM, also established on the PMM. This could prove particularly important for vegetation development if there is no nearby seed source in the area.

Further research on species migration and interactions between these two capping materials could help address the abundance of PMM as a reclamation material.

Tables

Table 3-1: Average initial soil characteristics for the forest floor material (FFM) and peat mineral mix (PMM) capping materials measured prior to fertilization. Bold p-values were significant at α=0.05 and different letters indicate a significant difference in means between capping materials (n=5). "*" Indicates non-normal data so a non-parametric test (Wilcoxon Mann-Whitney test) was used to analyze the data.

Response Variable	Unit	P-value	FFM	PMM
pH	-	0.859	6.64	6.67
Sand	%	**0.036**	58.22	52.28
Silt	%	0.070	28.01	33.53
Clay	%	0.921	13.76	14.19
TKP	%	0.073	0.01	0.02
TKN	%	0.066	0.12	0.32
Conductivity*	(dS m^{-1})	0.766	0.14	0.13
Available nitrate-N*	mg/kg	0.143	3.51	3.75
Available phosphate-P*	mg/kg	**0.005**	6.18	3.41
Available potassium-K*	mg/kg	**0.018**	78.75	65.53
Available sulfate-S*	mg/kg	**0.025**	11.04	16.70

Table 3-2: Average soil total available N, P and K for capping material and fertilizer treatments. Values are the sum of two PRS probe burials (early June to late July and late July to early September) for each 2011 and 2012 (n=5). P-values are shown for the main effects of capping material and fertilizer regime as well as their interaction. Bold values are significant at α=0.05.

	2011			2012		
Nutrients	**Capping Material**	**Fertilizer**	**Interaction**	**Capping Material**	**Fertilizer**	**Interaction**
Total N	**0.0001**	**<0.0001**	**0.0164**	0.4942	0.7682	0.2579
Total P	0.9685	**<0.0001**	0.5769	0.3566	**0.0007**	0.4533
Total K	**<0.0001**	0.1911	0.3313	**<0.0001**	**0.0025**	**0.0552**

Table 3-3: ANOVA results of average soil total available N, P and K for capping material and fertilizer treatments. Values are the sum of two PRS probe burials (early June to late July and late July to early September) for each 2011 and 2012 (n=5). Different letters indicate a significant difference in means between treatments (For all treatment effects see Table 3-2).

(A) 2011	FFM	PMM	s.e.
Total K	171^{a}	29^{b}	16.27

	Control	Low	High	Slow	s.e.
Total P	3^{b}	19^{a}	25^{a}	3^{b}	3.51

	FFM					PMM				
	Cont	Low	High	Slow	s.e.	Cont	Low	High	Slow	s.e.
Total N	40^{c}	232^{bc}	639^{b}	196^{bc}	125.17	41^{c}	746^{b}	1459^{a}	427^{bc}	88.96

(B) 2012	FFM	PMM	s.e.
Total N	16^{a}	20^{a}	4.51

	Control	Low	High	Slow	s.e.
Total P	2^{b}	7^{b}	26^{a}	3^{b}	3.51

	FFM					PMM				
	Cont	Low	High	Slow	s.e.	Cont	Low	High	Slow	s.e.
Total K	110^{b}	162^{b}	338^{a}	132^{b}	39.09	14^{b}	24^{b}	64^{b}	37^{b}	39.09

Figures

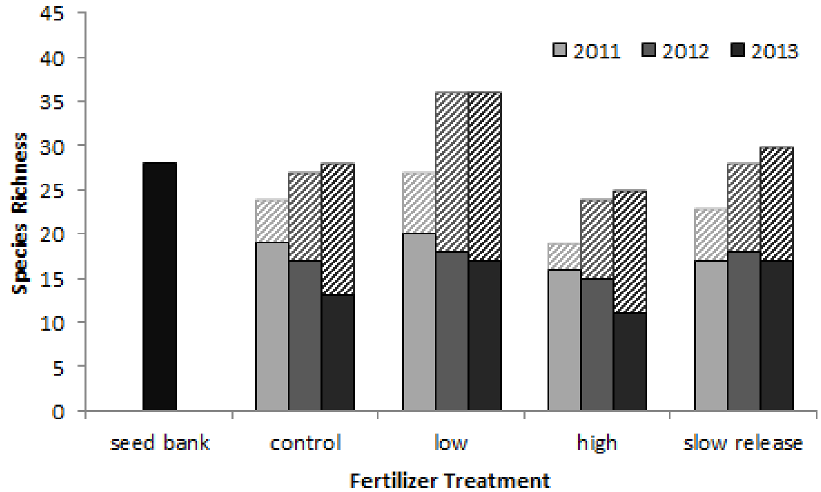

Figure 3-1: Total species richness of species in the different fertilizer treatments divided into species that were present (solid grey bars) and absent (striped grey bars) from the original seed bank in the FFM between 2011 and 2013 (n=5). Richness was measured in 10 1 m^2 plots per fertilizer treatment. The singular black bar indicates the total seed bank richness identified in the seed bank study (a total soil area of 5.2 m^2).

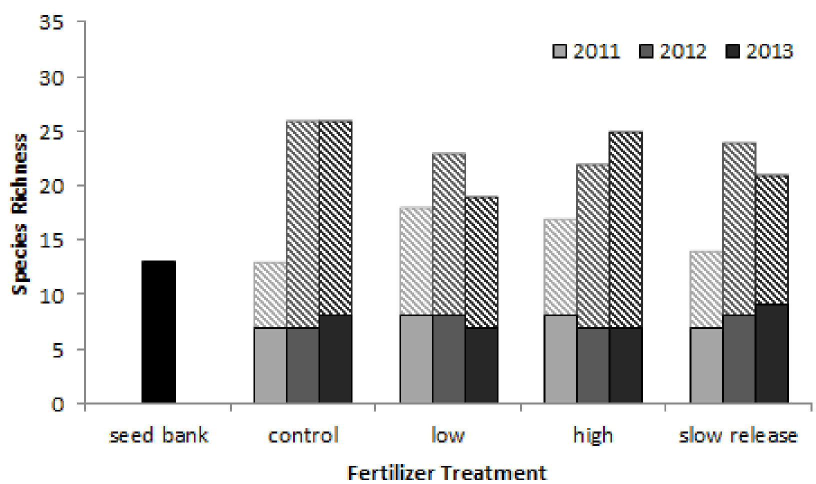

Figure 3-2: Total species richness of species in the different fertilizer treatments divided into species that were present (solid grey bars) and absent (striped grey bars) from the original seed bank in the PMM between 2011 and 2013 (n=5). Richness was measured in 10 1 m^2 plots per fertilizer treatment. The singular black bar indicates the total seed bank richness identified in the seed bank study (a total soil area of 5.2 m^2).

