Effects of Biochar, Fertilizer and Shelter Treatments on the Vegetation Development following Coal Mine Reclamation

By

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Abstract

Poor quality cover soil, a lack of propagules, and availability of suitable microsites can be serious challenges in the re-vegetation success of surface mines. In my thesis research, I examined the response of total cover, species richness and community composition of colonizing vegetation on a harsh coal mine reclamation site in Alberta, Canada to the presence of planted aspen seedlings and amendment with biochar and fertilizer. Additionally, I explored the effects of shelter type (wood vs. brick), along with the effects of aspect and distance from shelter on survival and growth of four seeded native species and on density of volunteer vegetation. Results suggested planted aspen seedlings and fertilizer increased both cover and richness of colonizing vegetation in the first growing season, but increasing the amount of fertilizer did not result in additional effects. The application of biochar did not influence the cover of colonizing vegetation but did result in decreased richness; no synergistic effects of biochar and fertilizer were found. The provision of shelter generally improved survival of seeded species and density of volunteer species although effects differed somewhat among species and with aspect and distance from shelter. Lastly, within the short timeframe of this study there was no clear evidence that either type of shelter was preferable. These findings emphasize the importance of planting aspen, amending with use of fertilizer and using shelter as means to improve poor cover soil quality, create suitable microsites and encourage natural re-vegetation on challenging reclamation sites. Continued research regarding biochar use and its combined effects with fertilizer on poor substrates is needed.

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Chapter 1: Introduction

1.1 Surface Mining and Reclamation

The boreal forest is a circumpolar vegetation zone consisting primarily of cold-tolerant forests and treeless areas such as wetlands, lakes and rivers (Brandt et al. 2013). The boreal forest spans across Canada from Newfoundland to the Yukon Territory covering approximately 552 million hectares (Jetté et al. 2009, Brandt et al. 2013, Natural Resources Canada 2016). The boreal forest supplies numerous ecosystem services (e.g., climate regulation, nutrient cycling, fresh water) as well as both renewable and nonrenewable resources (e.g. timber, minerals, oil and gas) which are the basis of industrial development and economic growth (Brandt et al. 2013, Venier et al. 2014). Accessing these resources often requires a disturbance, which in the case of belowground resources may require large scale surface mining.

Surface mining poses a unique challenge for reclamation; all vegetation, surface soils and overburden material are removed from the site during the mining process. Once mining is complete, reclamation begins by backfilling the excavated area with appropriate material that is available. Next the landform is reshaped and topographical features are recreated; the surface is then capped with suitable material which can facilitate soil development and then revegetation is done. The mining processes of removing, handling and stockpiling materials can have impacts on hydrology, soil structure and various soil properties (e.g. pH, texture, bulk density, microbe populations, loss of propagule bank viability, nutrient availability); these effects in turn can impede reclamation and eventually lead to delays in ecosystem recovery (Rokich et al. 2000, Sheoran et al. 2010, Mackenzie and Naeth 2010, Franklin et al. 2012, Macdonald et al. 2012, 2015a).

Reclamation often aims to return the disturbed landscape back to a state where it is capable of supporting similar land uses as it was prior to the disturbance (Alberta Environment 1999), however isn't the case in all jurisdictions. Poor topsoil and subsoil material quality can present a variety of limitations and challenges for reclamation including a lack of nutrients, low pH, poor water holding capacity, high electrical conductivity, and little to no propagule source available for natural vegetation recovery (Soil Quality Criteria Working Group 1987, McMillan et al. 2007, Sheoran et al. 2010, Mackenzie and Naeth 2010). Often, natural regeneration is the only option available for re-establishment of the herbaceous community since native species seeds and seedlings are often not commercially available (Brown and Naeth 2014) and salvaged forest floor material – which serves as a source of native species propagules – may not be available.

In order to improve sites and promote ecosystem recovery, landforms are usually capped with poor subsoil material and then with suitable surface soil materials if they are available. These materials are often salvaged during the mining process and stockpiled separately on site for future use or directly placed onto an active reclamation area. These reconstructed soils create the foundation for the reclaimed ecosystem, providing water storage and filtration, nutrient cycling and supply, and a seed bed for vegetation establishment (Macdonald et al. 2012, Pinno et al. 2012). Stockpiling of salvaged soil materials prior to use as a capping material has negative impacts including reduced nutrient availability and a lack of viable plant propagules, seeds, and soil biota (Sheoran et al. 2010, Mackenzie 2013, Macdonald et al. 2015b).

Many reclaimed sites only have nutrient poor subsoil material at the surface, which pose many challenges for re-establishment of vegetation (Sheoran et al. 2010, Macdonald et al. 2015a). In order to manage soil quality and increase vegetation development on reclamation sites, various organic and inorganic soil amendments can be applied. Additionally, increased landscape heterogeneity though the creation of microsites can significantly benefit plant species establishment (Bradshaw 1997, Frouz et al. 2008, Mudrák et al. 2010, Mackenzie and Naeth 2010, Gilland and McCarthy 2014, Schott et al. 2014, Macdonald et al. 2015a). Vegetation establishment is imperative for ecosystem recovery and thus a better understanding of how to manage poor soils is critical.

1.2 Revegetation

Forest restoration historically focused on revegetation, with less attention on reconstructing functional soils, landscape variation, and using native species or tree seedlings (Grant and Koch 2007, Macdonald et al. 2015a). Revegetation often begins by planting tree seedlings in order to aid in the development of a natural understory and functioning plant-soil nutrient cycle (Macdonald et al. 2012, Pinno and Errington 2015). Trembling aspen (*Populus tremuloides*

Michx.) is a wide-spread shade-intolerant deciduous tree species in the boreal biome, and is considered to be a pioneer species (Rowe 1972). Aspen is considered to be relatively tolerant to stress including cold air temperatures and drought, and as such is adapted to persist on harsh sites (Lieffers et al. 2001). Planting trees on reclamation sites is an effective means to restore disturbed areas back to forested landscapes, as a continuous tree canopy may suppress growth of unwanted vegetation, and increase litter production (Macdonald et al. 2012).

Establishment of understory vegetation is also important in order to create functioning forest ecosystems. Vegetation establishment on reclaimed sites is often dependent on site characteristics such as nutrient availability, available soil moisture, propagule availability, forest canopy structure, and propagule dispersal from surrounding areas (Macdonald et al. 2012, 2015a, Pinno and Errington 2015). Natural colonization by native forest species is often slow, especially in reclaimed areas located in mainly agricultural landscapes where nearby seed sources are comprised of weedy and agronomic species. In order to address this issue, native vegetation can be seeded or salvaged forest floor material can be used as a capping material since it will also act as a source of plant propagules; however, these options are not always available during reclamation. The development of tree canopy structure can help facilitate the transition from early successional, shade intolerant species to more native shade tolerant species. Understory plant communities in the boreal forest ecosystem are an important contributor to species diversity, and to ecological processes such as nutrient cycling through litter production and root turnover (Gilliam 2007, Muller 2014, Macdonald et al. 2015b).

1.3 Management Options

Management intervention through the use of organic amendments and fertilizer may be necessary to achieve reclamation targets on low productivity sites (Prach and Hobbs 2008). Challenges with soil quality include poor soil structure, low nutrient availability and low organic matter - particularly when no topsoil is applied (Frouz et al. 2009, Turcotte et al. 2009, Larney and Angers 2012, Turcotte and Quideau 2012, Macdonald et al. 2015a, Smebye et al. 2016). These challenges may cause poor water holding capacity as well as other issues, which can be addressed by use of organic amendments or fertilizer. Organic amendments add nutrients and organic matter thereby enhancing the chemical, physical and biological properties of poor soil

(Larney and Angers 2012). Organic amendments used in reclamation include livestock manure, biosolids, crop residue, pulp and paper mill sludge, food processing wastes, wood chips and shavings (Larney and Angers 2012, Government of Alberta 2013), and biochar (Han et al. 2016). Inorganic amendments like immediately available or controlled-release fertilizers add nutrients such as nitrogen, phosphorus, potassium and various micronutrients (e.g. Zn, Fe, Mg, Cu) (Hangs et al. 2003, Larney and Angers 2012, Pinno et al. 2012, Sloan and Jacobs 2013, Macdonald et al. 2015b). Chemical amendments such as lime may be applied to acidic soils, or gypsum and sulfur may be applied to alkali soils (Redley and Utkaeva 2009). Alternatively, to increase microsite heterogeneity on reclamation sites micro-topographical features such as hummocks or woody debris may be used. These microsites may provide shelter, create a range of environmental conditions, and improve vegetation growth on reclamation sites (Brown and Naeth 2014, Gilland and McCarthy 2014, Macdonald et al. 2015a).

1.3.1 Soil Amendments

Biochar is a carbon rich product produced through anaerobic thermal degradation of organic materials at temperatures less than 700°C (Lehmann and Joseph 2015). It is often produced specifically for application to soil for agronomic or environmental management purposes (Brown et al. 2015, Chia et al. 2015). Biochar characteristics vary based on the originating material used to make it; this can include forestry products, agricultural residue, or municipal waste products (Brown et al. 2015, Chia et al. 2015). Physical and chemical characteristics that may be used to evaluate biochar include pH, ash content, pore volume, and surface area (Sohi et al. 2010) and are used to determine the suitability of biochar for soil amendment use. Previous studies have shown that using biochar as a soil amendment can improve soil nutrient availability and nutrient retention, increase microbial activity, enhance vegetation growth, and improve various physical, chemical and biological soil properties (Glaser et al. 2002, Lehmann et al. 2006, Chaganti et al. 2015, Jay et al. 2015, Lehmann and Joseph 2015). Physical properties of soil that are affected by biochar include bulk surface area, pore size distribution and density which, in turn, affects the porosity and texture of the soil (Fellet et al. 2011, Chia et al. 2015).

Inorganic amendments such as fertilizer are often used to replace nutrients within reclaimed soils. At the beginning stages of revegetation, fertilizer application has been shown to increase early growth rates of vegetation and tree seedlings (Rowland et al. 2009, Sloan et al. 2016). This is usually done through broadcast fertilization using immediately available fertilizers (Pinno et al. 2012, Sloan et al. 2016); however, these types of fertilizers have been shown to have low recovery rates for planted trees seedlings because of uptake by competing vegetation, immobilization in the soil, or leaching (Sloan and Jacobs 2013, Sloan et al. 2016). In recent years, controlled release nitrogen fertilizers have been used in order to improve nutrient use efficiency (nutrient recovery) by vegetation and tree seedlings as compared to immediately available fertilizers (Arnon 1992, Sloan and Jacobs 2013). Controlled release fertilizers have the ability to last for 3-18 months, which is able to match plant demand and improve use efficiency throughout the growing season rather than have a flush of nutrients available at the time of application (Sloan et al. 2016). There is potential for using biochar and fertilizer amendments together in order to increase the retention of nutrients provided by fertilizer in the soil, as biochar amendments can increase soil porosity, plant-available water, and nutrient retention capacity (Chan et al. 2007, Lei and Zhang 2013, Xie et al. 2016).

Using organic and inorganic amendments such as biochar and fertilizer to improve poor quality substrate material may increase organic matter content and nutrients within the soil and in turn provide a more suitable substrate to sustain vegetation (Larney and Angers 2012). It is clear that biochar and fertilizer amendments improve soil properties and enhance vegetation growth; however, there is interest in combining these two amendments together to explore the effects on vegetation establishment as this has not been thoroughly explored in reclamation practices. When no surface soils are available for capping it is particularly important to employ methods to improve the reclaimed soils in order to support vegetation community establishment.

1.3.2 Shelter and Microsites

One management strategy for creating and increasing microsite variation across the reclamation surface is the placement and use of shelter. As previously mentioned natural regeneration may be the only available option for vegetation re-establishment, as salvaged forest floor material and seed for native species may not be available (Brown and Naeth 2014). In order to facilitate natural regeneration suitable growing conditions are required; one way of generating these conditions and additionally creating microsite variation across the soil surface is the placement and use of shelter. Macdonald et al. (2015a) notes that small scale variation on reclamation sites in the boreal region may play a role in seedling establishment and early success. Shelter can be created from hedgerows or windrows, nurse plants such as shrubs, young tree seedlings or taller vascular plants, large rocks or woody debris.

There are many potential benefits provided by shelter including a direct physical barrier from wind, increased shade and protection from sun exposure, assistance in the capture of windblown seeds, improve water used by vegetation, and indirect microsite effects (Carlsson and Callaghan 1991, Bird et al. 1992, Cleugh 1998, Nuberg 1998, Whisenant 2002, Buchman et al. 2007). Microsites occurring on northern aspects are generally cooler and have increased soil moisture whereas the southern aspects are generally warmer and have drier soils (Åström et al. 2007, González-Alday et al. 2008, Macdonald et al. 2015a, Måren et al. 2015). Shade provided by shelter may produce more favourable soil temperatures (Turner et al. 1966). Microsite and shading effects vary with angle of incidence with the sun, as the area shaded will be larger at latitudes further away from the equator (Rosenberg et al. 1983a, Adams 2010, Måren et al. 2015). The interactions between aspect of shelter and distance from shelter on microsite conditions can play a large role in determining vegetation growth and establishment. The favourable soil, moisture and temperature conditions created from shelter may positively influence colonizing vegetation, eventually promoting the establishment of species aligning with native boreal forest understory development. However, responses by vegetation will vary depending on species requirements.

A shelter material of particular interest for use on reclamation sites is coarse woody debris (CWD). In forest ecosystems CWD includes logs, large fallen branches, buried wood, dead roots, as well as standing dead trees and snags (Harmon et al. 1986). In natural forest ecosystems and on reclaimed sites woody debris fulfills critical ecological roles of providing shelter for many organisms, as well as contributing to ecosystem functions such as nutrient cycling, carbon storage, retention of moisture, facilitating vegetation establishment and productivity, reducing erosion, and contributing to microsite heterogeneity (see Harmon et al. 1986, Lindenmayer et al. 1999, 2002, Pyle and Brown 1999, Debeljak 2006, Grant and Koch 2007, Manning et al. 2013, Brown and Naeth 2014, Macdonald et al. 2015, and Kwak et al. 2016). The deficiency of nutrients in reclaimed soils can be limiting for vegetation establishment and CWD may improve this by contributing nutrients to the soil through decomposition (Harmon et al. 1986, Jia-bing et al. 2005, Zhou et al. 2007, Merganičová et al. 2012, Kwak et al. 2016). Although nutrient levels in CWD are initially low and the decomposition rates are slow, nutrient concentrations do increase over time potentially due to inputs from decomposing fungi and nitrogen fixation, and this can, in turn, enhance soil fertility and productivity (Laiho and Prescott 2004, Zhou et al. 2007, Hagemann et al. 2009, Wiebe et al. 2014).

Placement of CWD on reclamation sites could facilitate vegetation establishment through ameliorating the harsh environmental conditions often found on these sites (Brown and Naeth 2014). It is clear that CWD can act as a sheltering object and this should benefit establishing vegetation. However, its other properties such as moisture retention and provision of nutrients may also be important in its role in facilitating vegetation establishment. It is currently unknown if CWD is useful in reclamation simply because it is an object which provides shelter (in which case other types of shelter may be used) or if there is a greater overall effect because of its organic nature that may provide additional benefits to vegetation. This is of interest because CWD may not always be available for use; if its main effect is to provide shelter then this could be achieved using other shelter materials such as rocks, bricks, or creating micro-topography.

1.4 Whitewood Coal Mine Reclamation Site

In Alberta, Canada, the boreal forest is the largest natural region and is comprised of eight natural subregions that extend over 58% of the province (Natural Regions Committee 2006). The dry mixedwood forest subregion in northern Alberta is the second largest subregion; the southern extent of this region is largely comprised of cultivated landscapes (~ 70%) producing barley, canola and forage crops (Natural Regions Committee 2006). Significant land use from agriculture as well as grazing, forestry, oil and gas development, and coal mining occurs within the dry mixedwood forest subregion (Natural Regions Committee 2006); however, the landscape contains forest remnants that are aspen (*Populus tremuloides* Michx.) dominated and scattered with white spruce (*Picea glauca* Moench.) (Rowe 1972, Natural Regions

Committee 2006). Warm summers and milder winters compared to other boreal subregions are common, with approximately 70% of annual precipitation falling from April to August. Typical soils found in this area are well drained Orthic Gray Luvisols under aspen forests, Dark Gray Luvisols usually in cultivated areas, and Brunisols on sandy sites (Natural Regions Committee 2006). This research site is in a unique area, as the dry mixedwood subregion acts as a transitional zone between the Mixedwood boreal forest and the Aspen Parkland area.

In Alberta, surface mining for coal or bitumen deposits involves a similar process as previously described in Section 1.1, however legislature requires separate soil salvage, stockpiling and placement of topsoil and subsoil materials (Macdonald et al. 2012, Government of Alberta 2014). The main reclamation objective for these areas is to return disturbed landscapes back into sustainable landscapes or ecosystems with equivalent land capability or to specific pre-disturbance conditions (Leatherdale et al. 2012, Government of Alberta 2014). In a mined area that is being reclaimed back to both agriculture and forest areas, more of the high quality surface and top soils might be used for the agricultural areas, leaving little for the forest areas. Specifically, at the Whitewood coal mine in Wabamun, Alberta, reclamation using topsoil material involved direct placement onto areas which were then cultivated and fertilized for fall rye or oat cover crop production. The shortage of topsoil at the mine site resulted in remaining areas which were to be reclaimed to a forested landscape with no topsoil replacement and only the uncapped subsoil of unconsolidated geological material at the surface (Kuchmak 2015, personal communnication). In order to reclaim the mined area back into a forest landscape, challenges associated with creating suitable soil conditions for vegetation growth on the poor substrate material using various management strategies must be met.

1.5 Objectives

The goal of the research presented in this thesis was to develop an understanding of the role planted aspen trees, biochar and fertilizer amendments and coarse woody debris treatments play in the initial development of vegetation at a reclamation site with harsh growing conditions as a result of poor substrate material.

In Chapter 2, I investigate the effects of tree cover and the amendments of controlled release fertilizer and biochar on the early development of colonizing vegetation on a poor soil substrate. I assessed the cover and richness of colonizing vegetation over two growing seasons, and also explored potential interactions between the amendments.

In Chapter 3, I explore the use of shelters in to facilitate revegetation on harsh reclamation sites, and examine whether shelter material type (wood *versus* brick) and orientation (North-South *versus* East-West) affect native forb and grass seedling establishment and performance.

In Chapter 4, I provide a synthesis of my work along with suggestions for areas requiring future research and recommendations on how the results of this research can be incorporated into future reclamation practices in Alberta.

Chapter 2: Responses of colonizing vegetation to planted aspen, biochar, and fertilizer amendments on a reclaimed coal mine

2.1 Introduction

Surface mining is a severe large-scale disturbance where vegetation, surface soil materials and overburden material are removed to access the resource below. Once mining is complete, reclamation begins by replacing the overburden back onto the disturbed areas followed by contouring the land; the overburden material is then typically capped with subsoil material and topsoil that was salvaged during the initial excavation phase if they are available, and lastly revegetated (Macdonald et al. 2015a). In Alberta, re-establishing forest vegetation involves the use of specific native species and seedlings appropriate for the region (Alberta Environment 2003). The southern extent of the dry mixedwood boreal forest biome in central Alberta is rich in natural resources such as wood, oil and gas, and coal. This area is largely comprised of cultivated landscapes (~70 percent) producing barley, canola and forage crops (Natural Regions Committee 2006). However, the landscape contains forest remnants of this biome that are aspen (*Populus tremuloides* Michx.) dominated and scattered with white spruce (*Picea glauca* Moench.) (Rowe 1972, Natural Regions Committee 2006). Public awareness of the value of these forest areas has been increasing and as a result of this, forest restoration as part of mine reclamation has become an important aspect in recent years.

The mining process can result in poor soil conditions after mining is complete as it changes the soil horizon and structure, microbe populations, and nutrient cycles through the removal and stockpiling of materials (Alberta Environment 1999, Sheoran et al. 2010). Poor topsoil and subsoil quality can present a variety of limitations and challenges for reclamation including a lack of nutrients, low pH, poor water holding capacity, high electrical conductivity, and little to no propagule source available for natural recovery (Soil Quality Criteria Working Group 1987, McMillan et al. 2007, Sheoran et al. 2010, Mackenzie and Naeth 2010). Issues surrounding these poor soil characteristics during reclamation include difficulty for plant establishment, and increased risks of soil erosion and nutrient loss (Sheoran et al. 2010). In a mined area that is being reclaimed back to both agriculture and forest areas, more of the high quality surface and top soils might be used for the agricultural areas, leaving little for the forest areas. If top soil is unavailable during reclamation then uncapped subsoils may become the surface material in which reclamation occurs upon, bringing about the challenges for creating a new functional landscape.

Management intervention through the use of organic amendments and fertilizer may be necessary to achieve reclamation targets on low productivity sites (Prach and Hobbs 2008). Challenges with soil quality include poor soil structure, low nutrient availability and particularly when no topsoil is applied, low organic matter (Frouz et al. 2009, Turcotte et al. 2009, Larney and Angers 2012, Turcotte and Quideau 2012, Macdonald et al. 2015a, Smebye et al. 2016). These challenges may cause poor water holding capacity as well as other issues, which can be addressed by organic amendments or fertilizer. Amendments that add organic matter, including peat or forest floor capping materials (see Mackenzie and Naeth 2010, Turcotte and Quideau 2012), are often used in Alberta, however, are not always available during reclamation. Other organic amendments include gypsum, crop residue, and biochar (Larney and Angers 2012, Government of Alberta 2013, Han et al. 2016). Inorganic amendments like immediately available or controlled-release fertilizers add nutrients such as nitrogen, phosphorus, potassium and various micronutrients (Larney and Angers 2012, Pinno et al. 2012, Sloan and Jacobs 2013, Macdonald et al. 2015b). Alternatively, to increase microsite heterogeneity on reclamation sites micro-topographical features such as hummocks or woody debris may be used. These microsites may provide shelter, create a range of environmental conditions, and improve vegetation growth on reclamation sites (Brown and Naeth 2014, Gilland and McCarthy 2014, Macdonald et al. 2015a).

Research regarding the use of biochar for agricultural purposes has been growing in the last decade (see Glaser et al. 2002, Barrow 2012, Biederman and Harpole 2013), however there has been little research done on its application in reclamation of forest ecosystems. Biochar is a carbon rich product produced through anaerobic thermal degradation of organic materials at relatively low temperatures (Lehmann and Joseph 2015). Previous studies have shown that using biochar as a soil amendment can improve soil nutrient availability and nutrient retention, increase microbial activity, enhance vegetation growth, and improve various physical, chemical and biological soil properties (Glaser et al. 2002, Lehmann et al. 2006, Chaganti et al. 2015, Jay et al. 2015, Lehmann and Joseph 2015). Biochar can also affect the bulk surface area, pore size distribution and density of the soil; these, in turn, affect the porosity and texture of the soil (Fellet et al. 2011, Chia et al. 2015).

Inorganic amendments such as fertilizer are often used to replace nutrients within reclaimed soils. In recent years, controlled release nitrogen fertilizers have been used in order to improve nutrient use efficiency (nutrient recovery) by vegetation and tree seedlings as compared to immediately available fertilizers (Arnon 1992, Sloan and Jacobs 2013). Controlled release fertilizers have the ability to last for 3-18 months, which is able to match plant demand and improve use efficiency (Sloan et al. 2016). Research has shown that application of fertilizer makes soils more suitable for vegetation establishment (Prach and Hobbs 2008, Rowland et al. 2009, Macdonald et al. 2015a, 2015b, Sloan et al. 2016). The use of fertilizers may increase species cover and richness initially, however decreases in richness may occur after time through a loss of nitrophilous species (Gilliam 2006); competition may cause diversity to decrease as available soil resources increases (Rajaniemi 2002). There is potential for using biochar and fertilizer amendments together in order to increase the retention of nutrients provided by fertilizer in the soil, as biochar amendments can increase soil porosity, plant-available water, and nutrient retention capacity (Chan et al. 2007, Lei and Zhang 2013, Xie et al. 2016). To improve the use of organic and inorganic amendments in reclamation practices, a better understanding of their effects on soil conditions and vegetation establishment is essential.

Revegetation during reclamation often begins by planting tree seedlings in order to aid in the development of a natural understory and functioning plant-soil nutrient cycle (Macdonald et al. 2012, Pinno and Errington 2015). Trembling aspen (*Populus tremuloides* Michx.) is a wide-spread shade-intolerant deciduous tree species in the boreal biome, and is considered to be a pioneer species (Rowe 1972). Aspen is considered to be relatively tolerant to stress including cold air temperatures and drought, and as such is adapted to persist on harsh sites (Lieffers et al. 2001). Planting trees on reclamation sites is an effective means to restore disturbed areas back to forested landscapes, as a continuous tree canopy may trap wind dispersed seeds, suppress growth of unwanted vegetation, and increase litter production (Bullock and Moy

2004, Macdonald et al. 2012). Creating functional forest ecosystems also includes establishing a diverse array of herbaceous species in the understory. Vegetation establishment on reclaimed sites is often dependent on site characteristics such as nutrient availability, available soil moisture, propagule availability, forest canopy structure, and propagule dispersal from surrounding areas (Macdonald et al. 2012, 2015a, Pinno and Errington 2015). Natural colonization by native forest species is often slow, especially in reclaimed areas located in mainly agricultural landscapes where nearby seed sources are comprised of weedy and agronomic species. In order to address this issue, native vegetation can be seeded or forest floor material can be used as a capping material since it will also act as a source of plant propagules (Mackenzie and Naeth 2010); however, these options are not always available during reclamation. The development of the tree canopy structure can help facilitate the transition from early successional, shade intolerant species to more native shade tolerant species (Macdonald et al. 2012).