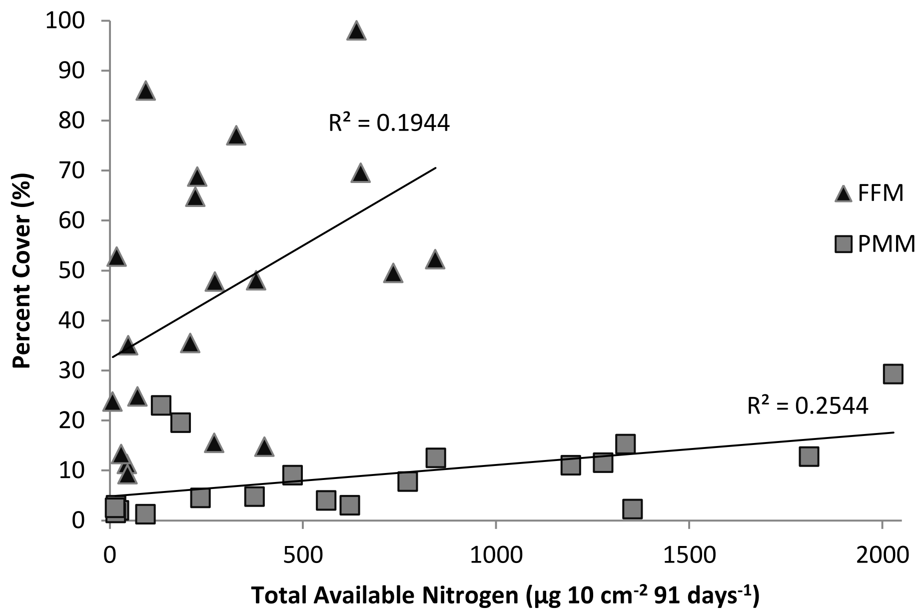

Figure 3-3: Relationship between average total cover and total available nitrogen for all fertilizer treatment plots in the FFM and PMM capping materials in 2011. Lines of best fit and R^2 values are displayed for each capping material (both regressions were significant at α=0.05).

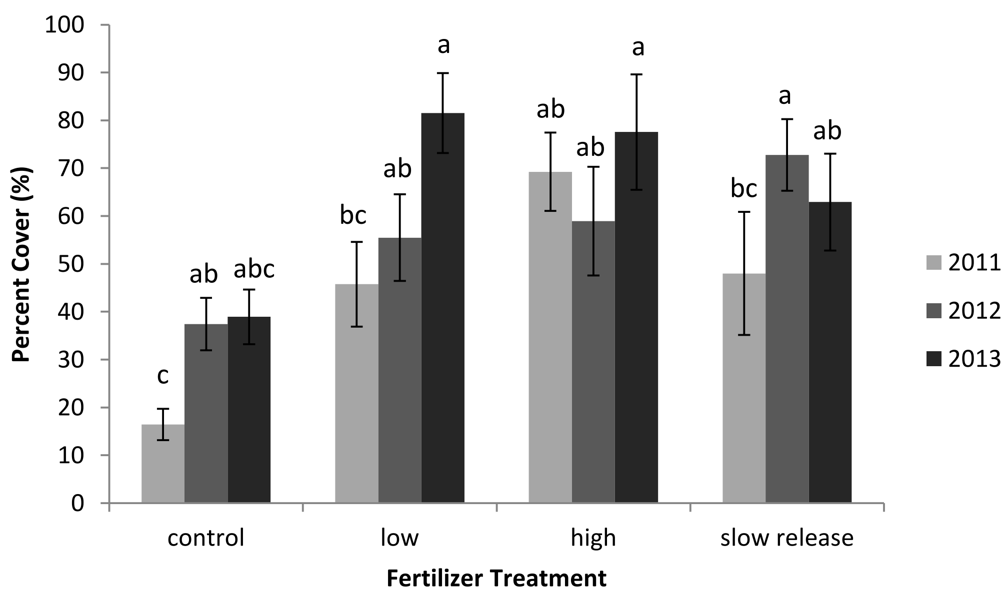

Figure 3-4: Average total cover by fertilizer treatment in each year for FFM plots. Error bars are standard error of the mean (n=5) and bars that have different letters have significantly different means (α=0.05).

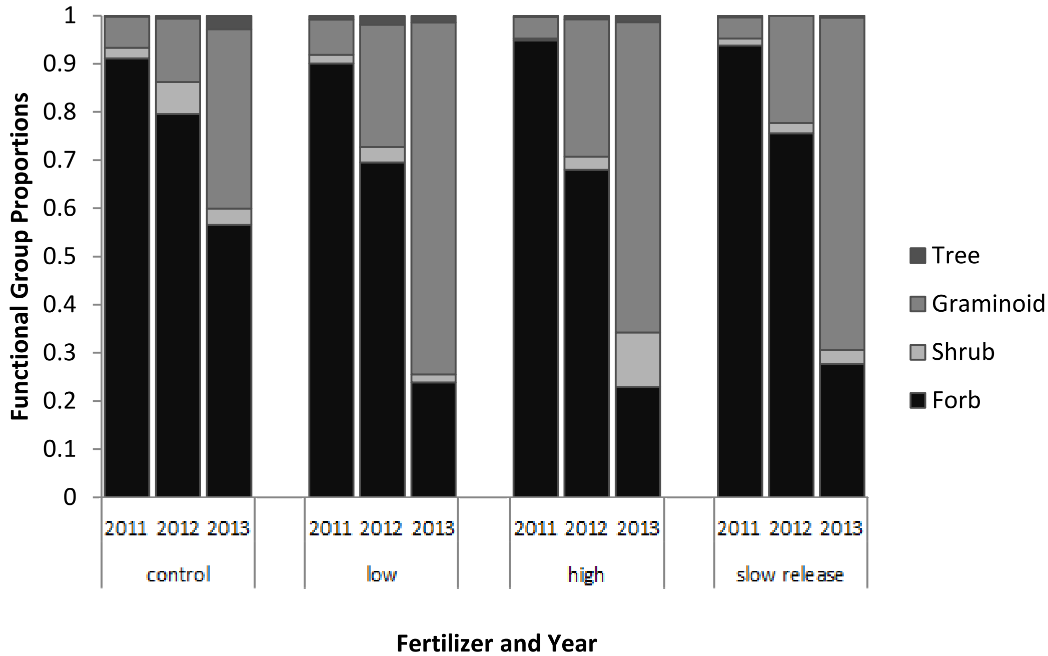

Figure 3-5: Functional group cover as a proportion of total cover in the FFM by fertilizer treatments in each year.

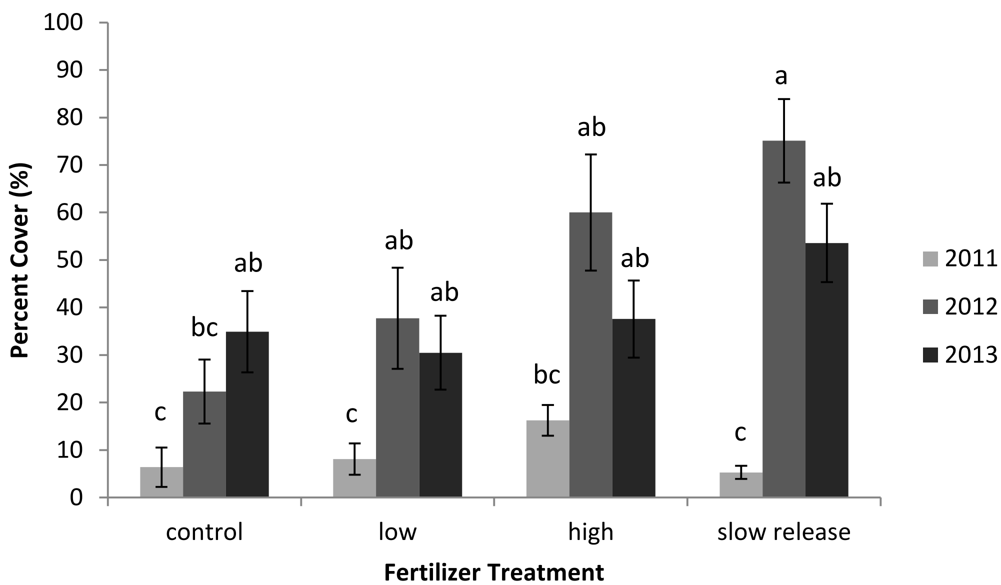

Figure 3-6: Average total cover by fertilizer treatment in each year for PMM plots. Error bars are standard error of the mean (n=5) and bars that have different letters have significantly different means (α=0.05).