This research aimed to examine the effects of planted aspen seedlings and application of a controlled-release fertilizer and biochar have on the early development of colonizing herbaceous vegetation on a reclaimed coal mine site that had poor quality soil substrate. It was hypothesized that:

1. Areas with planted aspen seedlings will have greater vegetation cover and species richness compared to areas with no planted trees.
2. Increasing fertilizer application will result in increased vegetation cover and decreased richness; however, these differences might not be long-lived.
3. The use of a biochar amendment will increase vegetation cover and species richness compared to areas with no biochar; and the application of both fertilizer and biochar will result in the highest vegetation cover and decreased species richness.

2.2 Methods

2.2.1 Research Area

Research for this study took place at the Whitewood Coal Mine (53° 33’ N, 114° 29’ W), located near Wabamun, Alberta, Canada approximately 70 km west of Edmonton. Open pit strip mining for coal occurred from 1962 to 2010; reclamation began in the late 1980s with the East Pit Lake (Ross and Hovdebo 1995) and has since occurred progressively across the 1900 hectares of mined land (TransAlta 2014). The goal of reclamation on the site was to return the mined area back to both agriculture and wildland areas. For many decades prior to mining, areas of forests had been cleared and used for pasture and forage crops; however, poorly drained soil and poor soil structure, along with the short growing season caused difficulty with agriculture production (Kuchmak 2015 personal communication). Soil conservation was not a requirement at the beginning of mine operations in 1962; only in 1983 were amendments made to the Land Surface Conservation and Reclamation Act that required topsoil and organic matter salvaging (Sinton 2011). As a result, there was a deficit of top soil by the end of mine life. Areas which were not suitable for productive agriculture use were slated to be reclaimed to wildland areas, with upland forests and open grass lands as the ecosystem targets (Kuchmak 2015 personal communication). Since there were topsoil and subsoil shortages, this wildlands area was recontoured using only the uncapped subsoil of unconsolidated geological material with no placement of surface soil materials (Kuchmak 2015 personal communication). Characteristics of the surface soil to be reclaimed (unconsolidated geological material that comprised the surface substrate) are described in further detail in Table A1.

The study area was located within the Dry Mixedwood natural subregion of the boreal forest which is characterized by aspen (*Populus tremuloides* Michx.), jack pine (*Pinus banksiana* Lamb.) and white spruce (*Picea glauca* (Moench) Voss) forests in pure or mixed stands (Natural Regions Committee 2006). Natural upland soils in this subregion are typically Orthic Gray Luvisols in aspen forests, which have Ae and Bt horizons, and occur when the mean annual soil temperature is less than 8°C. Dark Gray Luvisols are found in cultivated areas, and these have eluvial features with an Ah or Ahe horizon ≥ 5 cm in thickness (Soil Classification Working Group 1998, Natural Regions Committee 2006). The Dry Mixedwood natural subregion experiences the warmest summers and highest growing degree days of the boreal natural subregions. Climate data were gathered from a weather station approximately 30 km southwest of the mine (in the township of Tomahawk Alberta; 53°26'22.000" N, 114°43'06.000" W) (Government of Canada 2016). In 2013, the mean three-day average temperature during the growing season

(May 5 – September 30) was 14.5°C, and growing season total precipitation was 271 mm. During the 2014 growing season, the mean three-day average temperature was 13.2°C, and growing season total precipitation was 376 mm (see Figure B 1). Soil water moisture was continually measured at two depths (10 cm and 30 cm) in each of the six blocks during the experimental period using 5TM soil moisture sensors with EM50 dataloggers (Decagon Devices Inc. Pullman, WA, USA). In the 2013 growing season starting in July, the mean volumetric water content of the soil was 0.029 m3/m3 at a depth of 10 cm, and 0.223 m3/m3 at a depth on 30 cm, while from May 5 – July 27 2014 mean volumetric water content of the soil was 0.071 m3/m3 at a depth of 10 cm, and 0.232 m3/m3 at a depth of 30 cm (see Figure B 2).

2.2.2 Experimental Design

A field experiment with a blocked split-split plot design was set up in 2011 on approximately 3 hectares of land at the Whitewood mine with 6 blocks of 0.50 ha each, each of which included all treatment combinations (Figure 2-1). The treatments included planting of aspen seedlings, application of fertilizer, application of biochar and combinations of these. This study utilized biochar derived from *Pinus contorta* (Lodgepole pine) that was produced by slow pyrolysis with a pH of 7.3; the biochar had total carbon of 56%, total nitrogen of 1.3%, and a bulk density of 232 kg m^3 (Liu 2015). The target application rate of biochar for this study was 2500 kg/ha; however, a final estimated rate of 1800 kg/ha was manually spread on October 2011 across half of each block and was disked into the soil to a depth of 20 cm. Loss of biochar occurred during the application, mostly due to the loss of finer particles through wind erosion. Aspen seedlings (one-year old *Populus tremuloides*) with a plug dimension of 6 cm diameter and 15 cm depth (615A; 340 ml) were provided by the Smoky Lake Forest Nursery Ltd. (Smoky Lake, Alberta, Canada). In May 2012, the seedlings (which had an average initial height of 28±1.5 cm) were planted throughout each research block at a density of 6000 stems/ha; a small area in the center of each block was left unplanted as a control to explore the effect of planted tree seedlings on vegetation establishment. Polyon® controlled-release fertilizer obtained from Agrium Advanced Technologies Direct Solutions (Calgary, Canada) with an 8-9-month nutrient release period (15-9-12-6; N-P-K-S) was used for the fertilizer treatments. Application of the fertilizer occurred in June 2012 at two levels (low and high concentrations, equal to 50 kg N/ha and 100 kg N/ha, respectively) to the downslope half of each block; the remaining areas were left as controls with no fertilizer application.

Together these resulted in four treatment combinations for the areas without planted trees (unamended control, biochar only, low fertilizer only, low fertilizer + biochar) and six treatment combinations in the areas with planted trees (unamended control, biochar only, high fertilizer only, low fertilizer only, high fertilizer + biochar, low fertilizer + biochar); therefore, each block included 10 treatment combinations (Figure 2-1).

2.2.3 Field Vegetation Assessment

Vegetation development was assessed using a total of 46 1 m^2 quadrats randomly placed throughout all of the treatments in each block. There were five quadrats for all treatment combinations in which aspen seedlings were planted and four quadrats per treatment combination in the areas with no trees (Figure 2-2). Sampling took place during July 24-31 in 2013 and 2014. All plants were identified to species and ocular estimates of percent cover were made. When percent cover was less than 1% it was recorded as “trace” but for analysis was given the value of 0.5. A grid pattern outlining 1% increments in cover was present on the quadrat and observers were calibrated to one another to improve accuracy of the cover estimates. Whenever possible, vegetation was identified to species in the field; when that was not possible, the species was collected and later pressed for identification in the lab. Nomenclature for species adheres to the United States Department of Agriculture PLANTS database (USDA 2015). Species were categorized by functional group: life form (forb or graminoid) and distribution in Alberta, Canada (native or introduced) as per the United States Department of Agriculture PLANTS database (USDA 2015). A list of species and their functional groups can be found in Table A 2.

2.2.4 Statistical Analysis

All statistical analyses were conducted using R software, version 3.2.2 (R Core Team 2016). To test for treatment effects on total percent cover, species richness, and community composition, two separate mixed models were used since the treatments were not applied as a full factorial.

In areas where trees were planted, all three levels of fertilizer were present, however in areas where there were no trees planted only the control and low fertilizer treatments were present.

Model 1 was used to examine the influence of year, planted aspen trees, biochar, two levels of fertilizer, and their interactions. This was analyzed as a split-split-split plot design with pooled errors, including block as a random factor and fixed effects as follows: two levels of year (Y) as the main plot, two levels of biochar (B) as the split plot, two levels of fertilizer (F) as the split-split plot, two levels of aspen (T) as the split-split-split plot and all interactions between the fixed effects. Analyses were completed using the lme function from the nlme package (Pinheiro et al. 2016).

Model 2 was used to examine the influence of year, two levels of biochar, three levels of fertilizer, and their interactions. This analysis included data only from the areas in which trees had been planted to capture all three levels of fertilizer treatments (high, low, no). This was analyzed as a split-split plot design with pooled errors, including block as a random factor and fixed effects as follows: two levels of year (Y) as the main plot, two levels of biochar (B) as the split plot, and three levels of fertilizer (F) as the split-split plot and all interactions between the fixed effects. Analyses were completed using the lme function from the nlme package (Pinheiro et al. 2016).

For all analyses, residuals were assessed for conformation to the assumptions of normality and homogeneity of variance. If they were found to violate these assumptions the data were transformed using a ln+1 transformation. Transformations were used for all cover and species richness response variables but when examining the results, I present the untransformed least squared means and standard errors. Alpha was set at 0.05. If any statistically significant effects were found, post-hoc tests were conducted to compare between means as appropriate for the question at hand and with family-wise Bonferroni correction of alpha (Table 2-1). When an interaction was significant comparisons were made between levels of a treatment holding the level of the other treatments constant (e.g. comparison between fertilizer treatments for each year separately). For details see Table 2-1. Analyses were done on all species combined, then separately for the forb and introduced functional groups. Statistical analysis was not done on the native or graminoid functional groups because of their very low cover and richness.

To examine variation in species composition, we used Non-Metric Multidimensional Scaling (NMDS) following the metaMDS procedure from the vegan package in R (Oksanen et al. 2016), with a random starting configuration, a stability criterion on 0.00005, the Bray-Curtis distance measure, and the Wisconsin-style double standardized scaling for results. Ordination graphs were made using the first two dimensions; species that were highly correlated ($r^2 \geq 0.5$) with the ordination axes were overlaid as vectors (Goslee and Urban 2007). Permutational multivariate analysis of variance (PERMANOVA) was used to test for the effects of planted aspen trees, biochar, fertilizer, and their interaction on species composition (following the same models as the ANOVAs including a random term for block) using the adonis function within the vegan package (Oksanen et al. 2016). Indicator species analysis (Dufrêne and Legendre 1997) was used from the multipatt function within the indicspecies package (De Caceres and Legendre 2009) to identify species that were significant indicators for the treatments (trees, biochar and fertilizer). The PERMANOVA and NMDS analyses were run separately for each year.

2.3 Results

Of the 57 identified species, 72% were forbs and 51% were introduced species. Average cover for native species and graminoids was less than 3% each; therefore, no further statistical analyses was conducted for these two functional groups. Overall total cover on the reclamation site in 2013 was 10% and this increased to 11% in 2014. Total species richness on the site (all sample plots combined) was 49 in 2013 decreasing to 43 species in 2014.

2.3.1 Vegetation cover and richness as affected by planting, fertilizer and biochar amendments

When analyzed only for the low fertilizer plots (Model 1), total plant cover was significantly greater in plots with aspen seedlings present than in plots with no seedlings present (Table 2-2A); this effect was significant for all species combined, as well as for the forb and introduced species functional groups (Figure 2-3A). There was a significant year by fertilizer interaction (Table 2-2A); in plots with low fertilizer, total plant cover was higher in 2013 than in 2014 but there was no difference between years for plots in the no fertilizer treatment (Figure 2-3B1). Total plant cover was higher in the low fertilizer treatment than in the no fertilizer treatment in 2013 but not in 2014 (Figure 2-3B2). Biochar had no effect on total plant cover (Table 2-2A). When analyzed only for the planted seedlings plots (Model 2), total plant cover was significantly greater in 2013 than in 2014 (Table 2-2B; data not shown), however biochar or fertilizer treatments had no effect on total cover (Table 2-2B).

When analyzed only for the no and low fertilizer plots (Model 1), there was a significant year by fertilizer interaction (Table 2-2A) for cover both forb and introduced species. In 2013, cover of both forbs and introduced species was significantly higher with the low fertilizer treatment compared with the no fertilizer treatment; however, by 2014 this difference had disappeared (Table 2-3A). Biochar had no effect on forb or introduced plant cover (Table 2-2A). When analyzed for only the planted seedlings plots (Model 2), only effect of year on cover of forbs or introduced species were found (see Table 2-2B, Table 2-4A; data not shown); biochar and fertilizer had no effect on forb or introduced species cover (see Table 2-2B; data not shown).

When analyzed only for the no and low fertilizer plots (Model 1), there were significant year by tree treatment, year by fertilizer, and year by biochar interactions for richness of all species combined (Table 2-2A). In plots with no planted aspen, richness was significantly higher in 2013 than in 2014, but there was no difference between the years in plots with trees planted (Figure 2-4A1). In 2013 there was no difference between the plots with *versus* without planted trees but by 2014 richness was significantly higher in plots with planted aspen seedlings compared to no planted aspen seedlings (Figure 2-4A2). In plots with low fertilizer, richness was significantly higher in 2013 than in 2014 (Figure 2-4B1), and in 2013 richness was significantly higher in plots with low fertilizer compared to no fertilizer (Figure 2-4B2). In plots with biochar present richness was significantly higher in 2013 than in 2014 (Figure 2-5A1), however there was no difference between years for plots in the no biochar treatment. In 2013 there was a difference between plots with *versus* without biochar, but by 2014 there was no longer any difference between these two (Figure 2-5A2). When analyzed only for the planted seedling plots (Model 2) there was a significant year by fertilizer interaction (Table 2-2B). Richness was significantly higher in 2013 than 2014 in plots with either low or high fertilizer but there was no difference between the years in the no fertilizer plots (Figure 2-5B1). In 2013 richness was significantly greater in plots with either low or high fertilizer treatments compared to no fertilizer treatments, while in 2014 there were no differences between the fertilizer treatments (Figure 2-5B2). The biochar treatment had no effect on species richness in Model 2 (Table 2-2B; data not shown).

When analyzed only for the low fertilizer plots (Model 1), species richness of both forb and introduced functional groups had significant year by tree treatment and year by fertilizer interactions (Table 2-2A). For both forb and introduced species in plots with no planted aspen, richness was significantly higher in 2013 than in 2014, however there was no difference between the years in plots with trees planted (Table 2-3B). There was no difference between the plots with *versus* without planted trees in either year for introduced species, yet forb species richness was greater in plots with trees *versus* without trees in 2014 however this was not seen in 2013 (Table 2-3B). For both forb and introduced species in plots with low fertilizer, richness was significantly higher in 2013 than in 2014; however, there was no difference between years in plots with no fertilizer. In 2013 richness was significantly higher with the low fertilizer treatment *versus* no fertilizer, and there were no differences found between low *versus* no fertilizer in 2014 (Table 2-3A). Richness of forb species was not affected by biochar, however there was a year by biochar interaction for introduced species (Table 2-2A). In plots with biochar present, richness was significantly higher in 2013 than in 2014, however there was no difference between the years in plots with no biochar; additionally, there was no difference between plots with *versus* without biochar in either year (Table 2-3C). When analyzed only for the planted seedlings plots (Model 2) there was a significant year by fertilizer interaction for richness of both forb and introduced species (Table 2-2B). Richness was significantly higher in plots with high and low fertilizer in 2013 than 2014, while there was no difference in plots with no fertilizer in 2013 *versu*s 2014 (Table 2-4B). In 2013, richness was significantly higher in plots with either higher or low fertilizer treatments compared to no fertilizer, however there were no longer any differences between plots with high *versus* low *versus* no fertilizer in 2014 (Table 2-4B). The biochar treatment had no effect on richness of forbs or introduced species (Table 2-2B; data not shown).

2.3.2 Community composition as affected by planting, fertilizer and biochar amendments

Overall community composition (cover by species) was significantly influenced by the presence of trees but not by the amendment of soils with biochar or fertilizer (Table 2-5A). The Indicator Species Analysis suggested there were no indicator species for the tree treatments in 2013, but in 2014 four introduced species of forbs and one native grass species were indicators of the plots with planted aspen trees, while one introduced forb species was a significant indicator for the no tree plots (Table 2-6). The NMDS of the 2013 data indicates no distinct separation by tree treatment (Figure 2-6A), and variation along the first axis was driven by only one native forb (*Equisetum arvense* L.) along with three introduced forbs (*Crepis tectorum* L.*, Stachys palustris* L., *Taraxacum officinale* F.H. Wigg.) and one noxious weed (*Tanacetum vulgare* L.) which all loaded to the low end of axis 1. Variation on the second axis was driven by an introduced forb species (*Erysimum cheiranthoides* L.) (Figure 2-6A). In 2014, the NMDS showed some separation between the tree treatments, as the planted aspen plots loaded on the right side of the first axis and spread upwards along the second axis, while the no tree plots spread out across the first axis (Figure 2-6B). The planted aspen seedlings were associated with higher cover of one introduced forb (*Salsola kali* L.), while the no planted aspen plots were associated with two introduced forbs (*Melilotus* sp. and *Sonchus* sp.) (Figure 2-6B).

The model 2 PERMANOVA of community composition (cover by species) showed no significant effect of the fertilizer or the biochar treatments in either 2013 or 2014 (all $p>0.285$, Table 2-5B). The associated Indicator Species Analysis suggested that there were no significant indicator species for the treatments in either 2013 or 2014. The NMDS of the 2013 data indicates that variation along the first axis was driven by two native grasses (*Elymus trachycaulus* (Link) Gould ex Shinners subsp. *trachycaulus,* and *Phalaris arundinacea* L.), two native forbs (*E. arvense* and *Vicia americana* Muhl. ex Willd.), three introduced forbs (*C. tectorum, Galeopsis tetrahit* L., and *S. palustris*) and two noxious weeds (*T. vulgare* and *T. perforatum*), which all loaded towards to high end of the axis; the ellipse indicates that most were plots from Block 3. The primary species driving the second axis is *Polygonum aviculare* L. (Figure 2-7A). In 2014, the plots were mostly spread evenly across the first and second axis. Variation along the low end of the first axis was driven by one species from 2013 (native forb *V. americana)* along with one introduced forb (*Melilotus sp.*). *P. aviculare* is no longer driving the secondary axis, and there is no longer any one specific species driving this axis according to the correlated species scores (Figure 2-7B).

Block was found to be significant in the PERMANOVA of overall community composition in 2013 but not 2014. There were no significant interactions between block and the different treatments (Table 2-5B). Further, the analysis was repeated with Block 3 removed; however, the results didn't change; therefore, there was no further examination of the effect of block.

2.4 Discussion

The goal of this study was to examine the impacts of planted aspen trees, biochar, and fertilizer amendments on the early development of herbaceous vegetation on a coal mine reclamation site that had poor soil quality. The results showed that planted aspen seedlings and a fertilizer amendment increased both cover and richness of colonizing vegetation. Varying the amount (50 kg/ha and 100 kg N/ha) of controlled-release fertilizer did not result in additional effects. The application of biochar at a rate of 1800 kg/ha did not influence the cover of colonizing vegetation but did result in decreased richness.

Environmental and edaphic conditions on the reclamation site were challenging during the 2013 and 2014 growing seasons as the growing season total precipitation was very low (271 mm in 2013 and 376 mm in 2014; see Figure B 1). In combination with the soil texture, this resulted in an average of only 3% volumetric water content available within the first 10 cm of soil available for plant use (Figure B 2); this is below the permanent wilting point in a silt loam or loam textured soil (Figure B 3). Furthermore, at a depth of 30 cm the average volumetric water content was 22%, falling within the area of 50% available water content in a silt loam or loam textured soil (Figure B 3); only vegetation with deep roots or a taproot system would have been able to access this. The low precipitation most likely led to the moderate to abnormally dry conditions experienced on the reclamation site (Agriculture and Agri-Food Canada 2014); these dry conditions most likely played a role in the observed decrease of richness by six species from the first to the second growing season.

2.4.1 Influence of planted aspen seedlings

As hypothesized, the presence of planted aspen seedlings had a positive effect on the colonizing vegetation, with cover almost doubling and the community composition being notably different than in areas with no seedlings. Forest canopy structure including tree species composition and growth rates have an important influence on establishing the understory plant community (Macdonald et al. 2012, 2015a, Pinno and Errington 2015). Creating an intact forest canopy as quickly as possible on reclamation sites can help reduce the establishment of shade intolerant and ruderal species that can inhibit the establishment of more desirable forest understory species (Macdonald et al. 2015a). However, within the timeframe of this study the forest canopy had not developed enough to favour shade tolerant species. Since planting in 2012, the aspen seedlings grew in height by an average of 44 cm by the summer of 2014 to an average total height of 71 cm (Bockstette unpublished). In mature forests, the canopy plays an integral role in regulating resources such as light, soil water, and nutrients for the understory, as well as providing other ecosystem functions such as increasing organic matter through leaf litter deposition, decreasing soil temperature by shading, and providing shelter for vegetation through reducing wind speeds (Macdonald et al. 2012, Das Gupta et al. 2015). While the tree seedlings present on site were not very large and had not reached canopy closure, their presence may still have directly influenced the above- and below-ground growing conditions, although to a lesser extent than a mature forest. Species richness decreased from the first to the second growing season in plots where aspen seedlings were not planted and this could be due to a lack of shelter and poor soil conditions associated with the treeless landscape (Bradshaw 1997, Hart and Chen 2006, Gignac and Dale 2007, Frouz et al. 2009, Mudrák et al. 2010). When planted aspen seedlings were present, however, richness increased and vegetation cover doubled which could be attributed to the sheltering effect provided by the young seedlings as mentioned above. The planted aspen seedlings may have captured wind dispersed seeds which could increase species richness surrounding the seedlings; additionally, there may be increased ectomycorrhizal fungal establishment in the soil surrounding the roots of the aspen seedlings, which is important for nutrient acquisition, water uptake, and nutrient cycling (Hankin et al. 2015). Several vegetation species, specifically nitrogen fixing plants such as *Melilotus* species, may have taken advantage of this increased fungal presence and established in these areas.

The colonizing vegetation found on the site was comprised mostly of early successional introduced species typical of early reclamation sites (Frouz et al. 2008, Rowland et al. 2009, Macdonald et al. 2012). During both growing seasons planted aspen seedlings played a role in driving the understory community composition. In the first growing season the NMDS showed that mostly introduced ruderal species, including one noxious weed and additionally, one native species were driving the variation in the community. In contrast, by the second growing season there were distinctions in community composition between the planted and unplanted areas. *Salsola kali,* another early successional shade intolerant species (Moss 1994, Beckie and Francis 2009) remained driving the variation in the community under planted aspen seedlings; indicator species included four other early successional ruderal species (see Table 2-6) found in disturbed areas (Moss 1994, Johnson et al. 1995, Bubar et al. 2000) under planted aspen seedlings.

The chances for shade tolerant species to migrate onto this reclamation are poor since surrounding areas were reclaimed for agricultural purposes and there are thus no intact forests nearby. Reclamation practices typically include the use of a cover soil to provide nutrients, re-introduce microorganisms to the reclaimed landscape, as well as a supply a native propagule bank to facilitate vegetation colonization and community development (Mackenzie and Naeth 2010, Fair 2011, Brown and Naeth 2014). Since topsoil was placed near our study site, there were limited forest species' propagules that were present in the surrounding area which may have had the ability to migrate onto the site within the initial growing seasons. As a result, species persisting throughout the two growing seasons were predominantly introduced forbs typical of early successional sites, which can provide soil stability and initiate site recovery. The presence of the noxious weed *T. vulgare* in areas without planted aspen seedlings demonstrates that not all colonizing vegetation found on this reclamation is conducive to long-term reclamation success. Invasive species tend to outcompete native and introduced vegetation, and may degrade the landscape and change ecosystem structure (Prach and Hobbs 2008). With increasing site productivity, competition and dominance by invasive and noxious species may need to be controlled and managed by land owners; reclamation sites with the presence of weeds do not meet reclamation criteria requirements and therefore control of the noxious weed is required, generally through inhibition of its growth or spread (Province of Alberta 2011). The presence of noxious weeds in developing communities on reclamation sites hinders the process of forming a forest understory, in addition to posing challenges for receiving a reclamation certificate. It is promising that the presence of planted aspen seedlings on this site prevented the establishment of noxious weeds, further illustrating their benefit for colonizing vegetation.

In naturally disturbed or managed forests, there is a shift in community composition from shade intolerant species to shade tolerant species once canopy closure has occurred (Hart and Chen 2006), which should correspond to a decrease in introduced and noxious species. Similar shifts are expected to occur on this reclamation site in the future as the planted aspen seedlings continue to grow, providing greater canopy closure and increased shelter for the developing understory. The future trajectory of this reclamation site is satisfactory, as there was a distinction between understory communities in areas with *versus* without planted aspen in the second growing season without noxious weeds driving the variation in the understory community. Nevertheless, the potential for colonization of desirable forest understory species on this site is still unclear due to the limited nearby propagule availability.

2.4.2 Influence of the fertilizer amendment

Fertilization resulted in higher vegetation cover and richness compared to the unfertilized control in the first growing season. However, contrary to our second hypothesis, the higher fertilizer rate (100 kg N/ha) did not increase vegetation cover compared to the lower fertilizer treatment. In order for vegetation communities to establish and persist on reclaimed soil, fertilizer comprised of nitrogen, phosphorous and potassium is often necessary in order to make soil more suitable and to supplement the natural organic matter and nutrients that can be deficient due to stockpiling of soils (Norman et al. 2006, Sheoran et al. 2010, Macdonald et al. 2015a). However, as hypothesized the fertilizer effects appeared to be short-lived with no fertilizer effects detectable in the following growing season.

The use of controlled-release fertilizer has been gaining attention in the last 20 year as it allows nutrients to be released over a longer period of time in order to match demand from vegetation, thus improving use efficiency and minimizing negative environmental effects (Arnon 1992, Shaviv and Mikkelsen 1993, Sloan et al. 2016). While nutrient availability in the soil can be addressed using controlled-release fertilizer application, in this study fertilizer effects were only seen during the first growing season. On sites with poor soil conditions, it was anticipated that controlled-release fertilizer with a nutrient release rate of 8-9 months would allow for increased nutrients in the soil and vegetation by the second growing season, as opposed to multiple applications of immediately available fertilizer; this would reduce the potential for leaching and increase initial soil and microbial processes. Although fertilizer effects were only found for one year, presumably the nutrients were available for a greater proportion of that growing season than what would be from a traditional immediately available fertilizer.