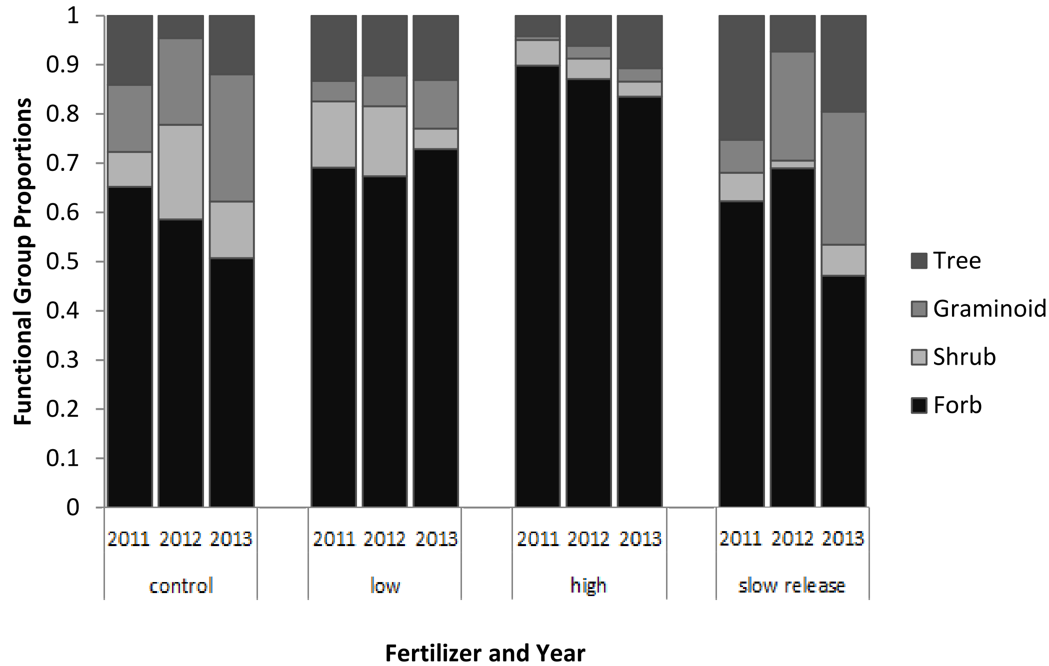

Figure 3-7: Functional group cover as a proportion of total cover in the PMM by fertilizer treatments in each year.

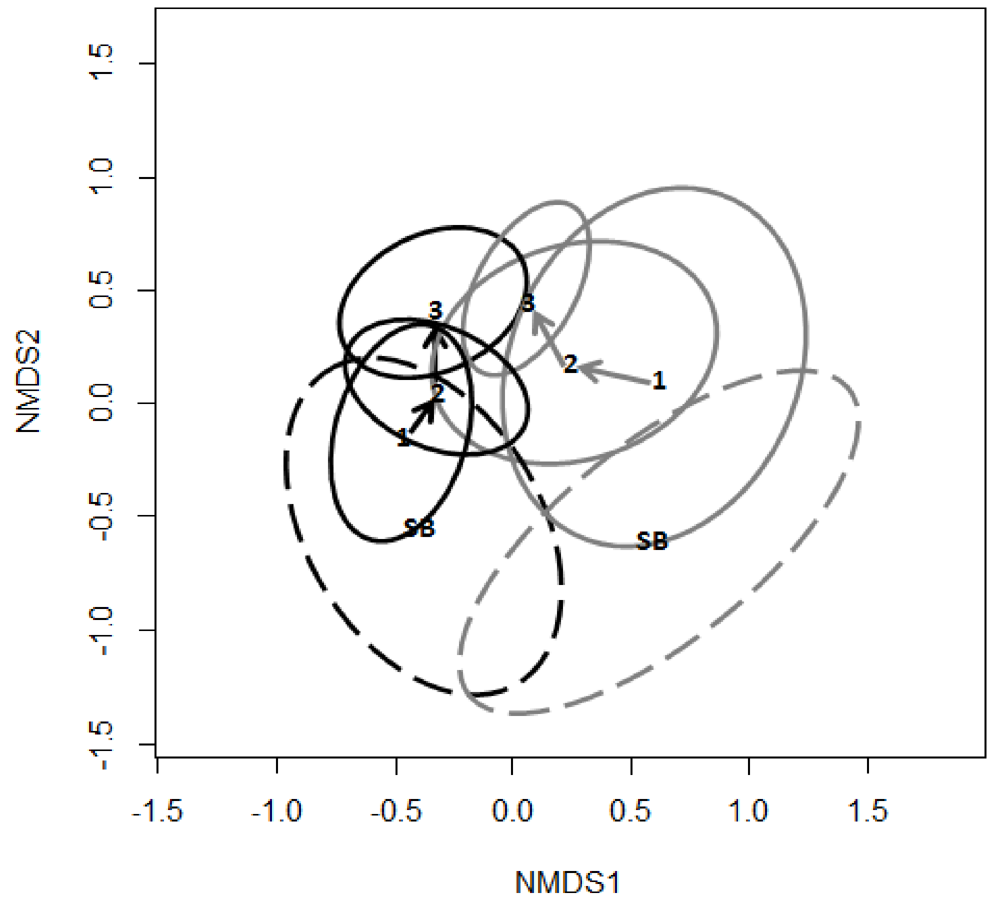

Figure 3-8: Results of an NMDS ordination for the original seed bank study and all three growing seasons at the reclamation site. Ellipses delineate a 95% confidence interval around plots of the different capping materials (black = FFM and grey = PMM) for their original seed banks (dashed lines) and each growing season at the reclamation site (solid lines). The centroid of each ellipse indicates the corresponding time period (SB = seed bank, 1 = 2011, 2 = 2012, and 3 = 2013) and arrows display the yearly shifts in community composition at the reclamation site.

Chapter 4: Discussion and Conclusions

4.1 Research summary

The main objective of this thesis was to investigate the impacts of two different reclamation techniques on initial tree and understory vegetation development. The first technique examined the use of a cover crop to promote early tree growth and survival by reducing competition from ruderal species on soil which was salvaged and directly placed at two different depths. The second technique compared the effects of controlled release fertilizer (CRF) and immediately available fertilizer (IAF) on initial vegetation development in two different capping materials.

In the first study, I looked at initial growth and mortality of *Populus tremuloides*, *Pinus contorta*, and *Picea glauca* when grown with yellow sweet clover (*Melilotus officinalis*) as a cover crop at two different soil salvage and placement depths (15 cm and 40 cm). Results of this study suggest that sweet clover does not benefit initial tree seedling performance; although a deeper soil salvage and placement depth does benefit performance. Sweet clover did not reduce initial growth of tree seedlings; however, mortality was increased by approximately 10% for all species combined in the first year following planting. In contrast, sweet clover provided the benefit of reducing the cover of perennial ruderal species, such as *Trifolium pratense*, and acted as a physical barrier to reduce ungulate browsing of *Pinus contorta*. The 40 cm soil salvage and placement depth had an overall positive impact on tree growth and experienced less mortality than the 15 cm depth despite both treatments reaching roughly 100% total cover of vegetation after only two growing seasons. The main difference between the two depth treatments was that the 15 cm depth had higher densities of competitive species which accounted for the majority of total cover in the plots. In contrast, the 40 cm depth treatment experienced greater dilution of the seed bank and this appeared to reduce the density of competitive species.

The second study examined the impacts of using different fertilizer regimes (*control*, *low* (250 kg/ha IAF), *high* (500 kg/ha IAF), and *slow release* (670 kg/ha CRF)) on the initial vegetation development of stockpiled FFM and PMM capping materials. Results suggested that fertilizer does not affect average species richness or the expression of the seed bank in either capping material; however total cover was affected. Both capping materials responded to the addition of nitrogen with increased total cover, but this effect was fleeting and by 2013 there were no differences in total cover for either capping material. In the FFM, fertilizer application is expected to have a long term negative effect as plots which received fertilizer showed a decrease in the portion of forbs over time and a corresponding increase in the portion of graminoids. It was found that initial vegetation establishment is primarily influenced by the propagule bank of the capping material used; however, over time, total richness increased and the two capping materials became more similar as a result of migration from surrounding areas and between plots. Throughout this study, in both capping materials there was a decline in cover of annual and non-native species as a proportion of total cover, whereas the proportional cover of trees and graminoids increased. In contrast, it was only for the FFM that the proportions of shrubs and forest species increased over time. By 2013, half of the indicator species in the FFM were forest species and while the PMM did not have any forest understory species as indicators, it did have a native boreal tree species. These differences suggest that the two capping materials are on different recovery trajectories and may require different amounts of time to recover.

While these are two very different studies, they both shed light on how initial reclamation practices influence the early trajectories of forest reclamation sites as related to the development of tree seedlings and understory vegetation. In particular, practices which initially promote the dominance of only a few species can hinder ecosystem recovery by providing too much competition for tree seedlings and not allowing enough room for the establishment or migration of desirable forest understory species.

4.2 Research applications

For the goal of forest restoration in central Alberta, I would advise to salvage forest floor material to a depth which corresponds to the effective rooting zone at the site (i.e. 40 cm in our study). At a greater salvage depth, the dilution of the seed bank and nutrients resulted in less initial competition which improved growth and survival of the planted seedlings. Also at this site, it was found that suckers of *Populus tremuloides* had better regeneration, growth, and survival in the greater cm depth (Wachowski 2012). Furthermore, *Populus tremuloides* seedlings which naturally established from seed grew the tallest on convex microsites in the greater cm depth, although at both depths establishment was highest on concave microsites with a mineral-organic substrate (Schott et al. 2014). After three growing seasons at this site, species richness was not affected by the salvage depth; however, non-native species cover was lower and understory forest species cover was higher at the greater depth (Macdonald et al. unpublished). Combined, these results suggest that forest restoration is naturally occurring on a more desirable trajectory and at a faster rate when forest floor salvaged at a greater depth. This was attributed to increased root-soil contact, in the case of *Populus tremuloides* root suckering (Wachowski 2012), as well as dilution of the seed bank from admixing with the lower mineral horizons (Fair 2011). This created conditions conducive to suckering and, in the first year, reduced species establishment and thus competition on the reclamation site. Additionally, dilution of the seed bank reduced the dominance of a few aggressive species which permitted planted tree seedlings to perform better early on and allowed space for desirable tree and understory species to establish or migrate in. It is therefore essential to consider the effects of species composition on long term forest restoration and not just initial establishment and cover.

Additionally, I would suggest that tree seedlings be planted on reclamation sites as soon as possible following the direct placement of salvaged forest floor material. High mortality rates observed in this study can be attributed to the delayed planting of tree seedlings which allowed for greater competition from previously established vegetation. I would also recommend that tree seedlings be planted at the same time sweet clover is seeded. In this study sweet clover slightly increased mortality of all tree seedlings combined and had a neutral impact on overall height growth. It is thought that in the shallower depth, sweet clover only added to the competition already faced by the tree seedlings and at the greater depth it acted more as a replacement competitor. Therefore it is conceivable that due to high establishment of plant cover, which can be associated with directly placing soil, the addition of a cover crop is not a beneficial reclamation practice and would potentially be more useful on stockpiled soils where it could help reduce weed problems. However, keeping in mind that in this study the trees were planted when sweet clover was mature and formed a canopy, it is still feasible that sweet clover could be a suitable cover crop. As a biennial, sweet clover alternates between an open canopy (immature phase) and a closed canopy (mature phase), beginning with an open canopy in the year it is seeded. In this study, tree seedlings had better annual growth in sweet clover plots during the open canopy year than those in control plots, suggesting a potential advantage to using sweet clover. The only drawback to the biennial life cycle is that during the open canopy year, ruderal annuals are also able to establish; however, these species are generally less competitive and persistent.

For stockpiled FFM and PMM capping materials in the Suncor study, the addition of fertilizer (250 kg/ha IAF, 500 kg/ha IAF, and 670 kg/ha CRF) in the spring following site set up cannot be recommended. Although average species richness and the expression of the seed bank were not influenced by fertilization, the total cover tended to be higher in response to fertilization for both capping materials in the first growing season (2011). Over time, fertilization, at all treatment levels, reduced the proportion of forbs (in relation to total cover) and increased the proportion of graminoids particularly in the FMM. If graminoids continue to increase and dominate these plots in the future, overall richness and particularly the presence of desirable forest species might be reduced. Additionally, following the third growing season there were no differences in total cover between fertilizer treatments for either capping material which suggesting that the effect of fertilization was brief and overall had little benefit. However, the increased plant cover likely allowed for the retention and future cycling of soil nutrients.

Initial establishment of vegetation was found to be most strongly influenced by the propagule banks of each capping material and the PMM generally had fewer species and lower total cover. This resulted in better outplanting performance of *Populus tremuloides* seedlings on the PMM in the first two growing seasons likely due to less competition compared to the FFM plots (Schott 2013). However, species migration from adjacent sites and neighbouring treatment plots resulted in some promising vegetation developments in both capping treatments over time. Therefore, I would suggest that the proximity of natural sites or more mature reclamation sites could be useful as a seed source and benefit new reclamation areas, particularly when using stockpiled capping materials. Throughout the duration of this study, species richness increased and community composition changed, mainly as a result of migration from the surrounding area and between capping materials. This likely occurred as a result of bare ground which allowed space for species migration and establishment.