Varying the amount (50 kg/ha and 100 kg N/ha) of controlled-release fertilizer did not impact plant community establishment; there was no additional effect from the high fertilizer treatment on the cover or richness of the vegetation as compared to the low fertilizer treatment. Nitrogen availability is often limiting for plant growth in the boreal forest (Näsholm et al. 1998, Turkington et al. 1998, Hangs et al. 2002), and the type and rate of application of fertilizers can impact the growth and nutrient uptake of vegetation (Sloan and Jacobs 2013, Sloan et al. 2016). Regardless of how much nitrogen is available in the soil, once a species fulfills its nutrient requirements, additional nutrients will not be taken up (i.e. the demand for N in the plant has been saturated) (Manninen et al. 2009). Growth strategies, nutrient requirements and utilization strategies vary between species and this can determine how much nitrogen is taken up by the plant (Manninen et al. 2009, Aubrey et al. 2011). Some species are less dependent on inorganic N in the soil, while in contrast other species may be able to efficiently take up and utilize inorganic N (Manninen et al. 2009). Previous research at this site found that the addition of the low fertilizer level increased both total nitrogen and ammonia content within the soil in 2012 (Liu 2015). However, by 2013 there were no longer any significant differences in nitrogen content in the soil, albeit high fertilizer levels did increase tree growth (Bockstette unpublished). Despite the lack of differences in soil nitrogen in 2013 the vegetation was most likely benefiting from the addition of fertilizer during this first growing season as seen by increased cover and richness in the low and high fertilizer plots. By the second growing season there were no longer any significant differences in the soil (Bockstette, unpublished) and vegetation between fertilizer treatments; this could be attributed to the fertilizer having been immobilized within the soil by microorganisms (Hangs et al. 2003), the vegetation, or the biochar amendment (Xie et al. 2016), or lost via physical processes such as leaching or surface runoff (Shaviv and Mikkelsen 1993).

Fertilization has to been shown to support seedling growth, benefit colonizing vegetation, and aid in ecosystem function recovery on restored sites (Rowland et al. 2009, Sloan and Jacobs 2013). As a result of limited site development time at the Whitewood coal mine research site (first growing season) it was expected that vegetation would still be representative of a recently disturbed site with an understory comprised of successional species. This site experienced a severe disturbance and was still very open; therefore, the presence of nitrophilous and early successional species dominating in the plots was not unexpected as these species have the ability to grow quickly (Grime 1977, Grainger and Turkington 2013). While the NMDS showed the community development was not driven by the two fertilizer levels or biochar amendments, the species associated with the first growing season were predominantly introduced forbs (*P. aviculare*, *S. palustris*, *G. tetrahit*, and *C. tectorum*) including two noxious weeds (*T. vulgare* and *Tripleurospermum perforatum* (Mérat) M. Lainz), as well as four native species (*E. trachycaulus* subsp. *trachycaulus, V americana*, *E. arvense*, and *P. arundinacea*). Only one native species and one introduced forb (*V. americana* and *Melilotus sp.*, respectively) drove the variation in the community in the second growing season. While there was a change in the vegetation community, there still was not a shift away from shade intolerant colonizing vegetation towards more shade tolerant species which are more suited to possible future boreal forest understory conditions. Additional application of low fertilizer treatments may be necessary to further increase soil nutrient availability, increase presence of vegetation to reduce soil erosion, and eventually increase site productivity.

2.4.3 Influence of the biochar amendment

In contrast with our third and final hypothesis, biochar application did not affect cover of colonizing vegetation, and richness was found to decrease over time in areas treated with biochar. In addition, the combined effect of biochar addition and fertilizer application were expected to provide the greatest increase to total cover and decrease species richness in this study, however there were no effects from combining the biochar application with either low or high fertilizer treatments, contradicting our hypothesis.

The absence of change in colonizing vegetation cover in this study is consistent with a study done by Gundale et al. (2015) who found that the addition of biochar to a clear felled forest had no effect on vegetation properties. These results are in contrast, however, with the meta-analysis done by Biederman and Harpole (2013) who found biochar promoted plant growth and productivity in numerous studies. Biochar can increase microbial activity, enhance vegetation growth, and can alter soil by improving various physical, chemical and biological properties such as soil density and changes in water infiltration and drainage rates (Glaser et al. 2002, Lehmann et al. 2006, Fellet et al. 2011, Chaganti et al. 2015, Chia et al. 2015, Jay et al. 2015, Lehmann and Joseph 2015, Masiello et al. 2015). Depending on feedstock type and temperature during pyrolysis (Gundale and DeLuca 2006, Deluca et al. 2015) biochar may contribute nutrients into the soil. Nonetheless, once in the soil, biochar is less likely to act as the main source of nutrients and instead contributes to impacting nitrogen transformations (Deluca et al. 2015), as particles of biochar can react with the soil material and organic matter and attract nitrate and ammonium ions, which in turn could reduce their loss through leaching (Cowie et al. 2015). Soil improvements from biochar, specifically increasing the cation exchange capacity, can prevent nutrient leaching and thus lead to an increased yield, as well as a decrease in the amount of fertilizer necessary (Blackwell et al. 2012, Deluca et al. 2015).

It was expected that colonizing vegetation would benefit from the combined application of biochar and fertilizer amendments, as Liu (2015) reported at this study site in 2012 soil nitrate levels were higher in areas with both biochar and either high or low fertilizer treatments compared to areas with no fertilizer or biochar only. These increases in available nitrate theoretically should have created better soil conditions where early successional vegetation could colonize more readily, thus increasing cover and decreasing richness from competition. The planted aspen did not have increased foliar nitrogen concentration regardless of fertilizer, biochar, or combination of the two treatments (Bockstette unpublished); from this I suspect that the nitrogen was immobilized by the herbaceous vegetation, the biochar, or the microorganisms present in the soil and subsequently no longer available, or was removed completely from the environment through leaching or surface runoff. Biochar may have also decreased nutrient availability, as the addition of biochar may cause nutrients to precipitate, converting them into unavailable forms through a change in soil pH or interactions with biochar-derived nutrients (Whitman et al. 2015). Additionally, biochar may increase the immobilization of mineral nutrients in the soil therefore increasing microbial biomass and reducing the available nutrients (Whitman et al. 2015).

When biochar was applied on site, species richness (particularly of introduced species) decreased over time regardless of the presence of planted aspen seedlings. Gundale et al. (2015) also found a small decrease in plant species richness from the application of biochar. The decrease of richness over time in our study also occurred in plots with the fertilizer amendment, bringing about the notion that richness was decreasing due to soil or environmental conditions. When the effect of biochar was only examined in planted aspen seedling plots (Model 2), no differences in cover or richness of colonizing vegetation were found, however the trend of decreasing richness with biochar application was still present albeit not significant. I speculate that the difference between the effects of biochar between Model 1 and Model 2 could be due to a difference in sample size, as Model 1 had a greater number of plots sampled than Model 2 (432 plots sampled in Model 1 vs. 360 plots sampled in Model 2).

2.5 Conclusion

In summary, the application of biochar as an organic amendment in this study caused richness of volunteer herbaceous vegetation to decrease and did not impact the total cover of colonizing vegetation in areas either with or without planted aspen seedlings. Although biochar had little influence on the vegetation on this reclamation site during the time period of this study, a future effect may be seen as biochar can immobilize nutrients in the soil, enhance vegetation growth and continue to improve soil properties. Further research on using biochar as an organic amendment alone and in combination with other amendments over the short and long-term is needed to develop greater understanding of how biochar interacts with soil and vegetation.

Planting aspen seedlings to begin initial forest recovery and using a slow release fertilizer did increase total cover and richness of colonizing vegetation on these poor substrates. The vegetation community was becoming more distinct in areas with planted aspen seedlings in the first two growing seasons, and the variation in the community was generally driven by early successional shade intolerant species. This change in vegetation community means that planting trees on reclamation sites can lead to changes in the colonizing plant community creating greater opportunities to develop a more desirable, shade tolerant understory by providing the necessary shelter from the harsh site and soil conditions.

The unassisted establishment of both desirable and undesirable species on the site demonstrates that reclamation was effective; however, progress towards the target boreal forest understory community will be slow. Additionally, further research on various application rates of controlled-release fertilizer as amendments is needed to develop greater understanding of how colonizing and established vegetation utilizes available nitrogen and other nutrients in the soil. Using a wider range of application rates, such as including 25 kg N/ha and 75 kg N/ha together in a study with 50 kg N/ha and 100 kg N/ha, may lead to optimizing the application rates based on vegetation uptake and utilization on harsh reclamation sites. It is unknown if a 25 kg N/ha rate of controlled release fertilizer would have benefited early vegetation development, as decreased fertilizer may have created an early successional vegetation community.

2.6 Tables

Table 2-1: Post hoc comparisons of means and how alpha was adjusted for these, following significant effects in the ANOVAs examining vegetation cover and species richness, and the PERMANOVA examining treatment effects on community composition (cover by species) in the two sampling years for the two different statistical models (Model 1 and Model 2).

	Significant Effect	Alpha value	Pairwise Comparisons
Model 1			
Year	Between the two Years	α = 0.05	2013 vs. 2014
Tree	Between Tree Planting treatments	α = 0.05	Planted Aspen vs. No Planted Aspen
Year x Biochar	Between Years within Biochar	α = 0.05/2 α = 0.025	Biochar: 2013 vs. 2014 No Biochar: 2013 vs. 2014
	Between Biochar within Year	α = 0.05/2 α = 0.025	2013: Biochar vs. No Biochar 2014: Biochar vs. No Biochar
Year x Fertilizer	Between Years within Fertilizer	α = 0.05/2 α = 0.025	Low Fertilizer: 2013 vs. 2014 No Fertilizer: 2013 vs. 2014
	Between Fertilizer treatments within Year	α = 0.05/2 α = 0.025	2013: Low vs. No Fertilizer 2014: Low vs. No Fertilizer
Year x Tree	Between Years within Tree	α = 0.05/2 α = 0.025	Planted Aspen: 2013 vs. 2014 No Planted Aspen: 2013 vs. 2014
	Between Tree Planting treatments within Year	α = 0.05/2 α = 0.025	2013: Planted vs. No Planted Aspen 2014: Planted vs. No Planted Aspen
Model 2			
Year x Fertilizer	Between Years within Fertilizer	α = 0.05/2 α = 0.025	High Fertilizer: 2013 vs. 2014 Low Fertilizer: 2013 vs. 2014 No Fertilizer: 2013 vs. 2014
	Between Fertilizer treatments within Year	α = 0.05/3 α = 0.017	2013: High vs. Low Fertilizer, High vs. No Fertilizer, and Low vs. No Fertilizer 2014: High vs. Low Fertilizer, High vs. No Fertilizer, and Low vs. No Fertilizer

Table 2-2: Results of linear mixed effects model for vegetation cover and species richness at the Whitewood coal mine reclamation site. Given are results for: all species, forbs, and introduced species (see Table A 2). P values are given and bolded when significant (α=0.05). Model 1 (A) examined the influence of year (Y), planted aspen trees (T), biochar (B), fertilizer (F, two levels), and their interactions. Model 2 (B) examined the influence of year, biochar, fertilizer (three levels), and their interactions. In all analyses cover and richness data were ln+1 transformed.

(A) Model 1

	All Species		*Forb Species*		*Introduced Species*	
Response Variable	Cover	Richness	Cover	Richness	Cover	Richness
Year	**<0.001**	**<0.001**	**0.001**	**<0.001**	**<0.001**	**<0.001**
Biochar	0.837	0.955	0.801	0.953	0.763	0.927
Fertilizer	0.071	0.139	0.085	0.185	0.099	0.251
Tree	**0.038**	0.146	**0.042**	0.188	**0.050**	0.244
Y x B	0.482	**0.037**	0.482	0.055	0.498	**0.048**
Y x F	**0.034**	**<0.001**	**0.042**	**<0.001**	**0.035**	**<0.001**
B x F	0.981	0.510	0.851	0.518	0.868	0.507
Y x T	0.891	**0.010**	0.840	**0.010**	0.904	**0.018**
B x T	0.622	0.733	0.530	0.615	0.550	0.668
F x T	0.612	0.609	0.558	0.470	0.544	0.509
Y x B x F	0.812	0.150	0.739	0.178	0.752	0.137
Y x B x T	0.807	0.881	0.750	0.724	0.745	0.791
Y x F x T	0.375	0.462	0.407	0.513	0.391	0.427
B x F x T	0.464	0.236	0.504	0.239	0.522	0.216
Y x B x F x T	0.582	0.272	0.494	0.437	0.512	0.425

(B) Model 2

	All Species		*Forb Species*		*Introduced Species*	
Response Variable	Cover	Richness	Cover	Richness	Cover	Richness
Year	**0.007**	**0.005**	**0.004**	**<0.001**	**0.003**	**<0.001**
Biochar	0.689	0.641	0.626	0.818	0.685	0.976
Fertilizer	0.502	0.108	0.560	0.123	0.587	0.136
Y x B	0.814	0.307	0.755	0.203	0.788	0.250
Y x F	0.121	**0.001**	0.129	**0.001**	0.124	**<0.001**
B x F	0.755	0.281	0.884	0.339	0.879	0.327
Y x B x F	0.544	0.267	0.568	0.274	0.568	0.170

Table 2-3: Summary of least squared means ± standard error (6 blocks per treatment) showing significant treatment effects and interactions for cover and richness of both forb and introduced functional groups from analysis using Model 1. Different lowercase letters in rows indicate significant differences between treatments for the given variable of the forb functional group, while different uppercase letters in rows indicate significant differences between treatments for the given variable of the introduced functional group. * and † below a column indicate significant differences between years for a given treatment for forb and introduced species, respectively. Table (A) examines significant year by fertilizer treatment interaction for cover and richness, (B) examines significant year by tree treatment interaction for richness, and (C) examines significant year by biochar treatment interaction for richness of Introduced species only. Note: Model 1 examined the influence of year, planted aspen trees, biochar, fertilizer (two levels), and their interactions; data were ln+1 transformed for analysis but the means presented here are on the untransformed data.

(A) Year x Fertilizer Interaction

		Forb Species		*Introduced Species*	
Response Variable		Low Fertilizer	No Fertilizer	Low Fertilizer	No Fertilizer
Cover	2013	10.83±2.16 **a**	7.25±2.16 **b**	10.68±2.03 **A**	7.28±2.03 **B**
	2014	10.32±2.16 **x**	7.84±2.16 **x**	9.34±2.03 **X**	7.88±2.03 **X**
Between Years within Fertilizer		*	**ns**	†	**ns**
Richness	2013	3.93±0.32 **a**	3.13±0.32 **b**	3.82±0.29 **A**	3.10±0.29 **B**
	2014	2.79±0.32 **x**	2.83±0.32 **x**	2.74±0.29 **X**	2.85±0.29 **X**
Between Years within Fertilizer		*	**ns**	†	**ns**

(B) Year x Tree Interaction

		Forb Species		*Introduced Species*	
Response Variable		Trees	No Trees	Trees	No Trees
Richness	2013	3.56±0.31 **a**	3.50±0.32 **a**	3.49±0.28 **A**	3.43±0.29 **A**
	2014	3.15±0.31 **x**	2.47±0.32 **y**	3.08±0.28 **X**	2.51±0.289 **X**
Between Years within Tree Treatment		**ns**	*	**ns**	†

(C) Year x Biochar Interaction

		Introduced Species	
Response Variable		Biochar	No Biochar
Richness	2013	3.63±0.29 **A**	3.29±0.29 **A**
	2014	2.69±0.29 **X**	2.90±0.29 **X**
Between Years within Biochar		†	**ns**

Table 2-4: Summary of least squared means ± standard error (6 blocks per treatment) showing significant treatment effects and interactions for cover and richness of both forb and introduced functional groups from analysis using Model 2. Different lowercase letters in rows indicate significant differences between treatments for the given variable of the forb functional group, while different uppercase letters in rows indicate significant differences between treatments for the given variable of the introduced functional group. * and † below a column indicate significant differences between years for a given treatment for forb and introduced species, respectively. Table (A) examines significant year effect for cover, and (B) examines significant year by fertilizer treatment interaction for richness. Note: Model 2 examined the influence of year, biochar, fertilizer (three levels), and their interactions using data from areas with planted aspen; data were ln+1 transformed for analysis but the means presented here are on the untransformed data.

(A) Year

	Forb Species		*Introduced Species*	
Response Variable	2013	2014	2013	2014
Cover	11.5±2.46 **a**	12.5±2.46 **b**	11.32±2.13 **A**	11.63±2.13 **B**

(B) Year x Fertilizer

		Forb Species			*Introduced Species*		
Response Variable		High Fertilizer	Low Fertilizer	No Fertilizer	High Fertilizer	Low Fertilizer	No Fertilizer
Richness	2013	4.43±0.38 **a**	4.55±0.38 **a**	3.26±0.38 **b**	4.30±0.32 **A**	4.30±0.32 **A**	3.19±0.32 **B**
	2014	3.16±0.38 **x**	3.26±0.38 **x**	3.25±0.38 **x**	3.11±0.32 **X**	3.10±0.32 **X**	3.23±0.32 **X**
Between Years within Fertilizer		*	*	**ns**	†	†	**ns**

Table 2-5: Results of Permutational multivariate analysis of variance (PERMANOVA) examining treatment effects on community composition (cover by species) in each of the two sampling years separately. P values are given and bolded when significant (α=0.05). Model 1 (A) examined the influence of planted aspen trees, biochar, fertilizer (two levels), and their interactions. Model 2 (B) examined the influence of the biochar, fertilizer (three levels), and their interactions using data only from areas planted with aspen.

(A) Model 1

	Cover by Species	
Response Variable	2013	2014
Biochar	0.062	0.504
Fertilizer	0.348	0.632
Tree	**0.004**	**<0.001**
Block	**<0.001**	0.065
Biochar x Fertilizer	0.554	0.697
Biochar x Tree	0.875	0.383
Fertilizer x Tree	0.681	0.131
Biochar x Fertilizer x Tree	0.928	0.807

(B) Model 2

	Cover by Species	
Response Variable	2013	2014
Biochar	0.519	0.546
Fertilizer	0.285	0.372
Block	**0.002**	0.118
Biochar x Fertilizer	0.512	0.632

Table 2-6: Results of indicator species analysis on vegetation composition (cover by species) from analysis using Model 1 for each year. Listed are species for which the indicator value (IndVal) was significant (p-value < 0.05). The analysis focused on the influence of the tree planting effect, which was the only treatment effect on community composition that was found to be significant by PERMANOVA (see Table 2-5 above). Note: Model 1 examined the influence of planted aspen trees, biochar, fertilizer (two levels), and their interactions.

Year	Treatments	Species	IndVal	p-value
2014	Trees	*Melilotus* sp.	0.950	0.001
		Crepis tectorum	0.645	0.005
		Taraxacum officinale	0.577	0.005
		Erysimum cheiranthoides	0.576	0.018
		Hordeum jubatum	0.556	0.011
	No Trees	*Trifolium repens*	0.540	0.008

2.7 Figures

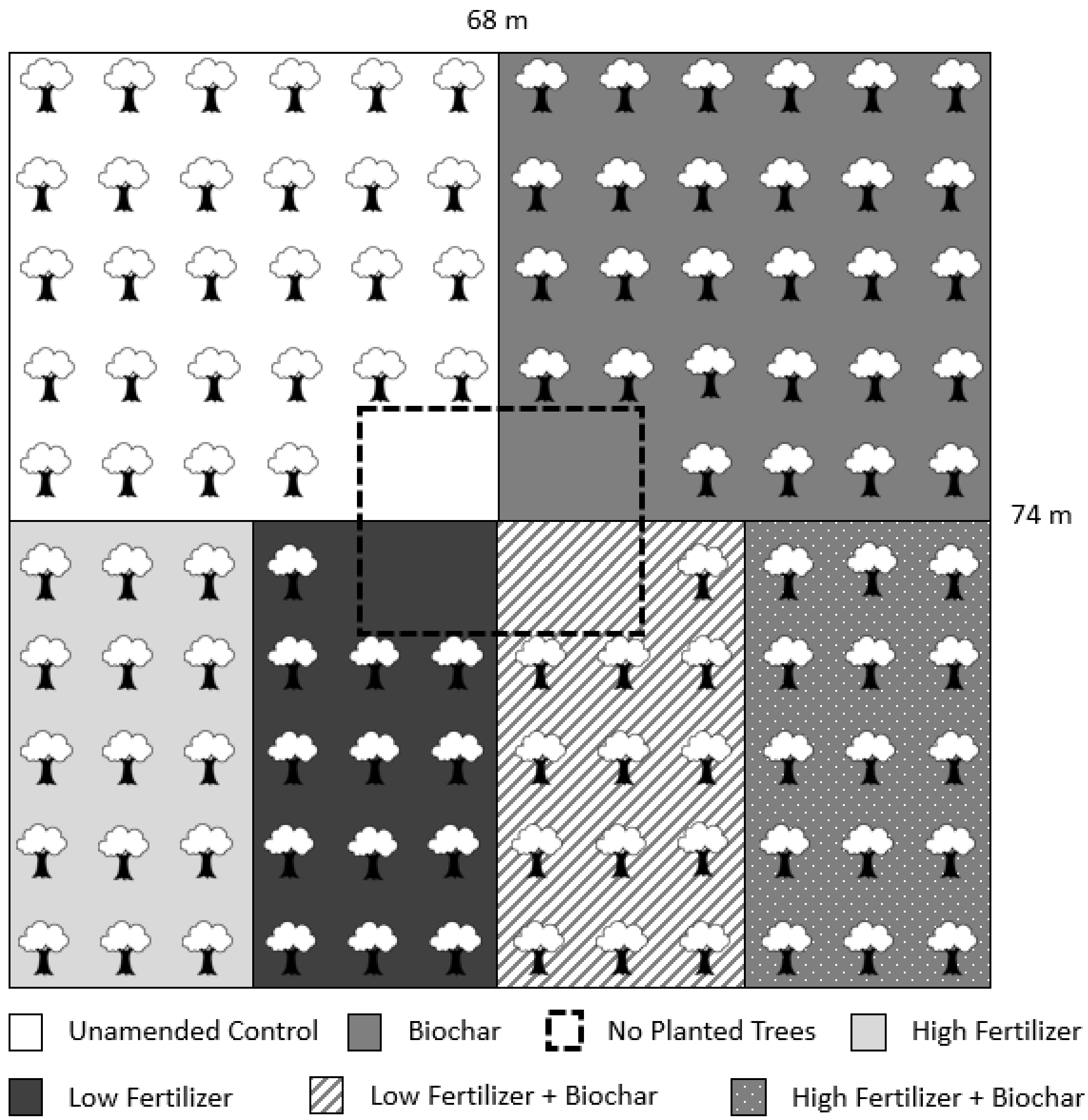


Figure 2-1: Field site layout for one of the six blocks of the reclamation experiment at the Whitewood coal mine, Alberta, Canada showing the soil amendments (fertilizer and biochar), aspen seedlings (planted evenly throughout the block), and the areas without planted trees.

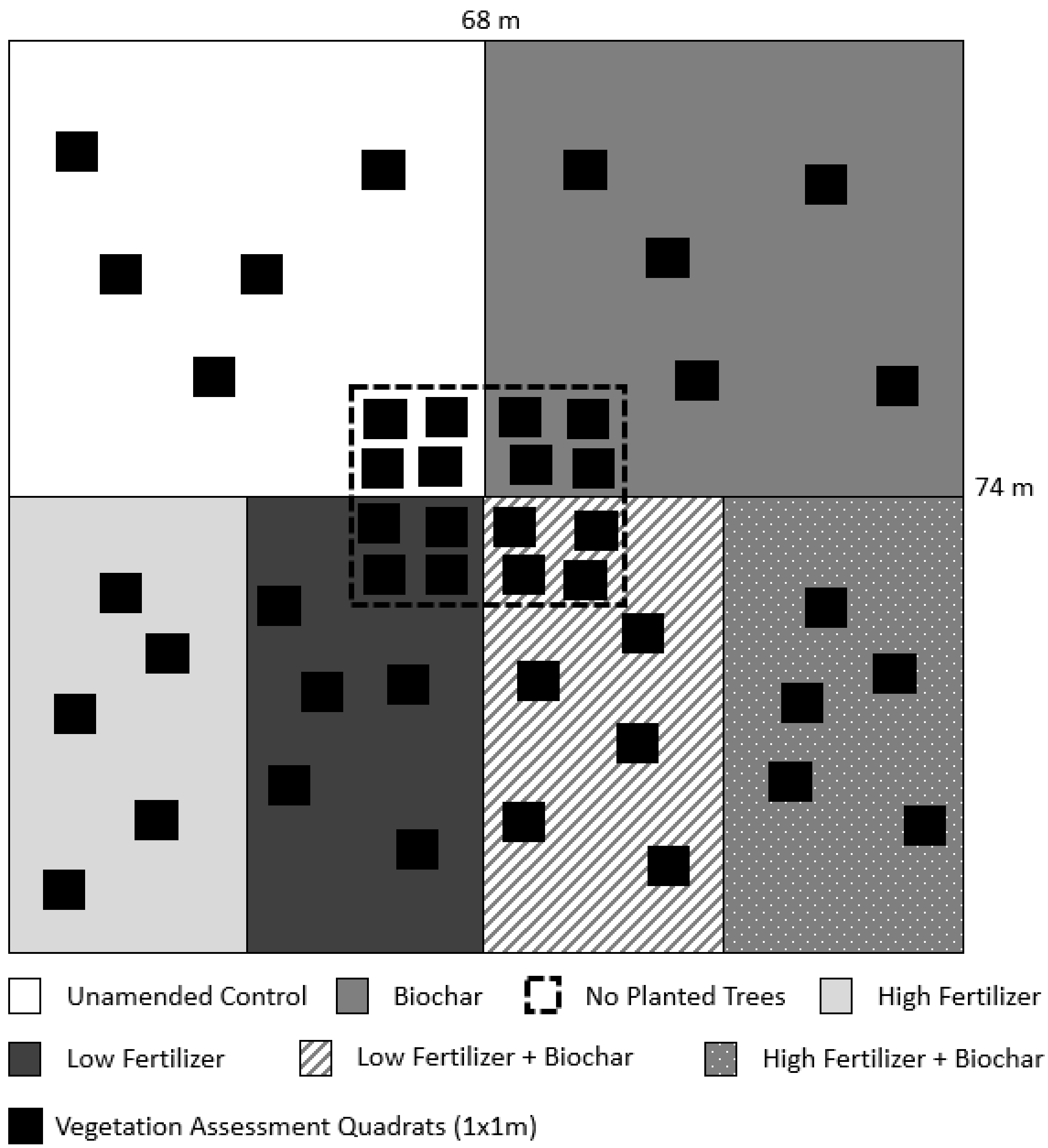


Figure 2-2: Field site layout for one of the six blocks of the reclamation experiment at the Whitewood coal mine, Alberta, Canada showing the soil amendments (fertilizer and biochar), and placement of quadrats for sampling understory vegetation.

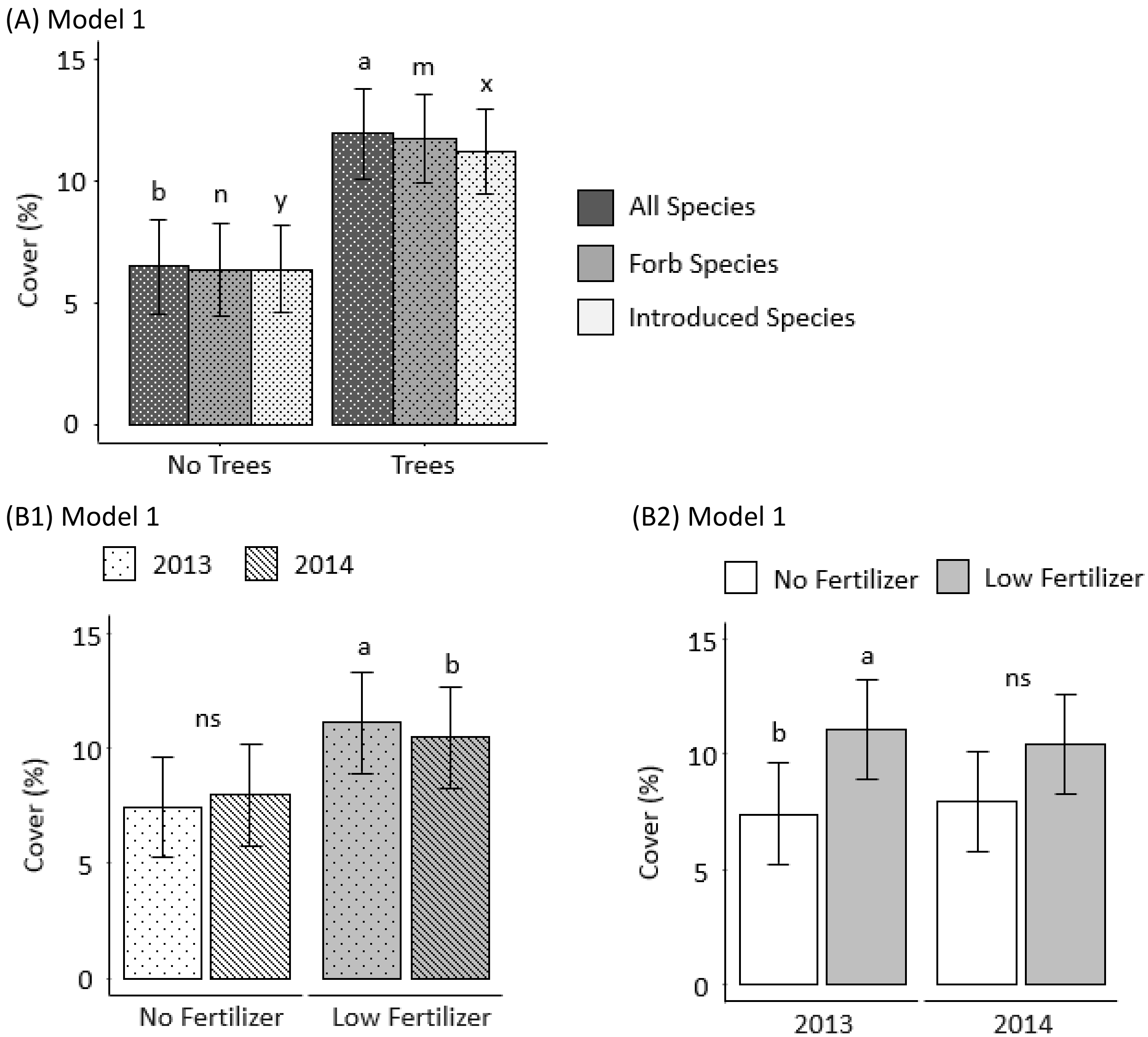


Figure 2-3: Least square means (and standard errors) for cover illustrating the significant effects from the analysis using Model 1 (see Table 2-3A). (A) Different letters indicate differences between tree and no tree treatments for each of: all plant species, forb, and introduced functional groups. (B) Cover for all plant species highlighting the significant year by fertilizer interaction; means with different letters indicate a significant difference between treatments within the given level of the other treatment; (B1) differences between years within each fertilizer level and (B2) differences between fertilizer treatments within each year. Note: ‘ns’ represents not significant.

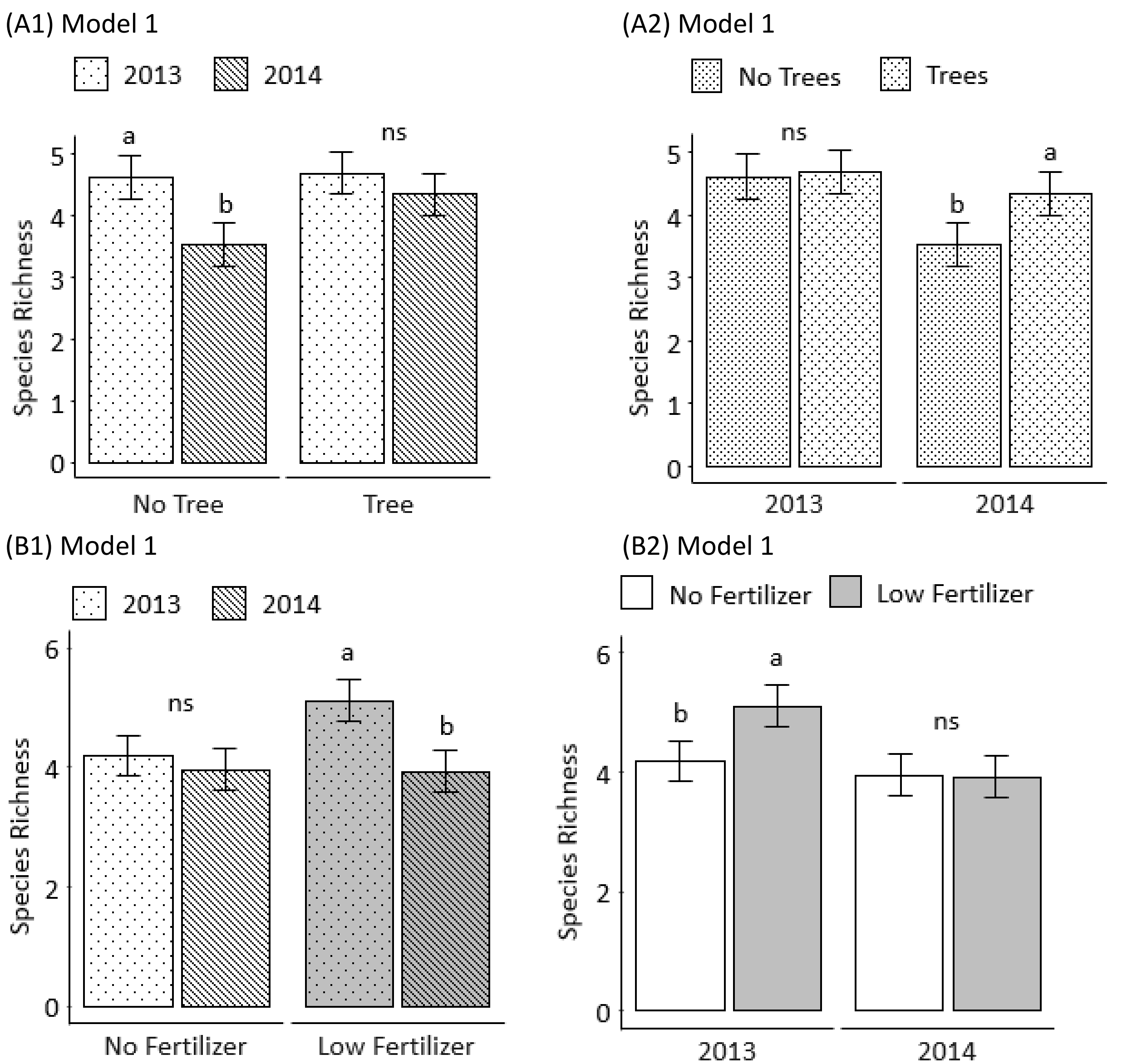


Figure 2-4: Least square means (and standard errors) for richness illustrating the significant effects from the analysis using Model 1 (see Table 2-3A). Means with different letters indicate a significant difference between treatments within the given level of the other treatment. (A) Richness for all plant species highlighting the significant year by tree interaction comparing (A1) differences between each tree level and (A2) differences between each year. (B) Richness for all plant species highlighting the significant year by fertilizer interaction comparing (B1) differences between each fertilizer level and (B2) differences between each year. Note: ‘ns’ represents not significant.

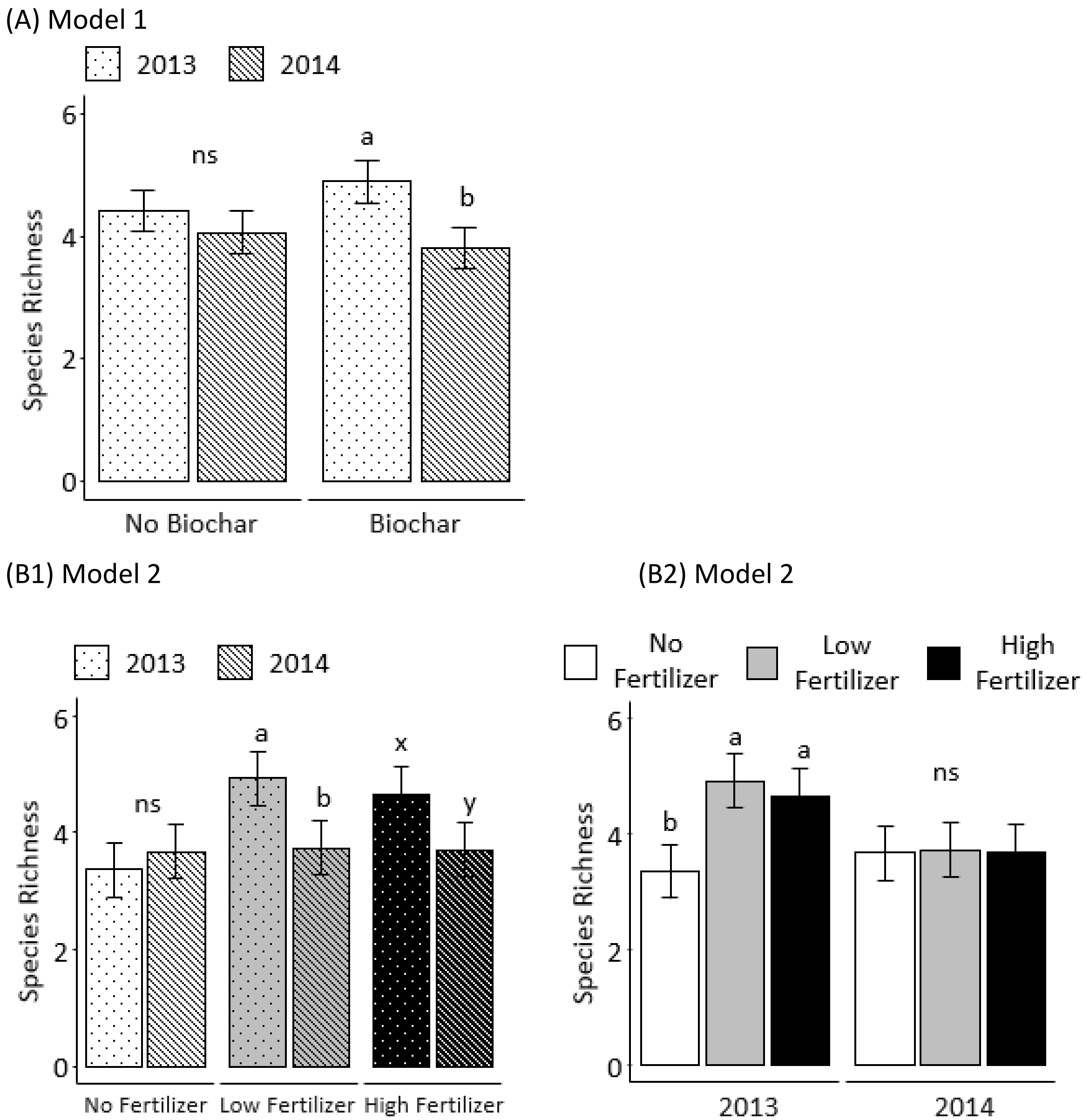


Figure 2-5: Least square means (and standard errors) for richness illustrating the significant effects from the analysis using Model 1 (see Table 2-3A). Means with different letters indicate a significant difference between treatments within the given level of the other treatment. (A) Richness for all plant species highlighting the significant year by biochar interaction comparing differences between biochar treatments within each year. (B) Richness illustrating the significant year by fertilizer interaction from the analysis using Model 2 (see Table 2-3B) for all plant species comparing (B1) differences among each fertilizer level and (B2) differences between each year. Note: ‘ns’ represents not significant.

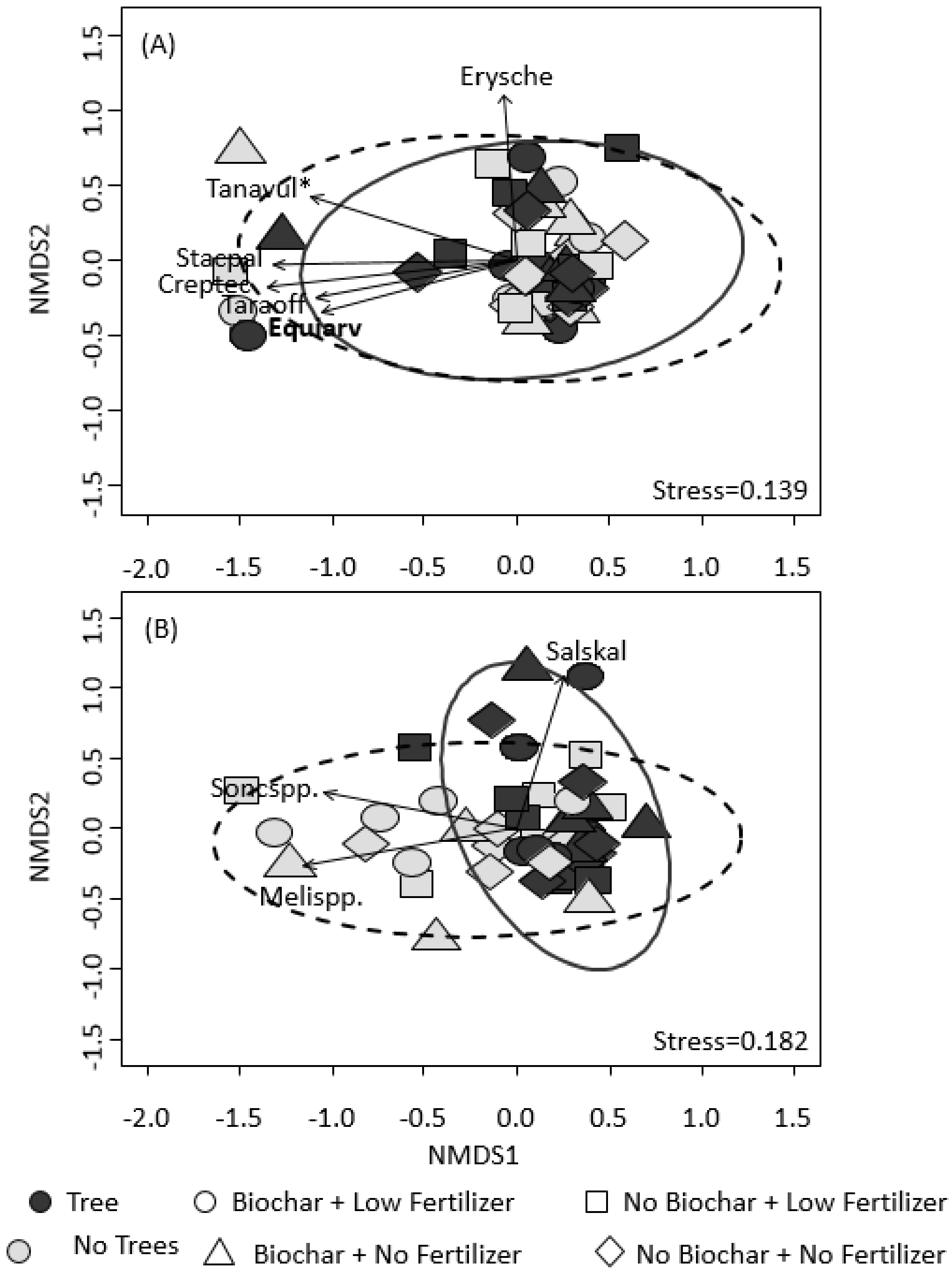


Figure 2-6: Results of NMDS ordination of vegetation composition at the reclaimed coal mine in 2013 (A) and 2014 (B). This analysis was based on data from the plots used for analyses of variance by Model 1 which tested for the effects of tree planting, fertilizer (two levels) and biochar. Points are the plots coded by tree planting treatment (colour), and fertilizer and biochar treatments (shape). Ellipses delineate the 95 percent confidence envelope (solid line = planted aspen, dashed line = no planted aspen). Vectors are shown for species that were strongly correlated with one of the ordination axes ($r^2 \geq 0.5$). Bolded indicates native species, * indicates noxious weed; species codes are shown in Table A 2.

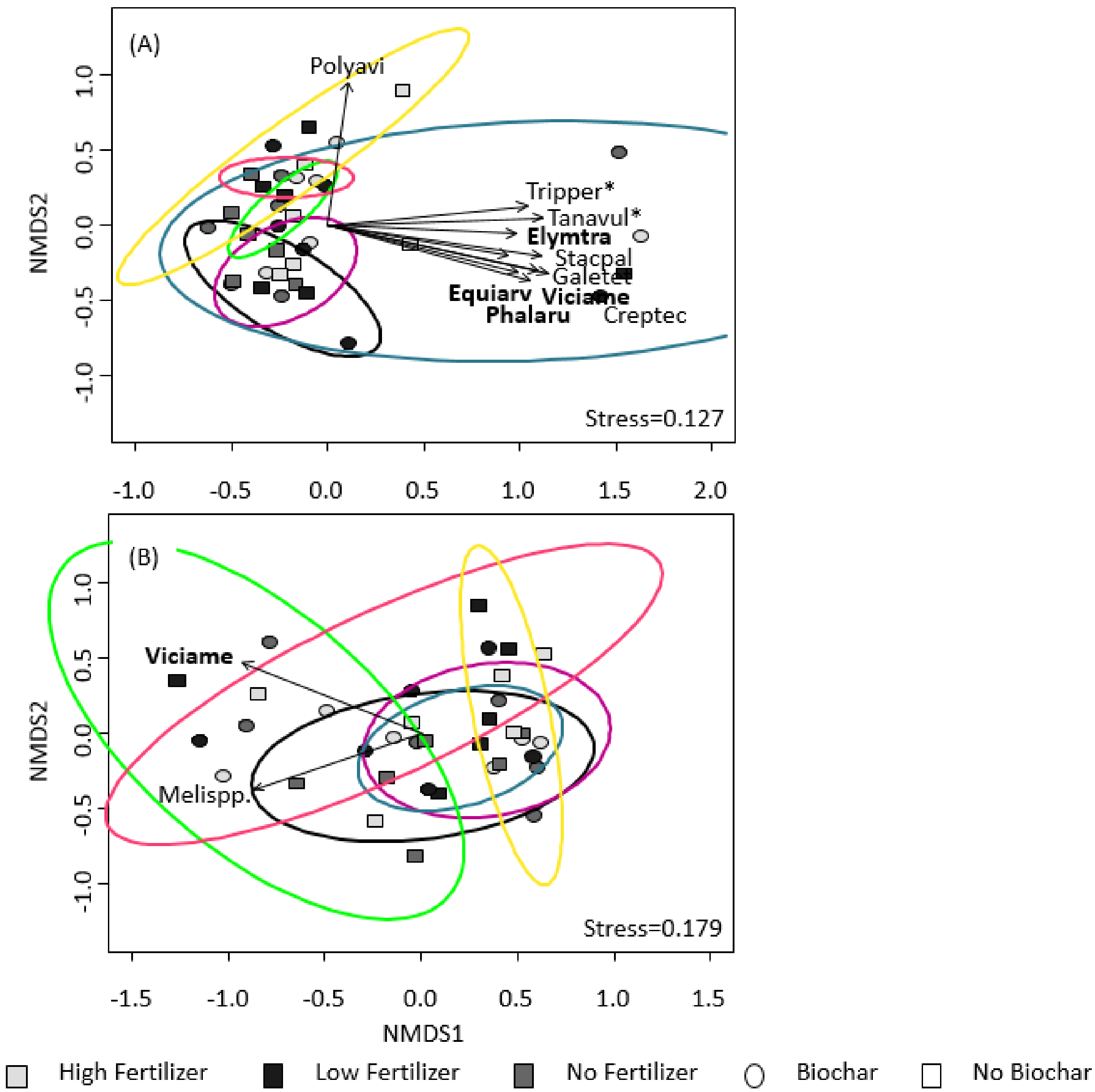


Figure 2-7: Results of NMDS ordination of vegetation composition at the reclaimed coal mine in 2013 (A) and 2014 (B). This analysis was based on data from the plots used for analyses of variance by Model 2 which tested for the effects of fertilizer (three levels) and biochar in plots with planted aspen seedlings. Points are the plots coded by fertilizer treatment (colour), and fertilizer and biochar treatments (shape). Ellipses delineate the 95 percent confidence envelope (Block 1: Black, Block 2: Purple, Block 3: Blue, Block 4: Lime Green, Block 5: Pink, and Block 6: Yellow). Vectors are shown for species that were strongly correlated with one of the ordination axes ($r^2 \geq 0.5$). Bolded indicates native species, * indicates noxious weed; species codes are shown in Table A 2.

Chapter 3: Effect of shelter materials on seedling germination and establishment

3.1 Introduction

Restoring plant communities on severely disturbed landscapes such as surface mines can be a challenging task (Macdonald et al. 2015a). Most reclamation practices involve placement of salvaged topsoil materials, which might include a natural seedbank, as well as broadcast seeding and planting to reestablish vegetation cover. In forest reclamation the planting of trees is often performed; however, for the re-establishment of an herbaceous community natural regeneration is often the only option available. Many seeds and seedlings of herbaceous species in forest reclamation are not commercially available, therefore it is not possible to sow or plant these species on reclamation sites. Research emerging from Europe emphasizes that allowing natural succession to occur on reclamation sites can sometimes result in higher species richness than occurs on highly managed reclaimed sites (see Prach et al. 2001, Pensa et al. 2004, Frouz 2013, Prach 2013). Reclaimed sites often have very harsh conditions driven by climate and surface soil conditions, making the establishment of plants via seed difficult. Issues such as higher surface temperatures and wind speeds, soil erosion, lack of nutrients, landscape homogeneity, and soil compaction (Sheoran et al. 2010, Shrestha and Lal 2011, Frouz 2013, Macdonald et al. 2015a) create many challenges. These harsh conditions may decrease soil water levels and in turn create challenging abiotic environments which lead to reduced vegetation growth and establishment (Whisenant 2002).

Management of soil conditions and increased landscape heterogeneity through creation of microsites are critical for early successional plant species establishment (Bradshaw 1997, Frouz et al. 2008, Mudrák et al. 2010, Mackenzie and Naeth 2010, Gilland and McCarthy 2014, Schott et al. 2014, Macdonald et al. 2015a). One way of creating and increasing microsite variation across the reclamation surface is the placement and use of shelter. Shelter can be created through hedgerows or windrows, nurse plants such as shrubs, young tree seedlings or taller vascular plants, large rocks or woody debris. The degree of shelter supplied will vary with the height, orientation, and location of the shelter object in the landscape (Nuberg 1998). Potential benefits of shelter include a direct physical barrier from wind, increased shade and protection from sun exposure, assistance in the capture of windblown seeds, and indirect microsite effects (Carlsson and Callaghan 1991, Bird et al. 1992, Cleugh 1998, Whisenant 2002, Buchman et al. 2007). Shade provided by shelter may produce more favourable soil temperature and moisture conditions (Turner et al. 1966) and this is directly related to the orientation of the shelter material. In the northern hemisphere, when shelter is oriented north-south, areas on the eastern and western aspect experience only a small amount of shade throughout the day - especially during the growing season when the sun is high. When shelter is oriented east-west, areas on the northern aspect experience greater amounts of shade compared to the southern aspect, where additional solar radiation reflection off the shelter material occurs (Rosenberg et al. 1983a). Microsites occurring on northern aspects are generally cooler and have higher soil moisture whereas the southern aspects are generally warmer and have drier soils (Åström et al. 2007, González-Alday et al. 2008, Macdonald et al. 2015a, Måren et al. 2015). These effects vary with angle of incidence of the sun, as the area shaded will be larger at latitudes further away from the equator (Rosenberg et al. 1983a, Adams 2010, Jones 2014, Måren et al. 2015).