Managing reclamation sites to allow for some bare ground is also recommended. I would suggest the use of FFM alone or in combination with PMM to act as a potential seed source and as a seed receptor for the establishment of forest understory species. While the PMM appeared to be a better substrate for the natural establishment of boreal tree species, the FFM was a better substrate for establishing forest understory forb and shrub species. The natural establishment of understory species is important because of the high costs and lack of commercially available seeds for these species (Lanoue and Qualizza 2000). Therefore, using FFM which preferably has not been stockpiled, and/or maintaining a natural forested area nearby are currently the best methods for establishing forest understory species in a cost effective manner.

The findings of these studies are limited to the site conditions and interpretation of results from the three-year monitoring period. Forest restoration is an extensive and lengthy process which requires the consideration of many different variables and therefore would benefit from more long term and multi-disciplinary studies. However, in general, the studies presented highlight the importance of initial reclamation practices in determining the early trajectory of forest restoration. For example, both studies had reclamation practices which promoted the establishment and dominance of grass species by the third growing season (e.g. the shallower soil salvage and placement depth and fertilizer application on the FFM). The long-term implications of these practices were beyond the scope of this thesis; however, it is conceivable that the dominance of grasses will prolong forest ecosystem recovery. Therefore, longer monitoring of these sites and others will help us to better understand the long term consequences associated with initial reclamation practices.

4.3 Future research and study limitations

Future forest restoration projects would benefit from additional research into the use of different soil salvage and placement depths for directly placed FFM in the boreal forest region. While it was apparent that the deeper salvage and placement depth produced more desirable outcomes than the shallow salvage and placement depth, there is the potential that salvaging soil at a deeper depth and then placing it at a shallower depth would have the same effects (Wachowski 2012). If this technique provided the same benefits to *Populus tremuloides* suckers, planted seedlings, and forest understory vegetation, then smaller donor sites could be used to reclaim larger areas. Although, depending on suitable soil conditions, salvaging to a deeper depth is not always feasible so future research should also consider exploring optimal salvage and placement depths for areas with shallow mineral soil horizons.

Additionally, the use of a cover crop which can imitate early canopy closure warrants further research. With its diffuse, biennial canopy, sweet clover showed potential for being able to benefit tree seedlings; however, to verify this there would need to be a study in which tree seedlings were planted at approximately the same time as sweet clover was seeded. Another potential option for future research would be to seed sweet clover at a lower density as it established well and appeared to only add to total competition or act as a replacement competitor at the seeding density used. However, one limitation of this study was that tree plots covered a larger area (2,400 m^2) than the vegetation assessment plots (96 m^2) and the two plots did not always overlap. To improve this study and draw stronger conclusions on the effect of sweet clover as a cover crop on tree seedling performance, tree and vegetation plots would need to cover the same area.

Alternatively, research could continue to investigate the potential of using a perennial, early successional native forb as a cover crop or even multiple native cover crop species. The use of fireweed (*Chamerion angustifolium* (L.) Holub) as a cover crop was unsuccessful in this study due to low establishment following seeding. Therefore, further research into successful establishment techniques for fireweed and other native forbs is needed. For example, a new technique is currently being tested which examines the potential for growing fireweed in the same plugs as white spruce (*Picea glauca* (Moench) Voss) seedlings (Schoonmaker et al. unpublished). Early successional native forbs could prove to be beneficial cover crops as they are generally of shorter stature than sweet clover and thus planted seedlings, such as *Populus tremuloides*, would not be negatively affected by shading from these species. Furthermore, there would be less concern of a native forb persisting in the restored ecosystem. The use of two or three native cover crop species could be investigated as another option because this would add to species diversity, community resilience, and also create more heterogeneity in competition by reducing the dominance of a single species. However, this option would potentially be better for stockpiled soils, which tend to have lower initial establishment. Also, further research which looks at species compatibility, seeding rates, and establishment success would be needed.

In terms of fertilizer application, reclamation practices could benefit from further research into the timing of application as well as the rate and blend of fertilizer applied. Fertilizer application in the first growing season following tree planting results in high nutrients losses and/or increased competition from nitrophilous species. While the CRF did have the ability to reduce losses by prolonging the release of nutrients at the reclamation site, only planted *Populus tremuloides* seedlings showed any benefit from this (Schott 2013). One limitation of this study was that the CRF was only applied at a rate which matched the amount of nitrogen in the high IAF. Therefore, it is unknown if a lower rate of CRF would have benefited early vegetation development. Delaying fertilizer application or applying less fertilizer may be more favorable options for allowing forest understory species the ability to become established and migrate onto a site by reducing the dominance of only a few species. Additionally, delaying fertilizer application would allow planted seedlings a chance to develop larger root systems which could take up more of the applied nutrients and reduce losses. More research into the application of site specific fertilizers is also necessary due to the range of nutrient availability in the capping materials used. For example, reclamation soils (PMM in particular) typically have increased nitrogen content (Rowland et al. 2009). Therefore applying fertilizer high in nitrogen to capping materials which contain adequate amounts of nitrogen could be more wasteful than beneficial. However, a study by Pinno et al. (2012) found that the application of N-P-K fertilizer resulted in the greatest height of *Populus tremuloides* on PMM, whereas on the FFM there were no differences in height between P-K, N, and N-P-K fertilizer applications. This suggests that the ratio of nutrients as well as the potential for growth limitations from other essential nutrients also needs to be considered when applying fertilizer.

Another area which requires further research is the use of nutrient loaded seedlings on reclamation sites. These seedlings are of higher quality, are able to grow as well as non-nutrient loaded seedlings which receive fertilizer, and thus can reduce the amount of fertilizer applied to reclamation sites (Schott 2013). Additionally, this reduces the losses associated with broadcast applying fertilizer as well as the increased competition from dominant nitrophilous species.

Whether topsoil is directly placed following salvaging or stockpiled, it can remain a valuable source of propagules for reclamation. This is important for the establishment of forest understory species and merits further research. The use of directly placed capping materials is often a preferred choice for accelerating ecosystem recovery, although this is often not possible. Further research looking at species migration between capping materials and from nearby forested areas is needed. Due to the generally small seed bank and lack of upland forest species present in the PMM, it would be beneficial to study the potential for migration and establishment of forest understory species onto PMM. Furthermore, if there is not a natural forest nearby, research should continue to explore the possibility of using FFM as a seed source for the PMM by either placing some nearby or mixing the two capping materials together. Mixing the two capping materials could prove to be the most beneficial method for increasing the natural establishment of *Populus tremuloides* and *Populus balsamifera* from seed, providing forest understory species, and maintaining moderate to low cover initially.

One of the main limitations in this study for assessing early vegetation development on FFM and PMM capping materials was the lack of reference communities. When using stockpiled soil, the donor site is no longer accessible, so comparisons could not be made with the preexisting vegetation of the two capping materials in this study. Additionally, the area sampled was only 80 m^2 and one experimental block had to be excluded because it was improperly set up during soil placement. Reference communities as well as the additional block and a larger sampling area would have strengthened this study. This is especially true in the third growing season (2013) which was a particularly wet year and six plots had to be taken out of the data set because they were partially covered by standing water.