Growing conditions for emerging vegetation can also be influenced by the distance to shelter as protection from wind, change in microclimate conditions, and the amount of shade will decrease with increasing distance from the shelter object (Rosenberg et al. 1983a). The interactions between aspect of shelter and distance from shelter on microsite conditions is expected to play a large role in determining vegetation growth and establishment. Vegetation immediately adjacent to shelter on southern aspects might benefit the least as there is little shade available, higher temperature and increased solar radiation reflecting off of the shelter, and soil conditions are drier. With increasing distance from shelter on a south aspect temperature may decrease however shade and soil moisture may not change (Rosenberg et al. 1983a, Jones 2014). The favourable soil, moisture and temperature conditions created from shelter may positively influence colonizing vegetation, eventually promoting the establishment of species aligning with native boreal forest understory development. However, responses by vegetation will vary depending on species requirements.

A shelter material of particular interest for use on reclaimed sites is coarse woody debris (CWD). In forest ecosystems CWD includes logs, large fallen branches, buried wood, dead roots, as well as standing dead trees and snags (Harmon et al. 1986). In natural forest ecosystems and on reclaimed sites woody debris fulfills critical ecological roles of providing shelter for many organisms, as well as contributing to ecosystem functions such as nutrient cycling, carbon storage, retention of moisture, facilitating vegetation establishment and productivity, reducing erosion, and contributing to microsite heterogeneity (see Harmon et al. 1986, Lindenmayer et al. 1999, 2002, Pyle and Brown 1999, Debeljak 2006, Grant and Koch 2007, Manning et al. 2013, Brown and Naeth 2014, Macdonald et al. 2015, and Kwak et al. 2016).

One benefit of using CWD for reclamation is the contribution of nutrients into the soil through decomposition processes (Harmon et al. 1986, Jia-bing et al. 2005, Zhou et al. 2007, Kwak et al. 2016). Decomposition involves both abiotic (leaching and fragmentation) and biotic (respiration) processes that release carbon and organic matter into the upper soil layer (Zhou et al. 2007, Merganičová et al. 2012). Nutrient concentrations in CWD increase over time, potentially due to inputs from decomposing fungi and nitrogen fixation; this can, in turn, enhance soil fertility and productivity (Laiho and Prescott 2004, Zhou et al. 2007, Hagemann et al. 2009, Wiebe et al. 2014). Another benefit of using CWD is that it is a porous object with increased water holding capacity that may additionally redistribute moisture onto surface soils (Harmon and Sexton 1995, Lindenmayer et al. 2002, Merganičová et al. 2012, Pichler et al. 2012). Harmon and Sexton (1995) demonstrated that water flowing in and through logs, even in early stages of decay, may transfer organic matter and nutrients into the soil. Additionally, moisture content of CWD is important for determining the decomposition rate of the logs (Harmon et al. 1986, Harmon and Sexton 1995, Laiho and Prescott 2004, Pichler et al. 2012). However, benefits from the decomposition and moisture inputs of CWD may not be apparent immediately as these are complex, long term processes which can take from 5 to 200 years depending on size, species and fungal communities present (Harmon et al. 1986, Pyle and Brown 1999, Tarasov and Birdsey 2001). Alternatively, a benefit of using inert materials such as rocks or concrete bricks for reclamation is they may act as an energy sink during the day when solar energy is highest, while acting as an energy source later in the day when solar energy is lower (Brown 1969). These inert materials may also redistribute incoming precipitation although they are not as porous as CWD.

On reclaimed sites in Alberta's boreal forest, Brown and Naeth (2014) found CWD to lower soil temperature; and in Wisconsin's northern Great Lakes region Haskell et al. (2012) found that soil temperature, maximum daily temperature, and soil moisture were lower in plots with downed woody material compared to plots without woody material. Placement of CWD on reclamation sites could facilitate vegetation establishment through ameliorating the harsh environmental conditions often found on these sites (Brown and Naeth 2014). It is clear that CWD can act as a sheltering object and this should benefit establishing vegetation. However, its other properties such as moisture retention and provision of nutrients may also be important in its role in facilitating vegetation establishment. It is currently unknown if CWD is useful in reclamation simply because it is an object which provides shelter (in which case other types of shelter may be used) or if there is a greater overall effect because of its organic nature that may provide additional benefits to vegetation. This is of interest because CWD may not always be available for use; if its main effect is to provide shelter then this could be achieved using other shelter materials such as rocks, bricks, or creating micro-topography.

This research aimed to examine if there is an effect of shelter on germination, establishment, mean dry weight and height of four native species (*Bromus ciliatus* L., *Dalea purpurea* Vent., *Leymus innovatus* (Beal) Pilg., and *Vicia americana* Muhl. ex Willd) seeded at a coal mine reclamation site and how these effects vary with aspect and with distance from the shelter. Further, we sought to compare the effects of CWD as shelter to an inert material (brick) of the same dimensions.

These objectives relate to four specific hypotheses:

1) The provision of shelter from both wood and brick will increase germination, establishment (survival from the first to the second growing season), and growth (dry weight and height) of seeded species, and density of volunteer species (any species other than what was sown) compared to when no shelter is present.

2) Wood (CWD) shelter material will increase germination, establishment, and growth of seeded species, and result in higher density volunteer species than an inert (brick) shelter material.
3) Effects of the shelter on germination, establishment, growth of seeded species, and density of volunteer species will be greater on a north facing aspect of the shelter than on a south facing aspect.
4) Effect of the shelter on germination, establishment, growth of seeded species, and density of volunteer species will decrease with the distance from the shelter material.

3.2 Methods

3.2.1 Research area

Research for this study took place at the Whitewood Coal Mine (53° 33' N, 114° 29' W), located near Wabamun, Alberta, Canada approximately 70 km west of Edmonton. Open pit strip mining for coal occurred from 1962 to 2010; reclamation began in the late 1980s with the East Pit Lake (Ross and Hovdebo 1995) and has since occurred progressively across the 1900 hectares of mined land (TransAlta 2014). Areas which were not suitable for productive agriculture use were slated to be reclaimed to wildland areas, with upland forests and open grass lands as the ecosystem targets (Kuchmak 2015 personal communication). Since there were topsoil and subsoil shortages, this wildlands area was recontoured using only the uncapped subsoil of unconsolidated geological material with no placement of surface soil materials (Kuchmak 2015 personal communication). Characteristics of the surface soil to be reclaimed (unconsolidated geological material that comprise the surface substrate) are described in further detail in Table A1.

The study area was located within the Dry Mixedwood natural subregion of the boreal forest which is characterized by aspen (*Populus tremuloides* Michx.), jack pine (*Pinus banksiana* Lamb.) and white spruce (*Picea glauca* (Moench) Voss) forests in pure or mixed stands (Natural Regions Committee 2006). Natural upland soils in this subregion are typically Orthic Gray Luvisols in aspen forests, which have Ae and Bt horizons, and occur when the mean annual soil temperature is less than 8°C. Dark Gray Luvisols are found in cultivated areas, and these have eluvial features with an Ah or Ahe horizon ≥ 5 cm in thickness (Soil Classification Working Group 1998, Natural Regions Committee 2006). The Dry Mixedwood natural subregion experiences the warmest summers and highest growing degree-days of the boreal natural subregions. Climate data were gathered from a weather station approximately 30 km southwest of the mine (in the township of Tomahawk Alberta; 53°26'22.000" N, 114°43'06.000" W) (Government of Canada 2016). In 2013, the mean three-day average temperature during the growing season (May 5 – September 30) was 14.5°C, and growing season total precipitation was 271 mm. During the 2014 growing season, the mean three-day average temperature was 13.2°C, and growing season total precipitation was 376 mm (see Figure B 1). Soil water moisture was continually measured at two depths (10 cm and 30 cm) in each of the six blocks during the experimental period using 5TM soil moisture sensors with EM50 dataloggers (Decagon Devices Inc. Pullman, WA, USA). In the 2013 growing season starting in July, the mean volumetric water content of the soil was 0.029 m^3/m^3 at a depth of 10 cm, and 0.223 m^3/m^3 at a depth on 30 cm, while from May 5 – July 27 2014 mean volumetric water content of the soil was 0.071 m^3/m^3 at a depth of 10 cm, and 0.232 m^3/m^3 at a depth of 30 cm (see Figure B 2).

3.2.2 Experimental design

This experiment was set up as a blocked split-split plot design within 5 blocks of a larger experiment at the Whitewood mine (described in Chapter 2). Two experimental plots per block were set up for a total of 10 experimental plots (Figure 3-1). Within each experimental plot, 12 shelter plots were created. Three shelter treatments (unsheltered control and two shelter materials: coarse woody debris (wood) and brick) were randomly assigned to four of the 12 treatment plots. Shelters were then randomly assigned to be oriented north-south or east-west (two of each; Figure 3-1B). Thus each of the six treatments were replicated two times (sub sample) for a total of 12 treatment plots in each experimental plot (with 2 experimental plots in each of the 5 blocks) (Figure 3-1A).

For the CWD shelter, stem sections (10 cm diameter, 40 cm length) of balsam poplar (*Populus balsamifera)* were used. Rectangular standard concrete pavestone bricks (10 x 20 x 5 cm in size) were used to build the inert brick shelter to the same size as the CWD by gluing one brick on top of another, and two of these brick pairs were placed end to end in the field (

Figure 3-2). Four native herbaceous species (*Bromus ciliatus* L., *Leymus innovatus* (Beal) Pilg., *Dalea purpurea* Vent., and *Vicia americana* Muhl. ex Willd) were seeded at three distances from each shelter type (0 – 2 cm, 2 – 7 cm, and 7 – 22 cm). These species were chosen because they are native to Alberta wildlands areas, and are commercially available for reclamation purposes. In order to maximize the number of seeds remaining in the soil once planted, seeds were sorted prior to germination testing and field seeding using a seed blower to separate the lightest from the heaviest seeds. Only the heaviest seeds were sorted, counted and used during the experiment.

3.2.3 Seed viability test

Prior to field planting, a germination study was executed to test the viability of the purchased seeds. The study ran for a total of 69 days beginning on June 7, 2013. Termination of the study occurred when no new germinants were observed for any species over a period of one week. Each germination tray had a total of 200 seeds with 50 seeds from each of the four species, and was replicated 8 times. The seeds were sprinkled onto KIMPACK 22-ply filter paper (one species in each quadrant of the tray) and were saturated with water. Trays were placed in a growth chamber with an 18-hour photoperiod (20°C daytime and 16°C nighttime temperature). To simulate initial soil germinating conditions, trays were kept dark for the first two days then were exposed to the regular photoperiod for the remainder of the germination study. Seeds were misted with water to ensure proper moisture availability, and counts of germinated seeds were recorded daily. The average number of seeds germinated over the 69-day time period was 347 (87%), 272 (68%), 290 (73%) and 186 (47%) for species *B. ciliatus, L. innovatus, D. purpurea* and *V. americana*, respectively (Figure 3-3).

3.2.4 Field seeding and data collection

Field seeding took place June 24-28, 2013; all debris and vegetation present was removed from the treatment plot areas and the soil was disturbed using a small garden hand rake. Shelter materials were placed in position, and planting guides were used to ensure seeds were placed at the proper distance and incorporated into the soil.

Shelter plots were placed in three rows with four shelter plots per row, and aspect was randomly assigned (Figure 3-1A). In each shelter plot the area to each side of the shelter item was subdivided into four subplots into which the four species were sown. The two grasses (*B. ciliatus and L. innovatus)* were sown next to one another as were the two legumes (*D. purpurea,* and *V. americana*); this was to minimize potential interference between species. The planting guide was 10 cm wide and 22 cm long from the inside margins and divided into subsections for the three distances from the shelter: 0 – 2 cm, 2 – 7 cm, and 7 – 22 cm. To keep seeding density constant eight seeds were sown in the 20 cm^2 area, 12 seeds in the 50 cm^2 area, and 40 seeds in the 150 cm^2 area (Figure 3-1B).

Germinant counts (emerging seedlings) were conducted weekly in July 2013 and twice per month in August and September 2013. In 2014, germinant counts were conducted monthly from May to August. At the end of August 2014 height was measured on up to eight individuals of the seeded species for each aspect and distance in each treatment plot, then all the above ground plant material of the planted seedlings was harvested to be dried and weighed. Number of individuals of different volunteer plant species (any species other than what was sown) were determined for each aspect and distance from the shelter material (combining the subplots of the four sown species), harvested and weighed; these counts were divided by the area of the distance plot, to calculate densities of volunteer plants per unit area. A full list of volunteer plant species present in treatment plots can be found in Table A 3. Percent germination of seeded species was calculated for each species as the count of individuals present on the last sampling day in 2013 as a percentage of the number of sown seeds. Establishment (survival from the first to the second growing season) was calculated as the number of individuals present on the last sampling day in 2014 as a percentage of the number of germinated individuals on the last sampling day in 2013.

3.2.5 Statistical analysis

All statistical analyses were conducted using R software, version 3.2.2 (R Core Team 2016). To test for treatment effects on percent germination, establishment, height, and weight, a linear mixed model was used for each sown species separately. To test for treatment effects on total density of volunteer species, the same linear mixed model was used (without the need for separate analysis by sown species).

The influence of shelter type (S), aspect (A), distance to shelter (D), and their interactions were examined. This was analyzed as a split-split plot design with pooled errors, including block, experimental plot (within block) and shelter plot (within experimental plot) as random factors and shelter treatments as fixed effects as follows: three levels of shelter type (fixed) as the main plot, four levels of aspect (fixed) as the split plot, and three levels of distance (fixed) as the split-split plot. Analyses were completed using the lme function from the nlme package (Pinheiro et al. 2015). Because germination and establishment of *D. purpurea* and *V. americana* were poor there were many plots with no value for height or dry weight; thus, analyses of height and weight for these species were run using the simplest linear mixed model (no interactions) as there were not enough data to include the interaction terms in the model.

For all analyses, the residuals were assessed for conformity to the assumptions of normality and homogeneity of variance. If they were found to violate these assumptions, any outliers were removed, or the data were transformed using a ln+1 or square root transformation. Ln+1 transformations were used for germination, height, establishment of *B. ciliatus, L. innovatus*, and *D. purpurea*, height of *V. americana,* and weight of *D. purpurea* and *V. americana*. Square root transformations were used for establishment of *V. americana,* weight of *B. ciliatus* and density of volunteer species. Alpha was set at 0.05. If any statistically significant effects were found, post-hoc tests were conducted to compare between means as appropriate for the question at hand and with family-wise Bonferroni correction of alpha (Table 3-1). When an interaction was significant comparisons were made between levels for a treatment holding the level of the other treatment constant. Post-hoc comparisons of aspect involved only comparing between North vs. South, and East vs. West as comparison between South vs. East etc. were not considered to be of interest. For details see Table 3-1.

3.3 Results

3.3.1 Impact of shelter type, aspect, and distance from shelter

3.3.1.1 Bromus ciliatus

Overall germination of *B. ciliatus* in the field was 4% (Figure 3-3). Shelter type and distance to shelter had no effect on germination of *B. ciliatus* but aspect did (see Table 3-2). Germination was higher in areas positioned on the north side of the shelter compared to the south side while there was no difference in germination between east and west aspects (Figure 3-4A).

After the second growing season 23% of the *B. ciliatus* seedlings that were present after the first growing season had survived and were established. Shelter type had no effect on *B. ciliatus* establishment but aspect and distance did *(*see Table 3-2). Establishment was higher in areas positioned on the north side of the shelter compared to the south side (Figure 3-4B), and was greatest furthest away from the shelter (7-22 cm compared to 0-2 cm or 2-7 cm, which did not differ from one another; Figure 3-4C).

Shelter type and aspect had no effect on dry weight of *B. ciliatus* and while there was a significant effect of distance from shelter on average dry weight of *B. ciliatus* in the linear model post-hoc tests did not reveal significant differences (not shown, see Table 3-2). Shelter had no effect on height of *B. ciliatus* however there was a significant aspect by distance interaction effect (see Table 3-2). In areas at a distance of 0-2 cm, average height was greatest in areas positioned on the east side of the shelter compared to the west side; there were no other effects of aspect for the distances (Figure 3-5A). In areas positioned on the west side of the shelter, average height was greatest further away from the shelter compared to closest (7-22 cm and 2-7 cm compared to 0-2 cm, which did not differ from one another); there were no effects of distance for the north, south or east aspects (Figure 3-5B).

3.3.1.2 Leymus innovatus

Overall germination of *L. innovatus* in the field was 8% (Figure 3-3). Distance to shelter had no effect on germination of *L. innovatus* seed but there was a significant shelter by aspect interaction (Table 3-2). When a shelter was present germination was higher on the north side than on the south side while east and west showed no difference (Figure 3-6A). For plots on the north or east side of the shelter germination was higher for wood and brick than for no shelter while for the south and west aspects there was no effect of shelter material (Figure 3-6B).

Overall establishment of *L. innovatus after* the second growing season was the highest out of the four seeded species, with 51% of the seedlings surviving from the first to the second growing season. Aspect had no effect on establishment of *L. innovatus* seedlings but there were significant effects of shelter type and distance (Table 3-2). Establishment was higher with brick compared to wood and no shelter (Figure 3-7A) and increased with distance from shelter (Figure 3-7B).

Average dry weight of *L. innovatus* was higher in plots at a distance of 7-22 cm than 0-2 cm or 2-7 cm (Figure 3-7C), however shelter type and aspect had no effect on dry weight of *L. innovatus* (Table 3-2). Average height of *L. innovatus* was higher with brick compared to no shelter (Figure 3-8A), closer to shelter than further away (0-2 cm *versus* 7-22 cm) (Figure 3-8B), and higher on the east side compared to the west side while north and south showed no difference (Figure 3-8C).

3.3.1.3 Dalea purpurea

Overall germination of *D. purpurea* in the field was 10% (Figure 3-3). Aspect and distance to shelter had no effect on germination of *D. purpurea* seed (Table 3-2). Germination was higher for both types of shelter material as compared to no shelter (Figure 3-9A).

Overall establishment of *D. purpurea* after the second growing season was 14% of the seedlings that were present after the first growing season. Shelter type and aspect had no effect on second year establishment of *D. purpurea* (see Table 3-2). Establishment was greater furthest away from the shelter (7-22 cm) compared to the two closer distances (0-2 cm and 2-7 cm; Figure 3-9B).

Average dry weight of *D. purpurea* was higher in plots at 7-22 cm away from shelter than 2-7 cm (Figure 3-9C). Shelter type and aspect had no effect on dry weight of *D. purpurea* (see Table 3-2). Average height of *D. purpurea* was not affected by shelter type, aspect or distance from shelter (not shown, Table 3-2).

3.3.1.4 Vicia americana

Overall germination of *V. americana* in the field was 13% (Figure 3-3). Germination was higher on the north side than on the south side of shelter (Figure 3-10A) and there was a significant shelter by distance interaction (Table 3-2). In plots with the wood shelter treatment average height was higher in plots closest to shelter (0-2 cm compared to 2-7 cm and 7-22 cm which did not differ from one another); additionally, in plots with the brick shelter treatment germination was higher at 2-7 cm than at 0-2 cm or 7-22 cm (Figure 3-11A). At a distance of 0-2 cm, germination differed among all three shelter treatments; it was higher with wood than with brick, which was higher than no shelter. At a distance of 2-7 cm, germination was higher with the brick shelter compared to either wood or no shelter (Figure 3-11B).

Overall second year establishment of *V. americana* was very low, with only 4% of the first year seedlings surviving. Distance from shelter had no effect on establishment of *V. americana* (Table 3-2). Establishment was greater when shelter was present, regardless of whether it was brick or wood, compared to no shelter (Figure 3-10B), and was higher in areas positioned on the north side of the shelter compared to the south side while east and west aspects did not differ from one another (Figure 3-10C).

Due to the very low establishment, only 70 plants were available to measure average dry weight and height of *V. americana.* Neither dry weight and height were effected by shelter type, aspect, or distance from shelter (not shown, Table 3-2).

3.3.2 Density of volunteer species as affected by shelter type, aspect, and distance from shelter material

Total density of volunteer species was extremely low with an average of 1 plant/100 cm^2 at 0-2 cm away from shelter, 0.5 of a plant/100 cm^2 at 2-7 cm away from shelter, and 0.4 of a plant per 100 cm^2 at 7-22 cm away from shelter. There were significant shelter by distance and aspect by distance interaction effects for the total density of volunteer species (Table 3-3). In plots with brick shelter density was greater closest to the shelter than further away; while distance did not play a role with the other shelter treatments (Figure 3-12A). At a distance of 0-2 cm, total density was greatest near brick shelter compared to wood and no shelter (Figure

3-12B; there were no other effects of shelter with the other distances. In areas at a distance of 0-2 cm, average height of volunteer species was greatest in areas positioned on the south side of the shelter compared to the north side; there were no other effects of aspect for the distances (Figure 3-13A). In areas positioned on the south and east side of the shelter, total density of volunteer species was greatest immediately next to shelter material compared to further away; there were no effects of distance for the north and west aspects (Figure 3-13B).

3.4 Discussion

The goal of this study was to examine the effect of shelter, its orientation and distance from it on germination, establishment, and growth of four seeded species and on the density of volunteer species on a reclamation site with poor growing conditions. Further, I explored whether the shelter effect was influenced by the type of shelter used (e.g. wood versus an inert stone material).

The provision of shelter generally improved germination, establishment, and seedling height of the seeded species and increased the density of volunteer species. Within the short timeframe of this study there were few differences between CWD and brick, and no clear evidence that either type of shelter was preferable as brick sometimes had more favourable results and other times wood did; this indicates that shelter from the harsh conditions was the primary factor in the early establishment and not potential products of decomposition. This appears to be corroborated by the results that found that the orientation of the shelter was also important where the seeded species performed better on the north facing aspect; however, seeded species generally performed best when 7-22 cm from the shelter. Interestingly the volunteer vegetation had greater density on the south facing aspect of a shelter.

Germination of seeded species was considerably lower than expected from the germination test; environmental and edaphic conditions were most likely limiting initial seed germination and growth as growing season total precipitation was very low (271 mm in 2013 and 376 mm in 2014; see Figure B 1), and average volumetric water content was only 3% within the first 10 cm of soil in 2013 (Figure B 2) which is below the permanent wilting point in a silt loam to loam textured soil (Figure B 3). Seeds were sown into shelter plots the week of June 24-28, 2013 and experienced a total of only 40 mm of rain during July, with minimal precipitation during August until a large rainfall event August 28-30, 2013 (Figure B 1). This low precipitation immediately following planting, as well as during the last half of the first growing season most likely led to the moderately dry conditions (Agriculture and Agri-Food Canada 2013) present on the reclamation site and may have contributed to the poor survival of germinants and growth of seeded vegetation.

3.4.1 Effect of shelter and shelter material type

The first hypothesis stated that the provision of shelter, from either wood or brick materials, would increase performance of seeded vegetation and density of volunteer species compared to areas with no shelter. This study demonstrates that the provision of shelter improved the germination, establishment and growth of seeded species and resulted in higher density of volunteer species. Potential benefits of shelter can include protection from wind, increased shade, increased capture of windblown seeds, and act as germination sites thus increasing productivity as a direct physical barrier and through indirect microsite effects (Carlsson and Callaghan 1991, Bird et al. 1992, Cleugh et al. 1998, Whisenant 2002, Buchman et al. 2007). When shelter was present germination of *L. innovatus, D purpurea*, and *V. americana*, establishment of *L. innovatus* and *V. americana*, and height of *L. innovatus* were greater. There was an increase of the total density of volunteer species closer to shelter materials. The effects of brick were occasionally positive as compared to wood while other times the opposite was true; however, the presence of shelter was generally beneficial compared to no shelter. It appears that the provision of shelter (from either wood or brick) can facilitate vegetation establishment by ameliorating the harsh environmental conditions such as strong winds, high temperature and poor moisture conditions found on reclamation sites.

Contrary to our second hypothesis, there were no clear differences between wood and brick shelter materials on an early reclamation site. CWD has a variety of benefits other than just providing shelter, such as contributing to nutrient cycling, retaining moisture, and providing sites for litter accumulation and seed capture (Harmon et al. 1986, Lindenmayer et al. 1999, 2002, Whisenant 2002, Opdam et al. 2006, Debeljak 2006, Manning et al. 2013). Alternatively, a benefit of using inert materials such as rocks or concrete bricks for reclamation is they may act as an energy sink during the day when solar energy is highest, while acting as an energy source later in the day when solar energy is lower (Brown 1969). These inert materials may also redistribute incoming precipitation although they are not as porous of a substrate as CWD. Within the short timeframe of this study, the benefits from decomposition and moisture inputs of CWD may not be apparent as these are complex, long term processes which can take from 5 to 200 years depending on size, species and fungal communities present (Harmon et al. 1986, Pyle and Brown 1999, Tarasov and Birdsey 2001). The positive effects of brick were occasionally greater than those of wood (establishment of *L. innovatus*, germination of *V. americana* at a distance of 2-7 cm; density of volunteer species close to the shelter) while for other response variables the effect of wood was more favourable than brick (germination of *V. americana* (close to shelter)). These differences between wood and brick usually occurred when there were other factors influencing the vegetation, such as aspect or distance to shelter. This demonstrates that the two types of shelter materials do differ in terms of their effects on microclimate but that these differences depend on other factors such as aspect and distance, resulting in more complex effects on vegetation establishment. Seeded native vegetation generally performed best between 7 – 22 cm away from shelter (establishment of *B. ciliatus*, and dry weight of *L. innovatus* and *D. purpurea*); this distance may provide an ideal balance between the amount of sun and shade throughout the day, potentially regulating air and soil temperature.

Using an inert material may initially provide similar benefits as CWD for vegetation establishment and growth through the provision of shelter. Continued short and long-term studies using CWD and inert materials as shelters are necessary to further our understanding about vegetation responses to shelter and to inform the development of various strategies for reclamation of sites with challenging growing conditions.

3.4.2 Effects of aspect and distance from shelter

In accordance with our third hypothesis, the north aspect did improve performance of seeded native species, yet in contrast to this, volunteer vegetation preferentially grew on the south aspect. Our fourth hypothesis stated that performance and growth will decrease with the distance from the shelter material, however it was found that generally greater benefits of shelter were found at farther distances. Nonetheless, occasionally the benefits of shelter were greater closer to the shelter, which conformed to the original hypothesis.

Native species experienced the greatest benefits at a distance of 7-22 cm from shelter, while volunteer vegetation generally benefited from being closer to shelter on southern or eastern aspects where conditions would be warmer and drier. Volunteer species found on the site were predominantly introduced forbs such as Russian thistle (*Salsola tragus* L.), Kochia (*Bassia scoparia* L.), Sweet clover (*Melilotus* Mill.), Dog mustard (*Erucastrum gallicum* (Willd.) O.E. Schulz), and Sow thistle (*Sonchus* L.). These species are primarily found growing in areas that usually have dry soil and harsh conditions (Turkington et al. 1977, Lemna and Messersmith 1990, Warwick and Wall 1998, Beckie and Francis 2009, Friesen et al. 2009). In contrast, the northern aspect of a shelter appeared to have the greatest benefit for seeded vegetation. Germination of *B. ciliatus, L. innovatus*, and *V. americana* and establishment of *B. ciliatus* and *V. americana* were the greatest on the north side of a shelter. The eastern and western aspects rarely differed with the exception of height of *B. ciliatus* and *L. innovatus*, while the south aspect had the lowest values for germination of *B. ciliatus, L. innovatus*, and *V. americana* and establishment of *B. ciliatus* and *V. americana*. Increased performance on the northern aspects by vegetation in this study suggest that these seeded native species are favoured by cooler and moisture climates whereas introduced volunteer species are favoured by warmer and drier climates.

In terms of microclimate, northern aspects are typically cooler and have higher moisture, whereas southern aspects are typically warmer with less moisture (Åström et al. 2007, González-Alday et al. 2008, Macdonald et al. 2015a, Måren et al. 2015). Areas on the north aspect will be shaded for long periods during the day whereas areas to the south receive a significant amount of direct sunlight and may experience solar reflection off the shelter throughout the day (Rosenberg et al. 1983a, Jones 2014). If shelter is oriented north-south, areas on the eastern or western aspect receive smaller amounts of shade throughout the day, specifically during the growing season when the sun is high; however, areas shaded in the morning on the western aspect will receive additional sunlight in the afternoon (Rosenberg et al. 1983a). These differences occur as a result of the amount of solar radiation received at the soil surface (Leij et al. 2004, González-Alday et al. 2008). Seeded species benefited by being on the northern side of the shelter, as conditions were favourable for germination and establishment. Microclimate effects (lower temperatures and increased moisture) may have extended further away from the shelter as the length of shade is increased compared to the eastern or western aspects. Volunteer vegetation benefitted from being closer to shelter on the southern aspect; these species are more suited to growing in the warmer and drier conditions, and as the wind direction on the research site is primarily coming from the north-west (Alberta Agriculture and Forestry 2015), seeds of these species may have been trapped and accumulated close to the shelters, as they rely on wind as a means of seed dispersal (Turkington et al. 1977, Lemna and Messersmith 1990, Warwick and Wall 1998, Beckie and Francis 2009, Friesen et al. 2009). Vegetation growing near shelter often shows more rapid growth and increased size due to the increased temperature and decreased potential evaporation, which can lead to higher stomatal conductance and rapid photosynthesis near the shelter (Rosenberg et al. 1983a, Jones 2014). The microclimate conditions and interactions between wind and aspect of the shelter may have been more favourable to these wind dispersed species.

The distance at which the greatest benefit of shelter can be detected varies depending on potential interactions between species, shelter type, and aspect. At a distance of 7 – 22 cm away from shelter overall benefits were greatest. However, when interactions with aspect or shelter type occurred generally benefits were greater closer to the shelter. At times, greater benefits were found closer to the shelter as seen for germination of *V. americana* (with wood and brick shelter material), height of *L. innovatus*, and density of volunteer species (with brick shelter on southern and eastern aspects). The distance at which these effects were strongly favoured was generally between 0 – 7 cm away from shelter. Research done by Goldin and Hutchinson (2014) found that soil moisture loss immediately adjacent to CWD was approximately 40% slower than at locations farther away (up to 80 cm), indicating that surface soils near CWD may be protected from moisture loss. This research provides some evidence that perhaps once established, height growth would be better closer to the shelter (height of *L. innovatus*) perhaps because greater soil moisture near CWD results in higher photosynthesis and therefore growth (Rosenberg et al. 1983b, Goldin and Hutchinson 2014).

The influence of aspect and distance from shelter can benefit the performance of emerging vegetation though small changes in the microclimate. In particular, the northern aspect which has larger areas under shaded conditions, cooler temperatures and higher moisture levels, favoured native seeded species at intermediate distances; conversely the southern aspect would be characterized by increased sunlight, warmer temperatures, and drier soil conditions which favoured introduced volunteer vegetation close to the shelter. Using shelter as a reclamation tool may help promote native species when placed in a manner which optimizes the amount of area shaded on a northern aspect. These complex interactions between species type, shelter type, aspect of shelter and distance from shelter all demonstrate the need for microsite heterogeneity in reclamation practices in order to encourage and facilitate natural revegetation as it affects hydrological conditions and soil properties (Mackenzie and Naeth 2010, Simmons et al. 2012, Goldin and Hutchinson 2013, Macdonald et al. 2015a).

3.5 Conclusions

In summary, the provision of shelter did generally improve performance of seeded native species and density of volunteer species, however effects differed somewhat among species and with aspect and distance from shelter. No clear evidence was found that either type of shelter material (CWD or brick) was better; at such an early stage in site development, there is most likely a physical effect from the shelter rather than a chemical one as decomposition was not seen within the time frame of this study. Microsite heterogeneity is important for facilitating natural revegetation on harsh reclamation sites as it affects hydrological conditions and soil properties (Mackenzie and Naeth 2010, Simmons et al. 2012, Goldin and Hutchinson 2013, Macdonald et al. 2015a). Exposure to the north facing aspect generally increased performance of seeded native species while exposure to the south resulted in greater density of volunteer vegetation. Further, benefits of shelter seeded species were best at farther distances, yet volunteer vegetation benefited most when closer to shelter, which may be an artifact as volunteer vegetation seeds are air born and may be trapped by shelter whereas seeded vegetation was seeded and buried at select distances.

On harsh reclamation sites, immediate establishment of vegetation is a short-term priority. In order to increase vegetation presence and microsite heterogeneity using shelter, management practices could include the creation of small areas with shelter materials placed as little as 22 cm away from each other in a north-south direction thus emphasizing eastern and western aspects. This maximizes the sheltering effect for vegetation and minimizes aspect effects, specifically reducing the potential benefits of southern aspects for introduced volunteer vegetation. Long term shelter placement could involve using pieces of CWD material placed 22 cm away from each other with all aspects represented. This provides sheltered areas with the northern aspect which native species benefit from, while also perhaps reducing the presence of introduced volunteer vegetation on the southern aspect by only using CWD material rather than brick which benefitted volunteer vegetation more. The benefits of shelter in this study extended up to 22 cm away from the shelter object; perhaps further studies examining distances beyond this limit may be able to better define the distance at which shelter benefits are no longer found.

3.6 Tables

Table 3-1: Post hoc comparisons of means, and how alpha was adjusted for these, following significant effects in the ANOVAs examining the influence of shelter, aspect, distance and their interactions on germination, establishment, dry weight, and height of sown species and on abundance of volunteer vegetation. Note: 'C' represent control shelter treatment, 'Wo' represent wood shelter treatment, 'B' represent brick shelter treatment, 'N' represents North, 'S' represents South, 'E' represents East, 'W' represents West, '0-2', '2-7' and '7-22' represent the distances (in cm) from shelter material.

Significant Effect		Alpha value	Pairwise Comparisons
Shelter	Pairwise between three treatments	α = 0.05 / 3 = 0.017	C vs. Wo, C vs. B, Wo vs. B
Aspect	North *vs* South; East *vs* West	α = 0.05 / 2 = 0.025	North vs. South, East vs. West
Distance	Pairwise between three distances	α = 0.05 / 3 = 0.017	0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22
Shelter x Aspect	Between Shelter treatments within Aspect	α = 0.05 / 3 = 0.017	N: C vs. Wo, C vs. B, Wo vs. B S: C vs. Wo, C vs. B, Wo vs. B E: C vs. Wo, C vs. B, Wo vs. B W: C vs. Wo, C vs. B, Wo vs. B
	Between Aspects within Shelter	α = 0.05 / 2 = 0.025	C: N vs. S, E vs. W Wo: N vs. S, E vs. W B: N vs. S, E vs. W
Shelter x Distance	Between Distances within Shelter	α = 0.05 / 3 = 0.017	C: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22 Wo: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22 B: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22
	Between Shelter treatments within Distance	α = 0.05 / 3 = 0.017	0-2: C vs. Wo, C vs. B, Wo vs. B 2-7: C vs. Wo, C vs. B, Wo vs. B 7-22: C vs. Wo, C vs. B, Wo vs. B
Aspect x Distance	Between Distances within Aspect	α = 0.05 / 3 = 0.017	N: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22 S: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22 E: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22 W: 0-2 vs. 2-7, 0-2 vs. 7-22, 2-7 vs. 7-22
	Between Aspects within Distance	α = 0.05 / 2 = 0.025	0-2: N vs. S, E vs. W 2-7: N vs. S, E vs. W 7-22: N vs. S, E vs. W

Table 3-2: Results of linear mixed effects model ANOVAs examining the influence of shelter, aspect, distance and their interactions on germination, establishment, dry weight and height of four native species sown at the Whitewood coal mine reclamation site. P values are given and bolded when significant (α=0.05). Note: n/a represents interactions not tested in simplest version of the model (dry weight and height of *D. purpurea and V. americana*).

Response Variable	Germination			
	B. ciliatus	*L. innovatus*	*D. purpurea*	*V. americana*
Shelter	0.081	**0.001**	**0.01**	**<0.001**
Aspect	**0.035**	**0.001**	0.423	**0.001**
Distance	0.094	0.453	0.074	**0.001**
Shelter x Aspect	0.446	**0.041**	0.215	0.122
Shelter x Distance	0.53	0.061	0.394	**<0.001**
Aspect x Distance	0.087	0.286	0.312	0.483
Shelter x Aspect x Distance	0.828	0.147	0.417	0.059
	Establishment			
	B. ciliatus	*L. innovatus*	*D. purpurea*	*V. americana*
Shelter	0.324	**0.003**	0.687	**0.001**
Aspect	**0.047**	0.133	0.473	**0.003**
Distance	**0.001**	**<0.001**	**<0.001**	0.523
Shelter x Aspect	0.138	0.216	0.218	0.603
Shelter x Distance	0.432	0.341	0.787	0.879
Aspect x Distance	0.538	0.183	0.408	0.332
Shelter x Aspect x Distance	0.439	0.061	0.775	0.830
	Dry weight			
	B. ciliatus	*L. innovatus*	*D. purpurea*	*V. americana*
Shelter	0.640	0.945	0.950	0.509
Aspect	0.464	0.602	0.172	0.832
Distance	**0.039**	**0.001**	**0.008**	0.115
Shelter x Aspect	0.752	0.243	n/a	n/a
Shelter x Distance	0.545	0.907	n/a	n/a
Aspect x Distance	0.085	0.650	n/a	n/a
Shelter x Aspect x Distance	0.611	0.063	n/a	n/a
	Height			
	B. ciliatus	*L. innovatus*	*D. purpurea*	*V. americana*
Shelter	0.853	**0.005**	0.641	0.906
Aspect	0.381	**0.004**	0.979	0.084
Distance	0.519	**0.001**	0.735	0.386
Shelter x Aspect	0.432	0.068	n/a	n/a
Shelter x Distance	0.066	0.989	n/a	n/a
Aspect x Distance	**0.018**	0.910	n/a	n/a
Shelter x Aspect x Distance	0.245	0.832	n/a	n/a

Table 3-3: Results of linear mixed effects model ANOVAs examining the influence of shelter, aspect, distance and their interactions on total density of volunteer species at the Whitewood coal mine reclamation site. P values are given and bolded when significant (α=0.05).

Response Variable	Total Density of Volunteer Species
Shelter	**0.012**
Aspect	0.081
Distance	**<0.001**
Shelter x Aspect	0.104
Shelter x Distance	**0.011**
Aspect x Distance	**0.032**
Shelter x Aspect x Distance	0.103

3.7 Figures

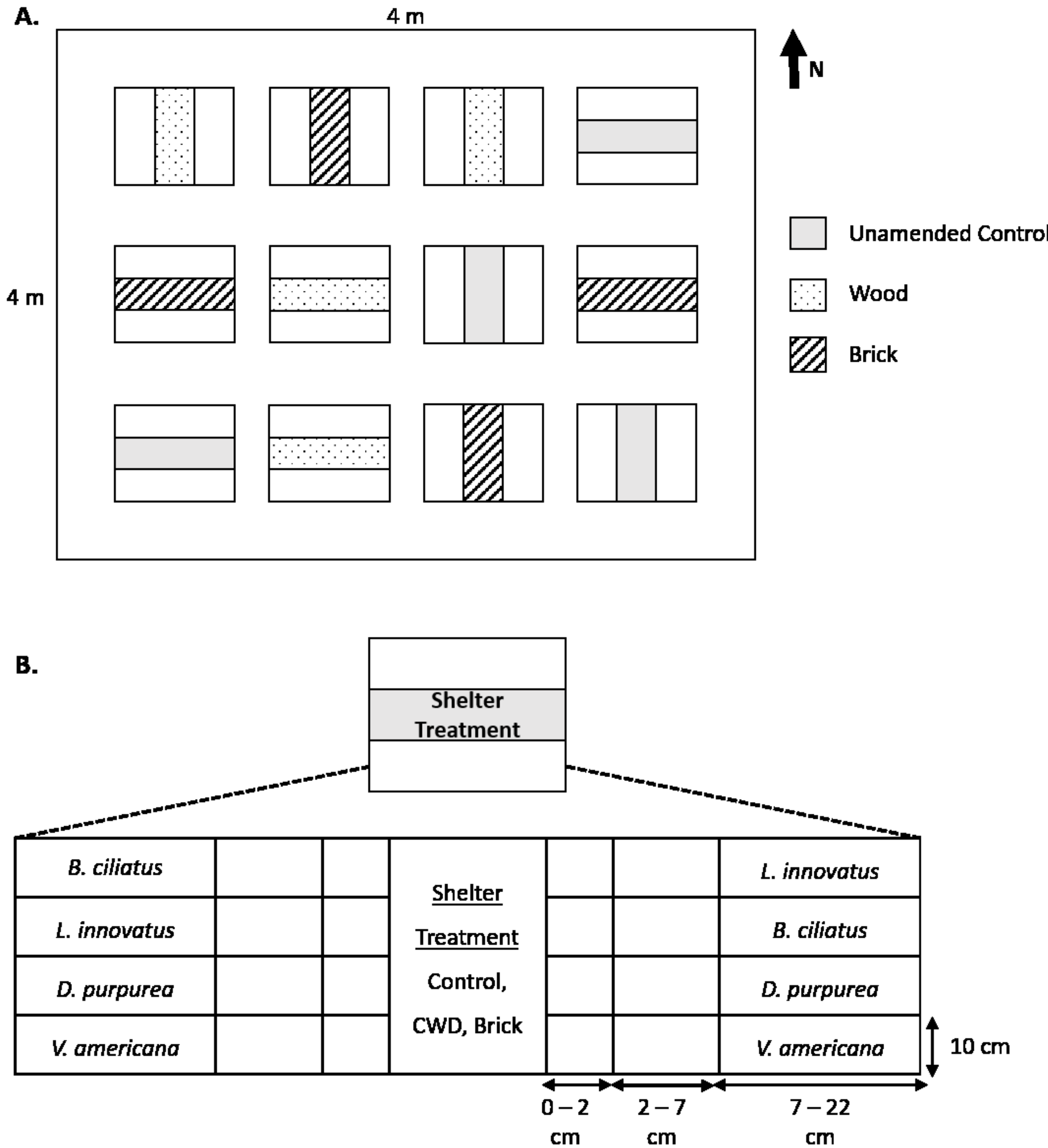


Figure 3-1: Field layout of experimental and shelter plots. **A.** Experimental plot: treatment plots were placed in three rows with four treatment plots per row, and aspect was randomly assigned; there were two replicates per shelter material oriented North/South and the other two oriented East/West; seeds were sown on both sides of the central treatment. **B.** Shelter plot: Includes the four seeded species and seed density for each distance away from the treatment type.

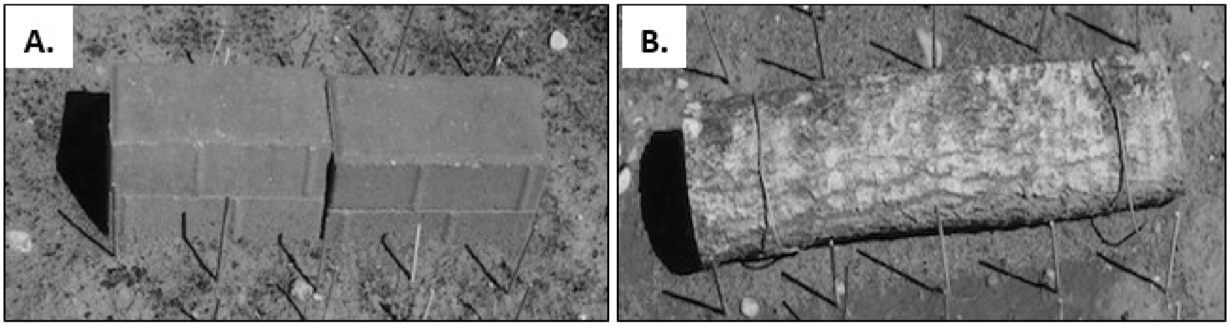


Figure 3-2: Comparison of the brick and wood (CWD) shelter treatment types. **A.** Two brick units placed end to end **B.** Landscaping staples can be seen over logs to ensure they were kept in place over winter.

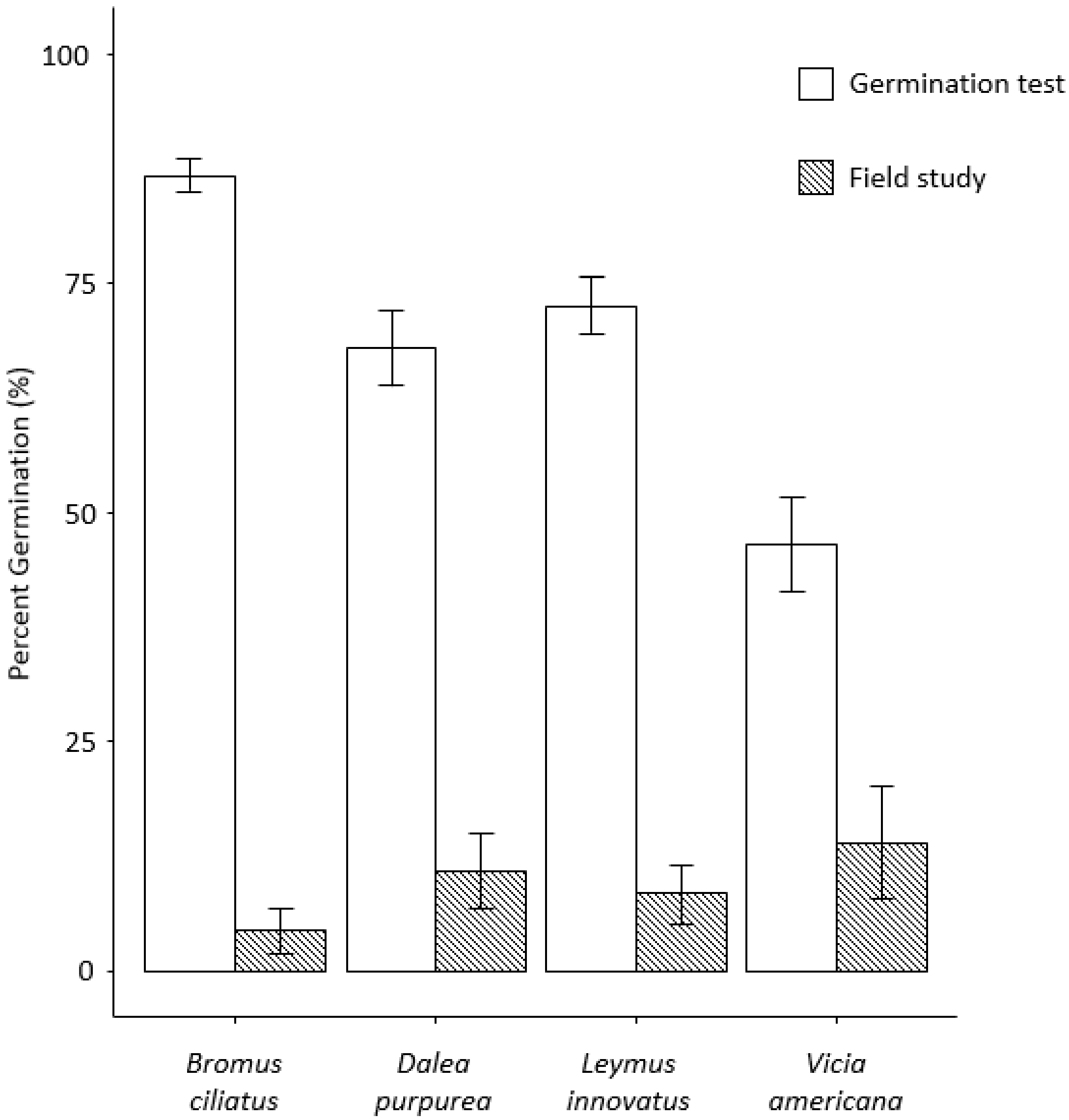


Figure 3-3: Average percent germination of the four seeded species (*B. ciliatus, L. innovatus, D. purpurea*, and *V. americana*) in the germination test and in the field study. Error bars represent the standard error (Germination test n=8, field study n=10).

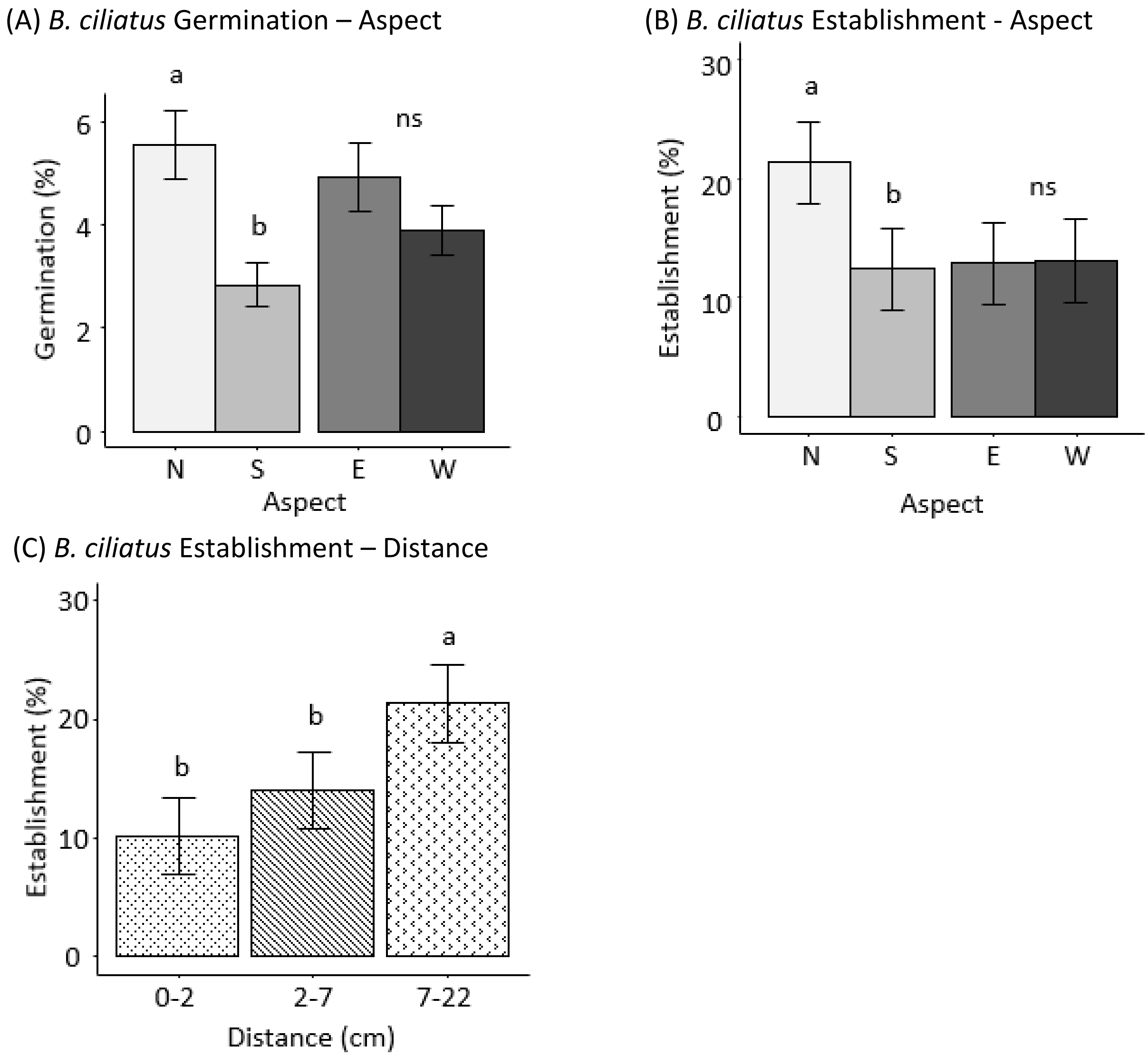
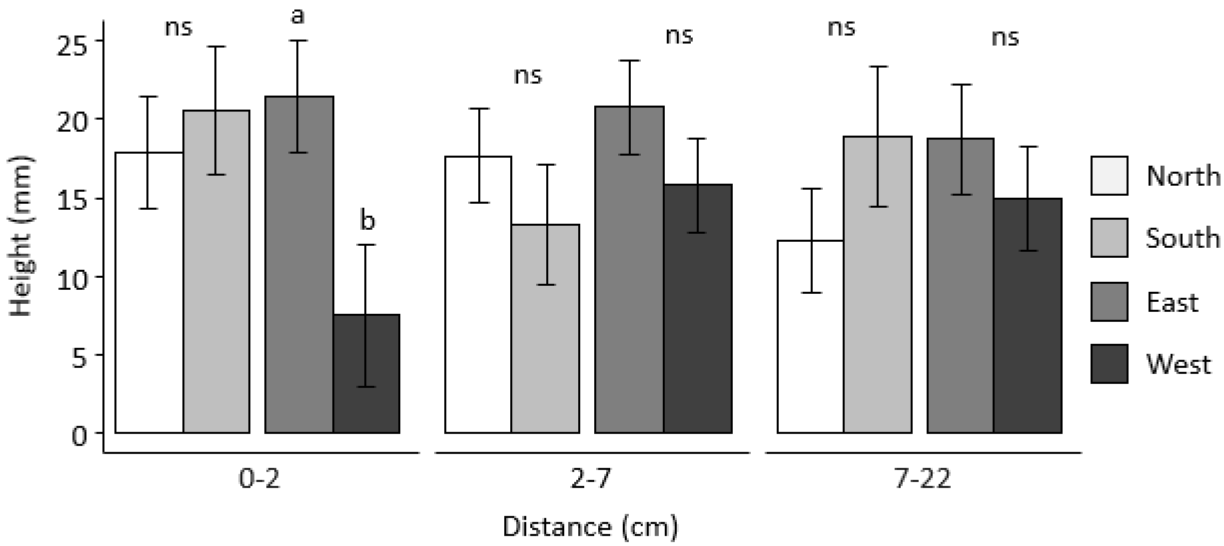


Figure 3-4: Least squared means (and standard error) for responses of *B. ciliatus* presented as (A) germination in response to aspect, (B) establishment in response to aspect of shelter and (C) to distance from shelter. Means with different letters indicate a significant difference between treatments. Note: 'N' represents North, 'S' represents South, 'E' represents East, 'W' represents West, and 'ns' represents not significant.