The future of reclamation needs to shift away from a focus on short term soil stabilization goals and instead integrate cover crops and fertilizer regimes which are compatible with tree seedlings and enable survival and growth of desirable understory species without impeding long term restoration. To do this, we need more studies which look at either planted or naturally regenerating trees and understory species together, instead of the more traditional approach of focusing predominantly on each of these parts separately. It is my hope that this research will help to build on the current knowledge and understanding of how to effectively and efficiently restore disturbed areas of the boreal forest back to functioning forest ecosystems. Furthermore, this research is not only relevant to the large mining areas in Alberta but can also be applied to other areas of the circumpolar boreal forest where extensive industrial disturbance is common.

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Appendices

Appendix I. Species which germinated during the seed bank study, their classification, and if they were present in the forest floor material (FFM) and/or the peat mineral mix (PMM). Growth forms included grass (G), forb (F), or shrub (S); origin was either native to Alberta (N) or introduced (I); and forest species were either characteristic of a mature forest understory (F) or other species (O). Rare species (*) had only 1 individual in the seed bank study.

Rare	Seed bank Species	Growth Form	Origin	Forest Species	FFM	PMM
*	*Achillea millefolium* L.	F	N	F		X
	Aster sp.	F	N	F	X	
	Barbarea orthoceras Ledeb.	F	I	O	X	
*	*Betula* sp.	S	N	F	X	
*	*Cardamine pensylvanica* Muhl. ex Willd.	F	N	O	X	
	Carex sp.	G	N	O	X	X
	Chenopodium album L.	F	N	O	X	
	Chenopodium capitatum (L.) Asch.	F	N	O	X	
*	*Cirsium arvense* (L.) Scop.	F	I	O	X	
	Corydalis aurea Willd.	F	N	O	X	
*	*Dracocephalum parviflorum* Nutt.	F	N	O	X	
	Epilobium palustre L.	F	N	O	X	X
	Equisetum sp.	F	N	O		X
	Fragaria virginiana Duchesne	F	N	F	X	
*	*Galium boreale* L.	F	N	F	X	
*	*Galium triflorum* Michx.	F	N	F	X	
	Geranium bicknellii Britton	F	N	O	X	
	Grass sp.	G	-	O	X	X
	Hieracium umbellatum L.	F	N	O	X	
	Juncus bufonius L.	G	N	O	X	
	Lathyrus ochroleucus Hook.	F	N	F	X	
*	*Ledum groenlandicum* Oeder	S	N	F		X
	Polygonum aviculare L.	F	I	O	X	
	Polygonum persicaria L.	F	I	O	X	
	Potentilla norvegica L.	F	N	O	X	X
	Ranunculus sceleratus L.	F	N	O		X
	Rubus idaeus L.	S	N	O	X	
*	*Rumex maritimus* L.	F	N	O	X	
	Salix sp.	S	N	O		X
	Sonchus arvensis L.	F	I	O	X	
	Stellaria media (L.) Vill.	F	I	O	X	X
	Typha latifolia L.	F	N	O		X
	Urtica dioica L.	F	N	O	X	X
	Vicia americana Muhl. ex Willd.	F	N	F	X	
*	*Vaccinium oxycoccos* L.	S	N	O		X
	Total Species Richness				**28**	**13**

Appendix II. Results of Indicator Species Analysis on vegetation composition data (percent cover by species) in the seed bank and for each growing season at the reclamation site. Given are species with their indicator values (I.V.) and significance (P-values). The analysis focused on the main effect of capping material. * indicates species which are native to Canada and ^ indicates species which are characteristics of a mature boreal forest understory.

Year	Treatment	Species	I.V.	P-value
Seed bank	FFM	*Potentilla norvegica**	0.920	0.0002
		Carex sp.*	0.917	0.0002
		*Geranium bicknellii**	0.894	0.0002
		Grass sp.	0.695	0.0100
		*Chenopodium album**	0.671	0.0012
		*Chenopodium capitatum**	0.671	0.0010
		Polygonum persicaria	0.671	0.0010
		*Rubus idaeus**	0.548	0.0224
	PMM	*Epilobium palustre**	0.830	0.0046
		*Equisetum arvense**	0.632	0.0048
2011	FFM	*Geranium bicknellii**	0.975	0.0002
		*Potentilla norvegica**	0.972	0.0002
		Carex sp.[1]*	0.949	0.0002
		Polygonum persicaria	0.898	0.0002
		*Rubus idaeus**	0.883	0.0002
		Kochia scoparia	0.842	0.0006
		Grass sp.[2]	0.825	0.0008
		*Corydalis aurea**	0.823	0.0002
		*Dracocephalum parviflorum**	0.742	0.0002
		*Lathyrus ochroleucus**^	0.742	0.0004
		*Chenopodium album**	0.671	0.0014
		*Chenopodium capitatum**	0.671	0.0006
		*Vicia americana**^	0.632	0.0044
		Aster sp.[3]*^	0.592	0.0068
		*Juncus bufonius**	0.592	0.0088
	PMM	*Equisetum arvense**	0.956	0.0002
		*Populus balsamifera**	0.852	0.0002
		Salix sp.[4]*	0.592	0.0096
2012	FFM	*Geranium bicknellii**	0.954	0.0002
		*Potentilla norvegica**	0.871	0.0002
		*Rubus idaeus**	0.871	0.0002
		Sonchus arvensis	0.853	0.0074
		Carex sp.[1]*	0.844	0.0024
		Polygonum persicaria	0.840	0.0056
		*Dracocephalum parviflorum**	0.736	0.0044
		*Vicia americana**^	0.696	0.0006
		*Juncus bufonius**	0.679	0.0066
		*Lathyrus ochroleucus**^	0.671	0.0010
		*Equisetum arvense**	0.955	0.0002

	PMM			
		*Chenopodium album**	0.950	0.0002
		*Populus balsamifera**	0.858	0.0016
		Salix sp.[3]*	0.794	0.0004
		*Epilobium angustifolium**	0.759	0.0140
		*Populus tremuloides**	0.619	0.0150
		Senecio vulgaris	0.548	0.0202
2013	FFM	*Vicia americana**^	0.939	0.0002
		*Agrostis scabra**	0.929	0.0002
		*Geranium bicknellii**	0.890	0.0002
		Carex sp.[1]*	0.888	0.0018
		Sonchus arvensis	0.849	0.0048
		*Lathyrus ochroleucus**^	0.840	0.0002
		*Achillea millefolium**^	0.761	0.0078
		*Fragaria virginiana**^	0.686	0.0028
		*Symphyotrichum ciliolatum**^	0.632	0.0182
		Polygonum aviculare	0.542	0.0446
	PMM	*Equisetum arvense**	0.942	0.0002
		*Populus balsamifera**	0.910	0.0004
		*Epilobium angustifolium**	0.895	0.0014
		*Equisetum sylvaticum**	0.860	0.0002
		*Erigeron canadensis**	0.765	0.0254

[1]Includes at least one or more sedge species that could not be identified to species
[2]Includes at least one or more grass species that could not be identified to species
[3]Includes at least one or more willow species that could not be identified to species

Appendix III. New species added to the FFM in the 2012 and 2013 growing seasons. “PMM” indicates that the species was previously found in the PMM capping material, “Seed Bank” indicates that the species was found in the FMM of the seed bank study, and “Other” indicates that the species was not previously encountered.