(A) *B. ciliatus* Height – Aspect x Distance



(B) *B. ciliatus* Height – Aspect x Distance

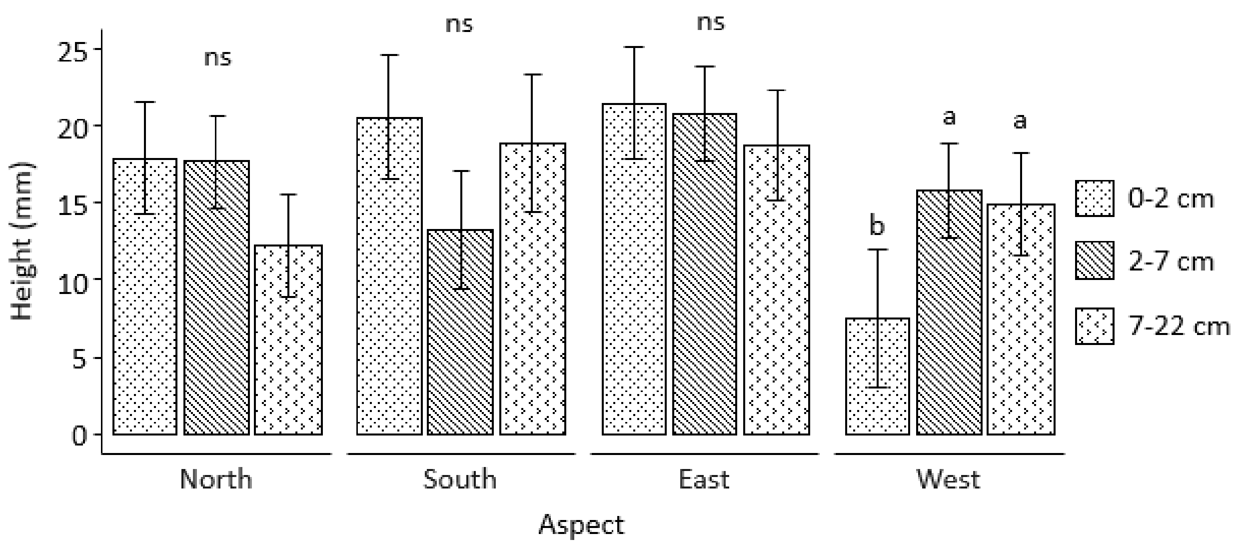
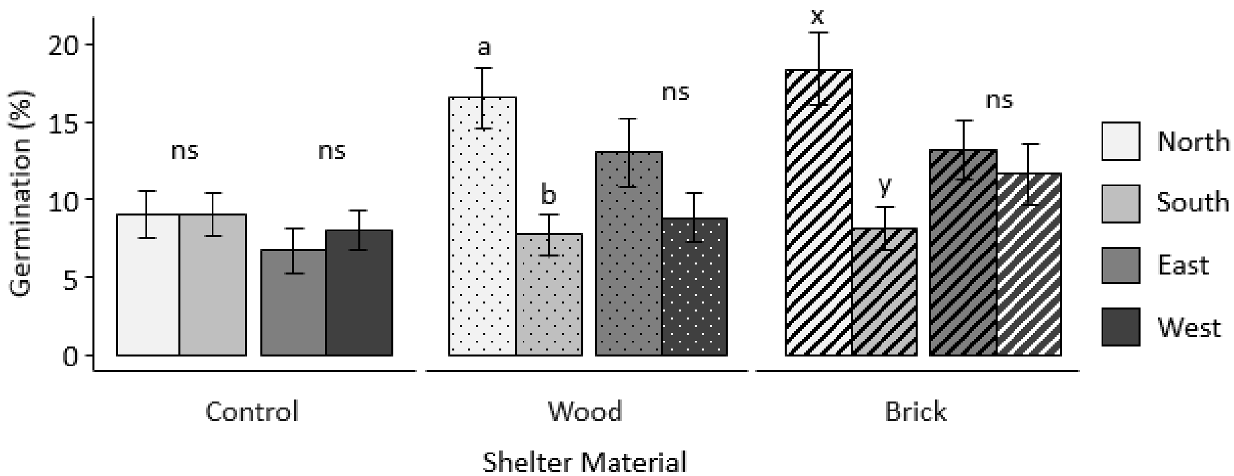


Figure 3-5: Least squared means (and standard error) for height of *B. ciliatus* in response to aspect and distance. (A) Means with different letters indicate a significant difference between the two aspects (north vs. south or east vs. west) within a given distance; (B) Means with different letters indicate a significant difference between distances from shelter for a given aspect. Note: 'ns' represents not significant.

(A) *L. innovatus* Germination – Shelter x Aspect



(B) *L. innovatus* Germination – Shelter x Aspect

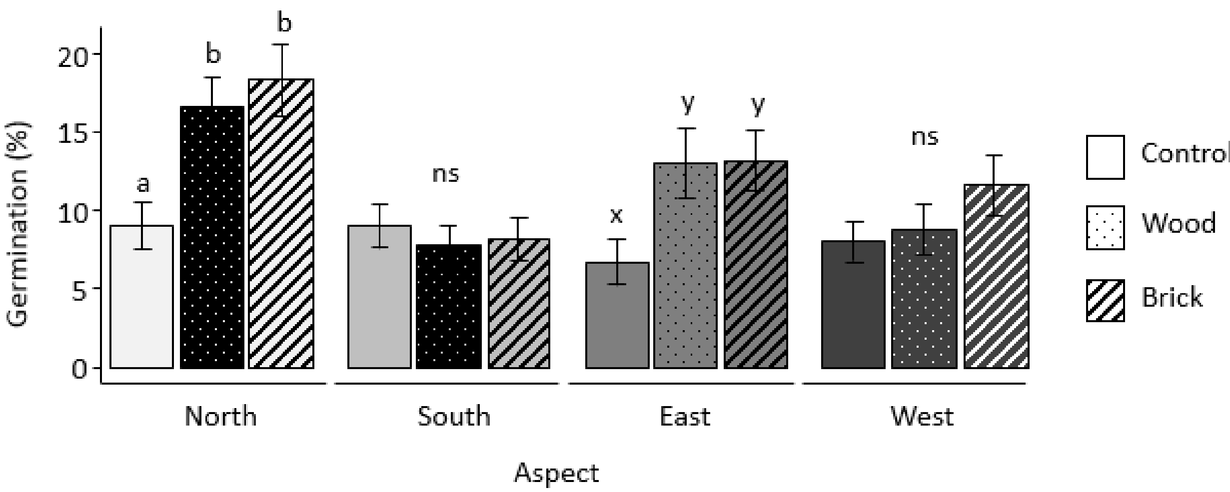


Figure 3-6: Least squared means (and standard error) for germination of *L. innovatus* in response to shelter type and aspect. (A) Means with different letters indicate a significant difference between the two aspects (north vs. south or east vs. west) within a given shelter type; (B) Means with different letters indicate a significant difference between shelter types for a given aspect. Note: 'ns' represents not significant.

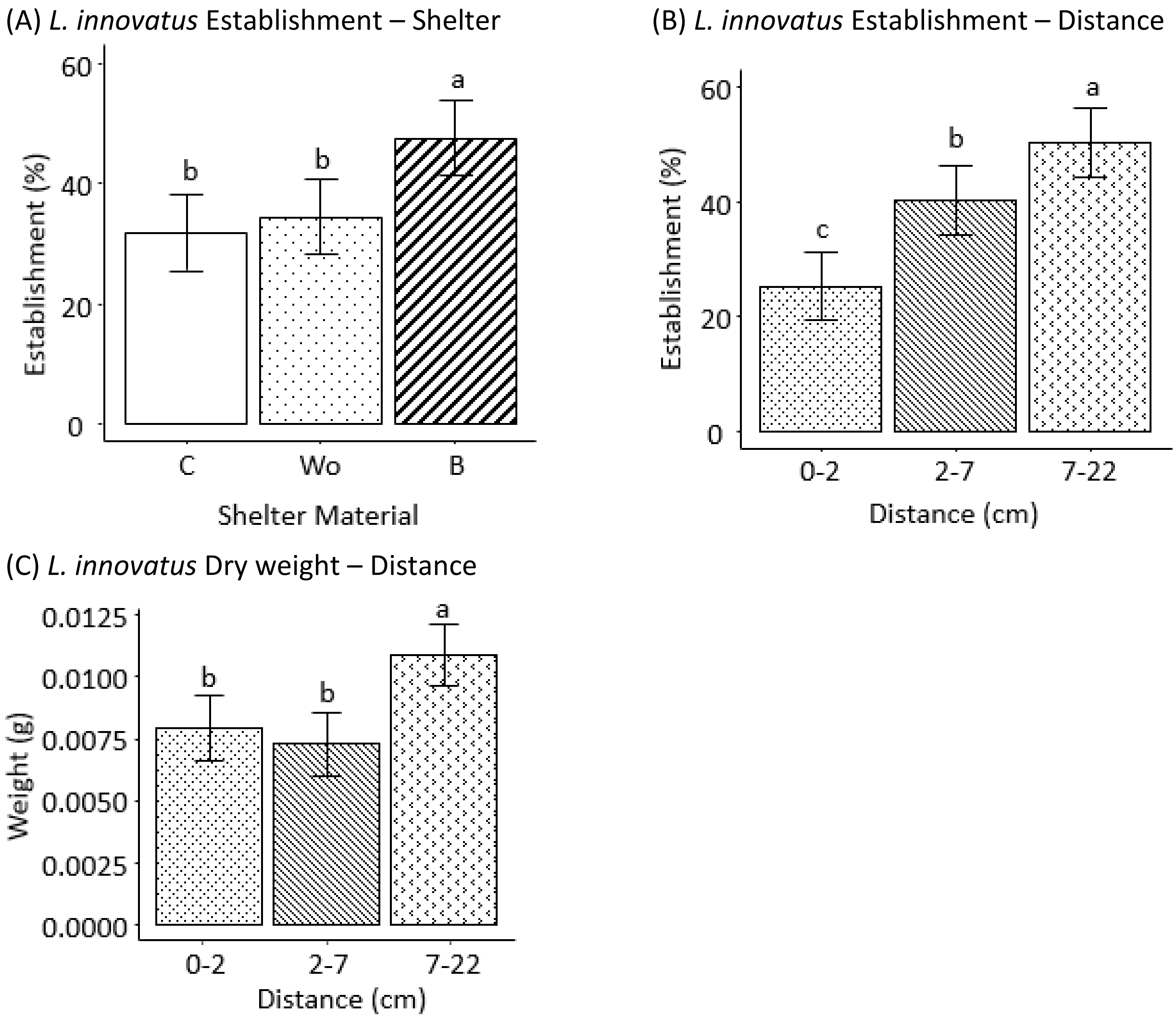
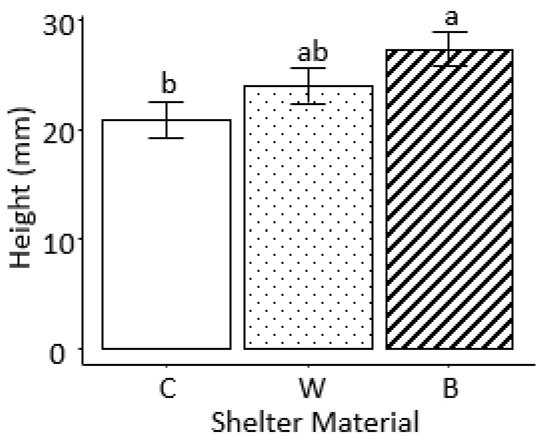
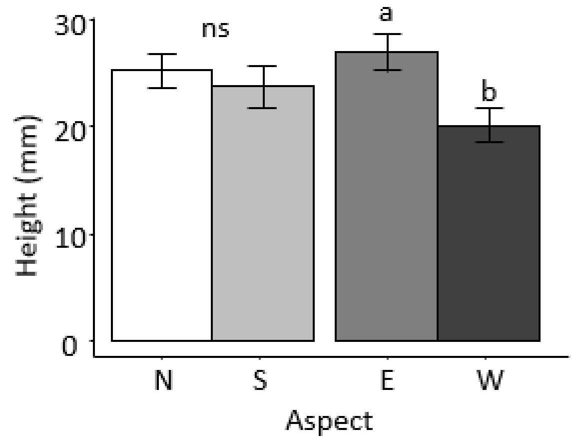


Figure 3-7: Least squared means (and standard error) for responses of *L. innovatus* presented as (A) establishment in response to shelter type, (B) distance from shelter, and (C) dry weight in response to distance from shelter. Means with different letters indicate a significant difference between treatments. Note: 'C' represents no shelter material (control), 'Wo' represents wood shelter material, 'B' represents brick shelter material, '0-2 cm', '2-7 cm' and '7-22 cm' represent the distances from shelter material.

(A) *L. innovatus* Height – Shelter



(B) *L. innovatus* Height – Aspect



(C) *L. innovatus* Height – Distance

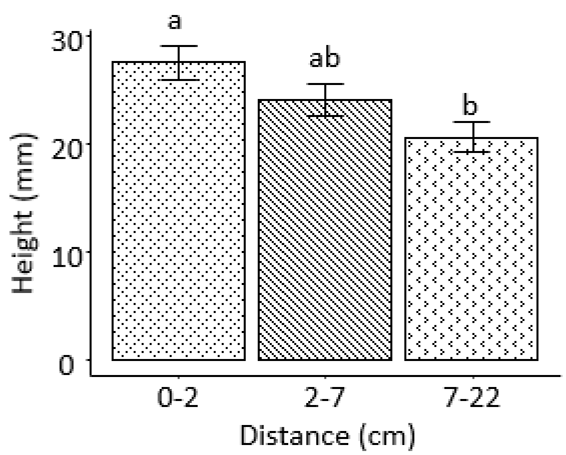


Figure 3-8: Least squared means (and standard error) for responses of *L. innovatus* presented as height in response to (A) shelter type, (B)aspect of shelter, and (C) distance from shelter. Means with different letters indicate a significant difference between treatments. Note: 'N' represents North, 'S' represents South, 'E' represents East, 'W' represents West, 'C' represents no shelter material (control), 'Wo' represents wood shelter material, 'B' represents brick shelter material, '0-2 cm', '2-7 cm' and '7-22 cm' represent the distances from shelter material

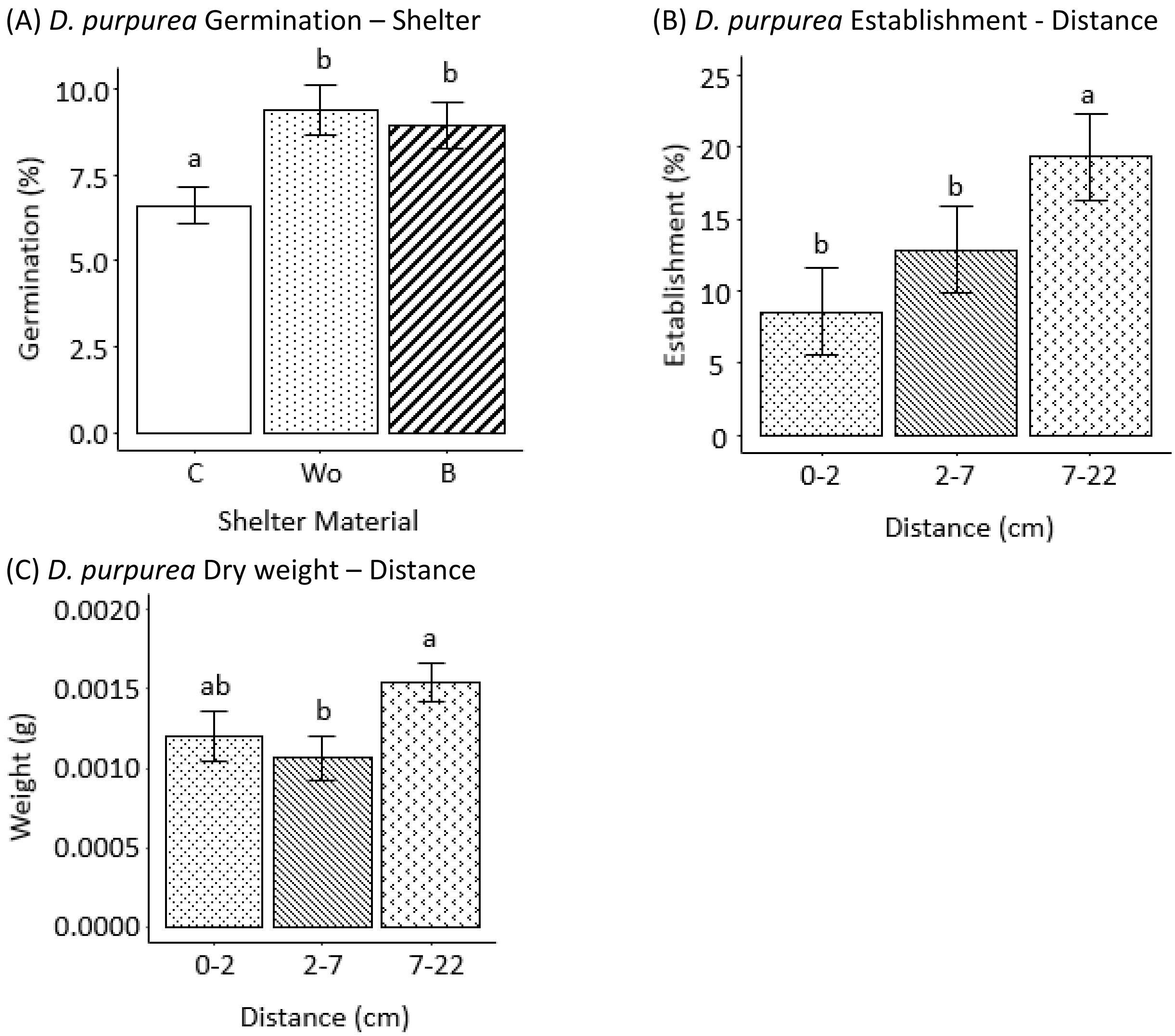


Figure 3-9: Least square means (and standard error) for responses of *D. purpurea* presented as (A) germination in response to shelter type, (B) establishment in response to distance from shelter, and (C) dry weight in response to distance from shelter. Means with different letters indicate a significant difference between treatments. Note: ‘C’ represents no shelter material (control), ‘Wo’ represents wood shelter material, ‘B’ represents brick shelter material, ‘0-2 cm’, ‘2-7 cm’ and ‘7-22 cm’ represent the distances from shelter material.

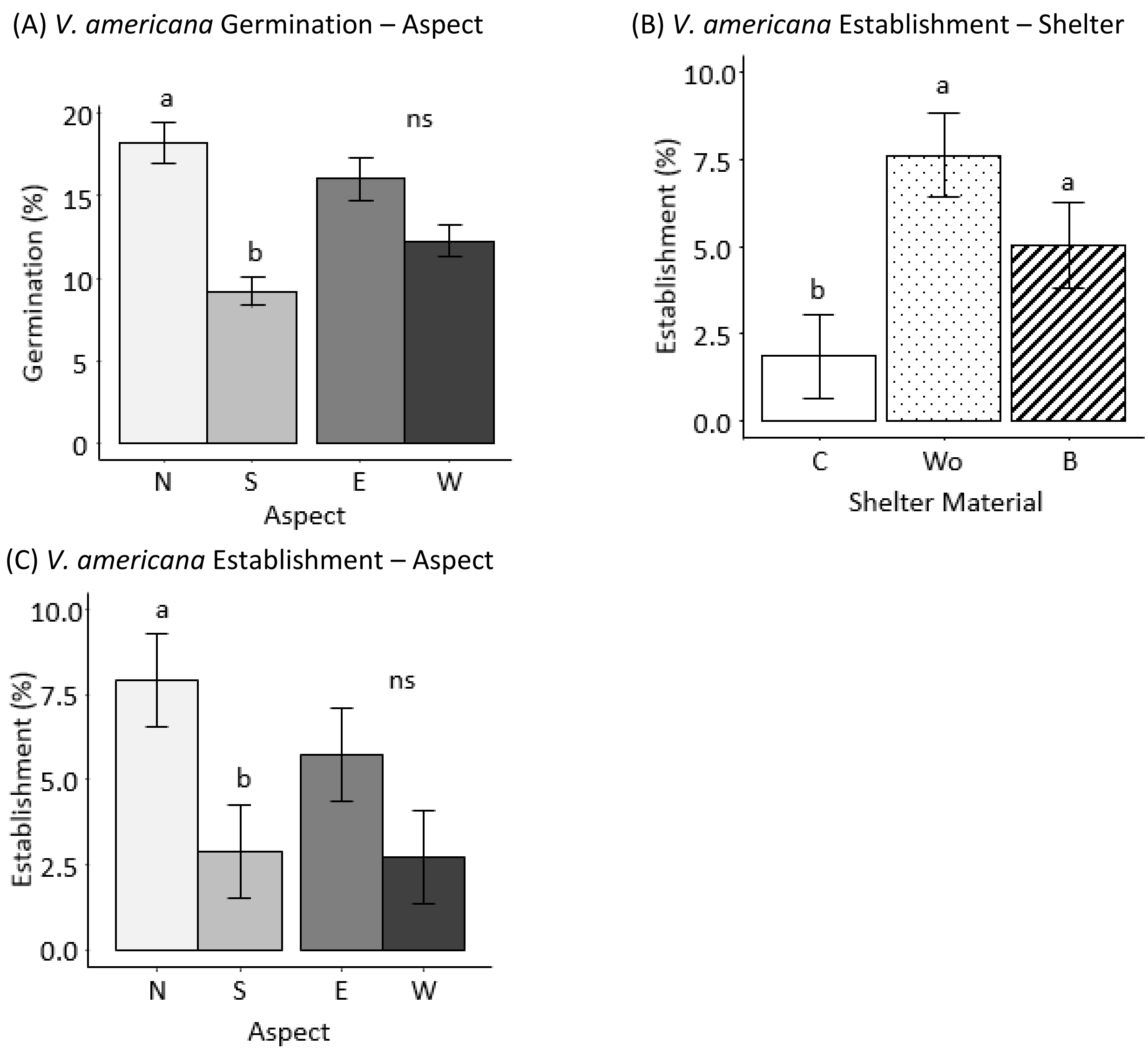
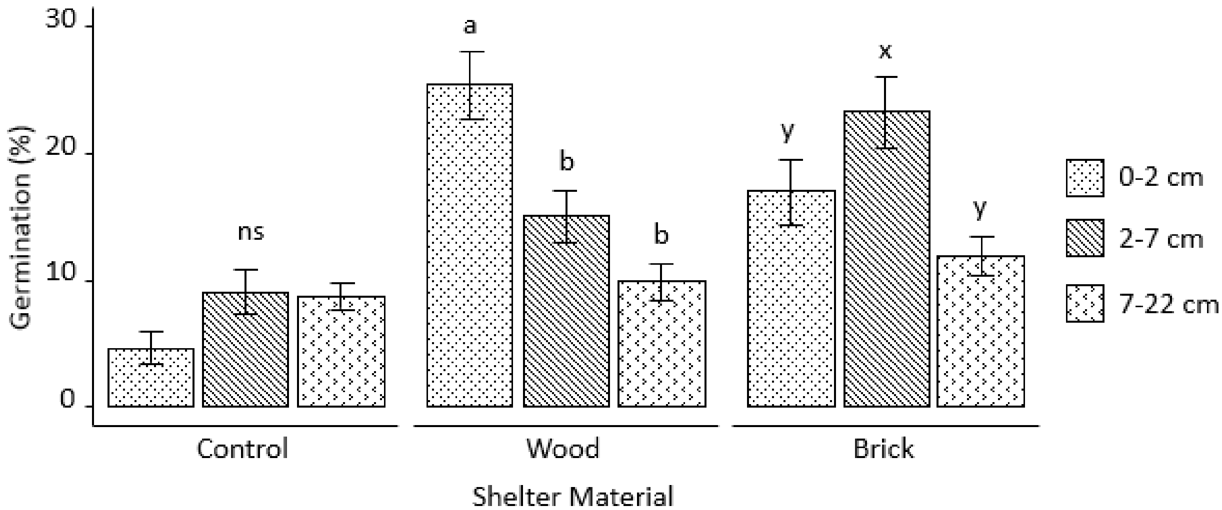


Figure 3-10: Means (and standard error) for responses of *V. americana* presented as (A) germination in response to aspect, (B) establishment in response to shelter type, and (C) response to aspect. Means with different letters indicate a significant difference between treatments. Note: ‘N’ represents North, ‘S’ represents South, ‘E’ represents East, ‘W’ represents West, ‘C’ represents no shelter material (control), ‘Wo’ represents wood shelter material, ‘B’ represents brick shelter material, and ‘ns’ represents not significant.