Species	PMM	Seed Bank	Other
2012			
Campanula rotundifolia L.			X
Erigeron canadensis L.			X
Fragaria virginiana Duchesne		X	
Galium boreale L.		X	
Gentianella amarella (L.) Börner			X
Lactuca serriola L.			X
Plantago major L.			X
Salix sp. L.	X		
Symphyotrichum ciliolatum (Lindl.) Á. Löve & D. Löve			X
Taraxacum officinale F.H. Wigg.	X		
Total	**2**	**2**	**6**
2013			
Aster puniceus (L.) Á. Löve & D. Löve			X
Crepis tectorum L.			X
Geum macrophyllum Willd.			X
Juncus alpinoarticulatus Chaix ssp. *nodulosus* (Wahlenb.) Hämet-Ahti			X
Melilotus sp. Mill.			X
Mertensia paniculata (Aiton) G. Don			X
Parnassia palustris L.			X
Petasites frigidus (L.) Fr. var. *sagittatus* (Banks ex Pursh) Cherniawsky			X
Ranunculus sceleratus L.			X
Rhinanthus minor L.	X		
Ribes oxyacanthoides L.			X
Ribes triste Pall.			X
Rosa acicularis Lindl.			X
Rumex maritimus L.		X	
Trifolium repens L.			X
Typha latifolia L.	X		
Viola adunca Sm.			X
Total	**2**	**1**	**14**

Appendix IV. Seed bank expression under different fertilizer regimes (C=control, L=low, H=high, and S=slow release) in the forest floor capping material. "X" indicates a species was present that year and "-" indicates that a species was present in a previous year but is no longer present.

	2011				**2012**				**2013**			
Species Name	**C**	**L**	**H**	**S**	**C**	**L**	**H**	**S**	**C**	**L**	**H**	**S**
Grass sp.	X	X	X	X	X	X	X	X	X	X	X	X
Aster sp.	X	X		X	X	X		X	X	X		X
Barbarea orthoceras Ledeb.		X				X		X		X		X
Betula sp.												
Cardamine pensylvanica Muhl. ex Willd.										X		X
Carex sp.	X	X	X	X	X	X	X	X	X	X	X	X
Chenopodium album L.	X	X	X	X	X	X	X	X	X	-	-	X
Chenopodium capitatum (L.) Asch.	X	X	X	X	-	-	-	-	-	-	-	-
Cirsium arvense (L.) Scop.												
Corydalis aurea Willd.	X	X	X	X	X	-	-	-	-	-	-	-
Dracocephalum parviflorum Nutt.	X	X	X	X	X	X	X	X	-	-	-	-
Epilobium palustre L.	X	X	X	X	X	X	X	X	X	X	X	X
Fragaria virginiana Duchesne						X		X		X	X	X
Galium boreale L.						X				-		
Galium triflorum Michx.												
Geranium bicknellii Britton	X	X	X	X	X	X	X	X	X	X	X	X
Hieracium umbellatum L.	X	X	X	X	X	X	X	X	-	-	-	-
Juncus bufonius L.	X	X	X	X	X	X	X	X	-	X	-	X
Lathyrus ochroleucus Hook.	X	X	X	X	X	X	X	X	X	X	X	X
Polygonum aviculare L.	X	X			X	X	X		X	X	X	X
Polygonum persicaria L.	X	X	X	X	X	X	X	X	X	X		X
Potentilla norvegica L.	X	X	X	X	X	X	X	X	X	X	X	X
Rubus idaeus L.	X	X	X	X	X	X	X	X	X	X	X	X
Rumex maritimus L.										X		
Sonchus arvensis L.	X	X	X	X	X	X	X	X	X	X	X	X
Stellaria media (L.) Vill.				X				X				-
Urtica dioica L.	X	X			-	-			-	-		
Vicia americana Muhl. ex Willd.	X	X	X		X	X	X	X	X	X	X	X
Total	**19**	**20**	**16**	**17**	**17**	**19**	**15**	**18**	**13**	**17**	**11**	**17**

Appendix V. New species added to the PMM in the 2012 and 2013 growing seasons. "FFM" indicates that the species was previously found in the FFM capping material, "Seed Bank" indicates that the species was found in the PMM of the seed bank study, and "Other" indicates that the species was not previously encountered.

Species	FFM	Seed Bank	Other
2012			
Aster sp. L.	X		
Chenopodium album L.	X		
Chenopodium capitatum (L.) Asch.	X		
Dracocephalum parviflorum Nutt.	X		
Erigeron canadensis L.			X
Juncus bufonius L.	X		
Lactuca serriola L.			X
Matricaria perforata Mérat	X		
Rhinanthus minor L.			X
Senecio vulgaris L.			X
Typha latifolia L.		X	
Vicia americana Muhl. ex Willd.	X		
Total	**7**	**1**	**4**
2013			
Achillea millefolium L.		X	
Mitella nuda L.			X
Picea glauca (Moench) Voss			X
Stellaria media (L.) Vill.		X	
Symphyotrichum ciliolatum (Lindl.) Á. Löve & D. Löve	X		
Total	**1**	**2**	**2**

Appendix VI. Seed bank expression under different fertilizer regimes (C=control, L=low, H=high, and S=slow release) in the peat mineral mix capping material. "X" indicates a species was present that year and "-" indicates that a species was present in a previous year but is no longer present.

	2011				2012				2013			
Species Name	**C**	**L**	**H**	**S**	**C**	**L**	**H**	**S**	**C**	**L**	**H**	**S**
Grass sp.	X	X	X	X	X	X	-	X	X	X	X	X
Achillea millefolium L.									X		X	X
Carex sp. L.	X	X	X	X	X	X	X	X	X	X	X	X
Epilobium palustre L.	X	X	X	X	X	X	X	X	X	X	X	X
Equisetum sp.	X	X	X	X	X	X	X	X	X	X	X	X
Ledum groenlandicum Oeder		X	X	X		X	-	-		-	-	-
Potentilla norvegica L.	X	X	X		X	X	X	X	X	X	-	X
Ranunculus sceleratus L.												
Salix sp.	X	X	X	X	X	X	X	X	X	X	X	X
Stellaria media (L.) Vill.	X		X	X	X		X	X	X	X	-	X
Typha latifolia L.							X				X	
Urtica dioica L.		X				X		X		-		X
Vaccinium oxycoccos L.												
Species Richness	**7**	**8**	**8**	**7**	**7**	**8**	**7**	**8**	**7**	**8**	**8**	**7**

Appendix VII. Results of a repeated measures ANOVAs examining the effects of fertilizer (Fert.), year, and their interaction (Int.) on percent cover by species group and richness of all species (per plot) for each capping material. Bold values represent significance at α=0.05.

Species group	**FFM (n=5)**			**PMM (n=5)**		
	Fert.	**Year[1]**	**Int.**	**Fert.**	**Year[1]**	**Int.**
Richness	0.4979	**0.0051**	0.2373	0.9824	**<0.0001**	0.4947
Total cover	**0.0120**	**0.0020**	**0.0232**	**0.0320**	**<0.0001**	**0.0105**
Forb	**0.0058**	**<0.0001**	**0.0053**	**0.0382**	0.1057	0.4290
Shrub	0.3978	**0.0083**	0.1393	0.4547	0.7061	0.3778
Graminoid	**0.0066**	**<0.0001**	**0.0173**	**0.0356**	**0.0331**	0.5701
Tree	0.4740	**0.0294**	0.2416	0.1654	**0.0238**	0.5254
Non-Native	0.2508	**0.0030**	0.4160	0.3409	**0.0127**	0.1055
Forest	0.1336	**0.0006**	0.3261	0.5804	**0.0354**	0.1537
Annual	0.6751	<0.0001	0.1391	0.2698	<0.0001	0.1833

[1]The 2013 dataset had an unequal sample size due to partial flooding of plots which resulted in the omission of six plots.

Appendix VIII. List of all identified species along with their origin, growth form, if they are present in a mature forest understory, and what year they were observed at the reclamation site. Growth forms included grass (G), forb (F), or shrub (S); origin was either native to Alberta (N) or introduced (I); and forest species were either characteristic of a mature forest understory (F) or other species (O). Rare species are indicated with a "*" and were only observed in one plot throughout the duration of the study. Species with a "-" were included in the genus category for species richness counts because they could not be identified in the first year. Year indicates in which growing season(s) a species was observed.

Rare	Species Name	Growth Form	Origin	Forest Species	Year
	Grass sp.	G	N,I	O	1,2,3
	Achillea millefolium L.	F	N	F	1,2,3
	-*Agrostis scabra* Willd.	G	N	O	3
	-*Alopecurus aequalis* Sobol.	G	N	O	2,3
	-*Arctagrostis latifolia* (R. Br.) Griseb.	G	N	O	3
	Aster sp. L.	F	N	F	1,2,3
*	-*Aster puniceus* (L.) Á. Löve & D. Löve	F	N	F	3
	Barbarea orthoceras Ledeb.	F	I	O	1,2,3
*	*Betula glandulosa* Michx.	S	N	F	1,2,3
	-*Calamagrostis canadensis* (Michx.) P. Beauv.	G	N	O	1,2,3
*	*Caltha palustris* L.	F	N	O	1,2
*	*Campanula rotundifolia* L.	F	N	F	2
	Cardamine pensylvanica Muhl. ex Willd.	F	N	O	3
	Carex sp.	G	N	O	1,2,3
	-*Carex atherodes* Spreng.	G	N	O	3
	-*Carex aquatilis* Wahlenb.	G	N	O	3
	-*Carex disperma* Dewey	G	N	O	3
	-*Carex siccata* Dewey	G	N	O	3
	Chenopodium album L.	F	N	O	1,2,3
	Chenopodium capitatum (L.) Asch.	F	N	O	1,2
	Cirsium arvense (L.) Scop.	F	N	O	SB
	Collomia linearis Nutt.	F	N	O	1,2
	Cornus canadensis L.	F	N	F	1
	Corydalis aurea Willd.	F	N	O	1,2
	Crepis tectorum L.	F	I	O	3
	Dracocephalum parviflorum Nutt.	F	N	O	1,2,3
	Epilobium angustifolium L.	F	N	O	1,2,3
	Epilobium ciliatum Raf. ssp. *ciliatum*	F	N	O	SB
	Epilobium palustre L.	F	N	O	1,2,3
	Equisetum sp.	F	N	O	1,2,3
	Equisetum arvense L.	F	N	O	1,2,3
	-*Equisetum pratense* Ehrh.	F	N	O	2,3
	-*Equisetum sylvaticum* L.	F	N	O	2,3
	Erigeron canadensis L.	F	N	O	2,3
	-*Festuca saximontana* Rydb.	G	N	O	2,3

	Fragaria virginiana Duchesne	F	N	F	2,3
*	*Galium boreale* L.	F	N	F	2
	Galium triflorum Michx.	F	N	F	SB
*	*Gentianella amarella* (L.) Börner	F	N	O	2
	Geranium bicknellii Britton	F	N	O	1,2,3
	Geum macrophyllum Willd.	F	N	O	3
	Hieracium umbellatum L.	F	N	O	1,2
	-*Hordeum jubatum* L.	G	N	O	3
*	*Juncus alpinoarticulatus* Chaix ssp. *nodulosus* (Wahlenb.) Hämet-Ahti	G	N	O	3
	Juncus bufonius L.	G	N	O	1,2,3
	Kochia scoparia (L.) Schrad.	F	I	O	1,2
	Lactuca serriola L.	F	I	O	2,3
	Lathyrus ochroleucus Hook.	F	N	F	1,2,3
	Ledum groenlandicum Oeder	S	N	F	1,2,3
	-*Lolium persicum* Boiss. & Hohen. ex Boiss.	G	I	O	2,3
	Matricaria perforata Mérat	F	I	O	1,2,3
	Melilotus sp. Mill.	F	I	O	3
*	*Mertensia paniculata* (Aiton) G. Don	F	N	F	1,2
*	*Mitella nuda* L.	F	N	F	3
*	*Parnassia palustris* L.	F	N	O	3
*	*Petasites frigidus* (L.) Fr. var. *sagittatus* (Banks ex Pursh) Cherniawsky	F	N	F	3
*	*Picea glauca* (Moench) Voss	T	N	O	3
	Plantago major L.	F	I	O	2,3
	-*Poa palustris* L.	G	N	O	2,3
	Polygonum aviculare L.	F	I	O	1,2,3
	Polygonum persicaria L.	F	I	O	1,2,3
	Populus balsamifera L.	T	N	O	1,2,3
	Populus tremuloides Michx.	T	N	O	1,2,3
	Potentilla norvegica L.	F	N	O	1,2,3
	Ranunculus sceleratus L.	F	N	O	3
	Rhinanthus minor L.	F	N	O	2,3
*	*Ribes glandulosum* Grauer	S	N	F	2
*	*Ribes oxyacanthoides* L.	S	N	F	3
*	*Ribes triste* Pall.	S	N	F	1,3
*	*Rosa acicularis* Lindl.	S	N	F	3
	Rubus idaeus L.	S	N	O	1,2,3
*	*Rubus pubescens* Raf.	F	N	F	1,2,3
*	*Rumex maritimus* L.	F	N	O	3
	Salix sp.	S	N	O	1,2,3
	-*Salix myrtillifolia* Andersson	S	N	O	2,3
	-*Salix planifolia* Pursh	S	N	O	2,,3
	Senecio vulgaris L.	F	I	O	2,3
	Sonchus arvensis L.	F	I	O	1,2,3
	Stellaria media (L.) Vill.	F	I	O	1,2,3

	-*Symphyotrichum ciliolatum* (Lindl.) Á. Löve & D. Löve	F	N	F	2,3
	Taraxacum officinale F.H. Wigg.	F	I	O	1,2,3
*	*Trifolium repens* L.	F	I	O	3
*	*Typha latifolia* L.	F	N	O	2,3
	Urtica dioica L.	F	N	O	1,2,3
	Vaccinium oxycoccos L.	S	N	O	SB
	Vicia americana Muhl. ex Willd.	F	N	F	1,2,3
*	*Viola adunca* Sm.	F	N	F	3
	Total Species Richness = 77 species				