(A) *V. americana* Germination – Shelter x Distance



(B) *V. americana* Germination – Shelter x Distance

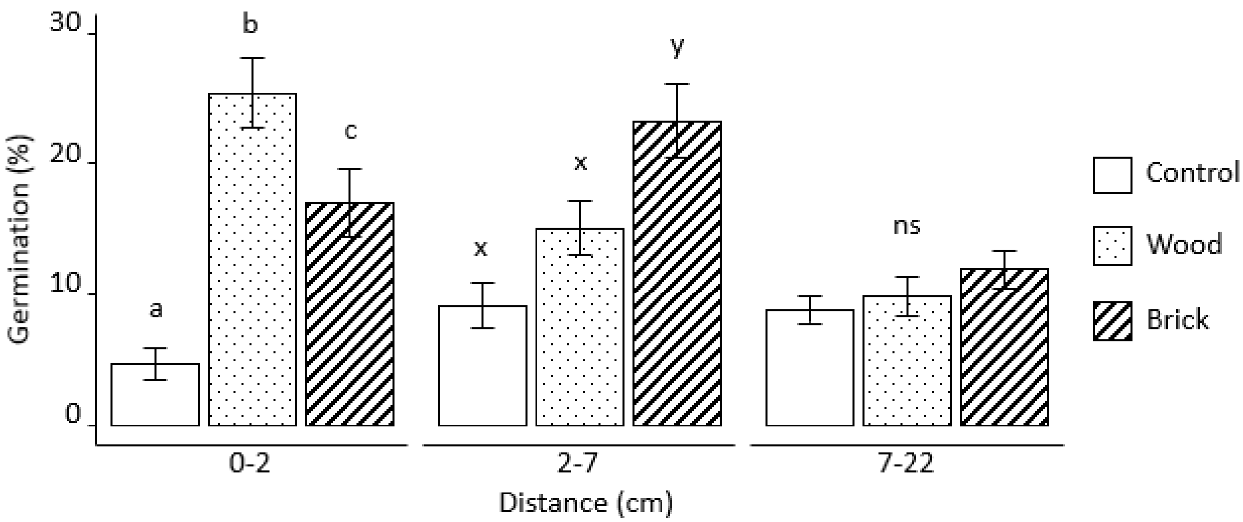
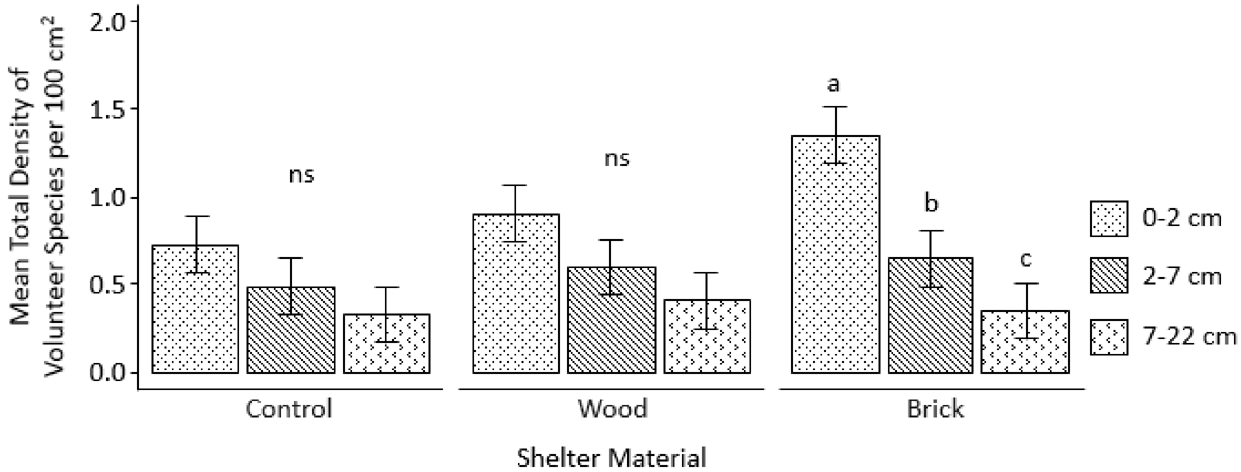


Figure 3-11: Means (and standard error) for germination of *V. americana* in response to shelter type and distance. (A) Means with different letters indicate a significant difference between the distance from shelter within a given shelter type; (B) Means with different letters indicate a significant difference between shelter types for a given distance. Note: ‘ns’ represents not significant.

(A) Total Density of Volunteer Species – Shelter x Distance



(B) Total Density of Volunteer Species – Shelter x Distance

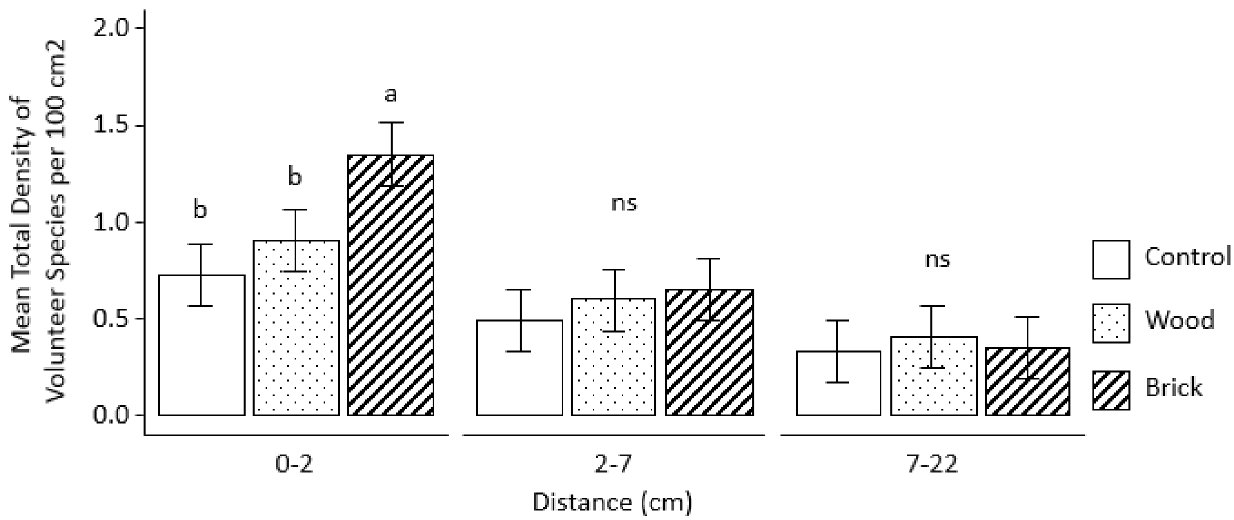
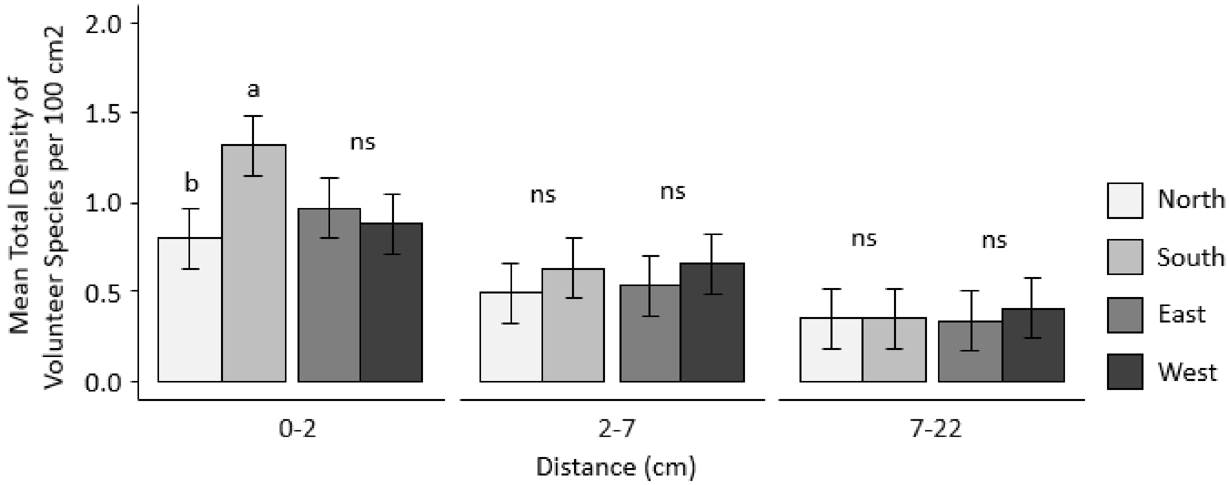


Figure 3-12: Least squared means (and standard error) for total density of volunteer species in response to shelter type and distance. (A) Means with different letters indicate a significant difference between the distance from shelter within a given shelter type; (B) Means with different letters indicate a significant difference between shelter types for a given distance. Note: 'ns' represents not significant.

(A) Total Density of Volunteer Species – Aspect x Distance



(B) Total Density of Volunteer Species – Aspect x Distance

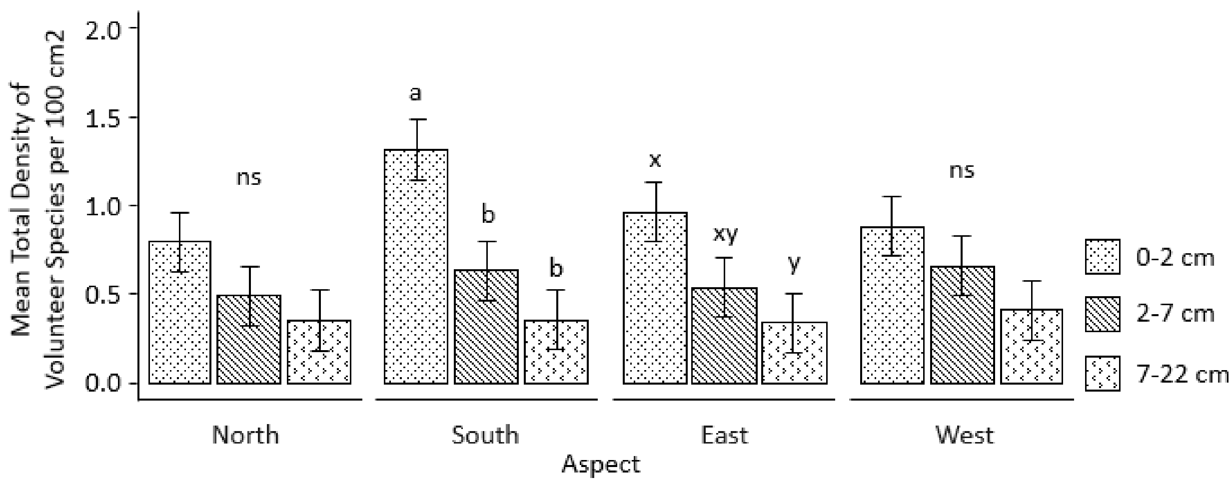


Figure 3-13: Least squared means (and standard error) for total density of volunteer species in response to aspect and distance. (A) Means with different letters indicate a significant difference between the two aspects (north vs. south or east vs. west) within a given distance; (B) Means with different letters indicate a significant difference between distances from shelter for a given aspect. Note: ‘ns’ represents not significant.

Chapter 4: General Discussion and Conclusions

4.1 Research Summary

On harsh reclamation sites with poor soil substrates, increasing the presence of vegetation - whether early successional or native species - is important for initial ecosystem recovery. The goals of my research were to: (1) examine reclamation practices involving planted aspen seedlings, biochar, and controlled-release fertilizer amendments on initial vegetation development; and (2) explore effects of shelter provided by coarse woody debris and inert material on the initial vegetation development at a young reclamation site with poor substrate material. In order to address the first goal, total cover, species richness and community composition of colonizing vegetation were measured in response to planted aspen seedlings, biochar, and fertilizer amendments to assess the success of these amendments on a harsh reclamation site. To address the second goal, the effect of shelter of two different materials, along with the effects of aspect and distance from shelter, on performance (survival) and growth of four native seeded species and volunteer vegetation were measured.

In the first study, I assessed the cover and richness of colonizing vegetation over two growing seasons. Additionally, I explored whether there was an additive effect of increasing fertilizer levels or a combined effect of biochar and fertilizer application. I predicted that planted aspen seedlings, biochar and fertilizer would increase cover, while increasing levels of fertilizer would reduce richness however differences would not be long lived. I also hypothesized that the application of both fertilizer and biochar would result in the highest cover and richness.

Results of this study suggest that planted aspen seedlings and a fertilizer amendment increased both cover and richness of colonizing vegetation in the first growing season, yet varying the amount of controlled-release fertilizer did not result in additional effects. Planted aspen seedlings resulted in a doubling of vegetation cover, and community composition was notably different than in areas with no seedlings. While the tree seedlings present on site were not very large and had not reached canopy closure, their presence may still have directly influenced the above- and below-ground growing conditions, although to a lesser extent than a mature forest. Additionally, richness increased which could also be attributed to the sheltering effect provided by the young seedlings (Bradshaw 1997, Hart and Chen 2006, Gignac and Dale 2007, Frouz et al. 2009, Mudrák et al. 2010). Fertilization at a rate of 50 kg N/ha resulted in higher vegetation cover and richness compared to the unfertilized control in the first growing season but this effect was short lived with no effect detected in the following growing season. The application of biochar at a rate of 1800 kg/ha did not influence the cover of colonizing vegetation but did result in decreased richness. It was expected that colonizing vegetation would benefit from the combined application of biochar and fertilizer amendments, as previous research on this site in 2012 found increased soil nitrate in areas with biochar and either high or low fertilizer treatments (Liu 2015). However, there were no synergistic effects of biochar and fertilizer found to influence the vegetation. It is suspected that the nitrogen was immobilized by herbaceous vegetation or microorganisms present in the soil, or was removed completely from the environment through leaching or surface runoff. Biochar may have also decreased nutrient availability, as the addition of biochar may cause nutrients to precipitate, converting them into unavailable forms through a change in soil pH or interactions with biochar-derived nutrients (Whitman et al. 2015). Additionally, biochar may increase the immobilization of mineral nutrients by microbes in the soil therefore increasing microbial biomass and reducing the available nutrients (Whitman et al. 2015).

In the second study the effects of shelter, its orientation and distance from shelter on germination, establishment (survival from the first to the second growing season), and growth (dry weight and height) of four seeded species (*B. ciliatus, D. purpurea*, *L. innovatus*, and *V. americana*) and on the density of volunteer species (any species other than what was sown) on a reclamation site with poor growing conditions were examined. Further, I explored whether the shelter effect was influenced by the type of shelter used (e.g. wood versus an inert stone material). I predicted that shelter provided from both wood and brick would increase germination, establishment, and growth (height and weight) of native species and density of volunteer plants, and further, that wood shelter would perform better than brick. Next, the northern aspect would provide greater shelter than southern aspects, and lastly effects of shelter would diminish with increasing distance from shelter, resulting in increased establishment and growth of seeded vegetation. Results suggested that the provision of shelter did generally improve performance of seeded species and density of volunteer species although effects differed somewhat among species and with aspect and distance from shelter. Within the short timeframe of this study there were few differences between CWD and brick, and no clear evidence that either type of shelter was preferable as brick sometimes had more favourable results and other times wood did; this indicates that shelter from the harsh conditions was the primary factor of these materials and not the potential products of decomposition. The orientation of the shelter was also important where the seeded species performed better on the north facing aspect; interestingly the volunteer vegetation had greater density on the south facing aspect of a shelter. Native seeded species generally performed best when 7-22 cm from the shelter, while volunteer vegetation preferred to be closer to the shelter. These results suggest that seeded native species are favoured by cooler and moisture climates whereas introduced volunteer species are favoured by warmer and drier climates, and that vegetation will vary in their response to aspect and distance to shelter depending on how they respond to microsite conditions such as shade, temperate and moisture differences (Nuberg 1998, González-Alday et al. 2008, Goldin and Hutchinson 2013, 2014, Goldin and Brookhouse 2015).

The results from this research complement current research surrounding forest reclamation in Alberta, while proving a unique perspective with regards to reclamation occurring within agricultural areas and involving poor substrate material. While these two studies are different, they both demonstrate how initial reclamation practices can influence the early trajectory of reclamation sites as related to the development of understory vegetation on different scales. As seen in Chapter 2, shelter provided by planted aspen trees increased presence of vegetation on site while in Chapter 3, the benefits of shelter on a microsite scale also increase performance and growth of vegetation. Increasing richness of native understory species on sites which may be limited by surrounding agricultural landscapes is important when restoring areas to forested landscapes. Initial community development can influence future site conditions (Norman et al. 2006, Grant and Koch 2007, Macdonald et al. 2015a) however this community is limited to propagules from the soil, dispersal from surrounding areas, and commercially available seeds. If soil conditions on the reclamation site are poor, establishment of vegetation is much more difficult, thus it is important to introduce shelters. Potential benefits from shelter include protection from wind, increased shade, increased capture of windblown seeds, and may act as germination sites thus increasing establishment and growth as a direct physical barrier and through indirect microsite effects (Carlsson and Callaghan 1991, Bird et al. 1992, Cleugh et al. 1998, Whisenant 2002, Buchman et al. 2007). Increasing heterogeneity across the landscape at various scales (including micro-topographic features such as hummocks, ridges, hollows; placing larger rocks, stones, or coarse woody material; or planting shrub and tree seedlings across the site or as windrows) can facilitate establishment of colonizing vegetation. Sheltered areas additionally impact soil properties and may ultimately create an area with higher probability of capturing native understory species on the site.

Forest restoration as part of mine reclamation has become important in recent years, as forests provide various ecosystem services as well as renewable and non-renewable resources. Reclaiming areas with poor substrate material back to forested landscapes presents a large challenge, as forests deserve good soil for development. This study attempted to establish tree seedlings and vegetation in an area with poor substrate material and used various amendments to aid both soil and vegetation development and establishment. The extremes in environmental and edaphic conditions during this study provided a unique situation where treatments were tested in the most challenging environmental conditions, and found that they were not as successful as hoped for. The results of this study show that some treatments may not be beneficial on sites with such poor quality substrate material, and research using other materials is necessary to find a solution for providing better substrate quality for forested landscapes on reclaimed mine sites.

4.2 Future Research and Limitations

Initial establishment of vegetation was found to be most strongly influenced by the planted aspen seedlings although herbaceous species composition consisted mainly of early successional, introduced species. I would suggest planting tree seedlings on site as soon as possible, as they provide important ectomycorrhizal fungal establishment in the soil (Hankin et al. 2015), capture wind-blown seeds, create shelter for emerging vegetation, and at high densities can reduce establishment and competition from weedy species (Macdonald et al.

2012). Planting aspen tree seedlings (with fast growth, drought tolerance, and vegetative reproduction characteristics (Macdonald et al. 2012)) will also help initiate above- and below-ground forest recovery. I also suggest seeding with native vegetation if available from commercial supplies, as this may increase richness in addition to natural revegetation in areas where native species dispersal is limited.

Continued research involving the use of biochar as an organic amendment over short and long-term time periods, as well as further exploring its use along with other amendments such as fertilizer, is needed. While biochar research has been gaining a lot of attention in the last decade for its use in agricultural settings and recently in land reclamation, an increased understanding of how biochar influences the physical, biological and chemical properties of soil is critical. Studies show that biochar can alter soil properties such as pH and bulk density, thereby improving soil nutrient availability (Warnock et al. 2010). The increased soil nutrient availability may result in increased plant biomass (Glaser et al. 2002, Steiner et al. 2007, Jeffery et al. 2011), yet nutrient cycling processes involving biochar are not fully understood (Deluca et al. 2015). Increasing knowledge in biochar-soil relationships will allow reclamation practitioners to appropriately prescribe biochar or combined biochar and fertilizer amendments on sites with poor quality soil, thus improving nutrient availability and in turn increasing plant establishment.

Currently, research surrounding fertilizer amendments on reclamation sites with forest floor or peat mineral capping materials is comprehensive (e.g. Rowland et al. 2009, Pinno et al. 2012, 2014, Grainger and Turkington 2013, Sloan and Jacobs 2013, Macdonald et al. 2015, Sloan et al. 2016). It is common practice to use a fertilizer amendment in order to replace nutrients that were lost in the soil during mining and reclamation practices. However, further research on various application rates and timing of application of controlled-release fertilizers on sites with a poor reclamation substrate is necessary. Application during the first growing season may have prolonged the release of nutrients however tree seedlings and vegetation did not show benefits from this in the second growing season. Using a wider range of application rates such as including 25 or 75 kg N/ha together in a study with 50 kg N/ha and 100 kg N/ha may lead to optimizing the application rates based on vegetation uptake and utilization on harsh reclamation sites. It is unknown if a lower rate of controlled release fertilizer would have benefited early vegetation development, as decreased fertilizer levels may have created a less ruderal, early successional vegetation community. On this research site with limiting soil conditions, the immediate application of fertilizer was likely necessary in order to supplement any initial vegetation growth, however on other less limiting sites, delaying application of fertilizer may allow for a greater amount of understory species to migrate and establish. Applying large amounts of fertilizer on sites with minimal vegetation cover may be unnecessary as there is limited vegetation to use the available fertilizer, creating the risk for increasing nitrophilous and weedy early successional species colonizing the site, or having the excess fertilizer leach from the environment.

In order to increase carbon and organic matter in the surface material, another amendment that could be used on this research site is compost. Increasing organic matter and soil organic carbon is important for soil stabilization, increasing water-holding capacity, and increasing below-ground heterogeneity (Larney and Angers 2012). As there are no longer any resources of organic matter remaining in the area surrounding the reclamation site, using compost from the City of Edmonton is a viable option for increasing the carbon and organic matter on the reclamation site. Fertilizing poor substrate material was not enough to re-establish suitable forest understory vegetation or tree seedlings at this site; increasing organic matter and soil organic carbon may have increased biological activity in the soil which in turn may have contributed to a greater above ground vegetation response.

The effects of shelter are expected to be mediated through effects on the microclimate. Further research involving data collection of the microclimate near shelter to test this would provide a nice complement to the results on performance and growth of vegetation. As no clear evidence was found from this research that either type of shelter material was superior, fine scale microclimate measurements at varying distances away from the shelter material could have provided a greater understanding of how microclimates are influenced with different shelter materials, and complemented the results on vegetation performance. Additionally, further research involving increased microsite heterogeneity on reclamation sites through placement of shelter material, specifically CWD and inert materials should be explored. This research demonstrated the importance of shelter on emerging vegetation and it potential in reclamation on poor sites. Long-term studies using CWD shelters on reclamation sites are necessary to further understand the impacts of CWD on nitrogen cycles, plant community and soil development. These long term assessments will provide information regarding successional paths on reclamation sites; studies involving other microsites such as inert materials and possibly tree seedling windrows may provide further insight on the effects of shelter, microsite and vegetation development.

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Appendices

Appendix A Tables

Table A1: Characteristics of the subsoil material found within the reclamation research area at the Whitewood coal mine in Alberta, Canada. Note: 'EC' represents electrical conductivity, 'SAR' represents sodium adsorption ratio, 'OC' represents organic carbon, 'Sat' represents base saturation, 'NO_3^-' represents nitrate, 'P' represents potassium, 'K^+' represents phosphate, and 'SO_4^-' represents sulphate. Data were obtained and used with permission from Navus Environmental Inc. (Navus) and TransAlta.

		(ds/m)		(%)	(%)	Available Nutrients (mg/kg)				
Substrate	pH	EC	SAR	OC	Sat	NO_3^-	P	K^+	SO_4^-	Texture
Unconsolidated geological material	6.8 to 7.2 (neutral)	0.66 to 2.08 (non-saline)	0.29 to 4.42 (non-sodic)	2 to 6	38 to 68	1.79 to 5.29	< 0.5 to 9.0	172 to 268	12 to 173	Clay loam to sandy clay loam

Table A 2: Plant taxa found in the study plots and the functional group they are part of. Codes are given for species that are shown in the ordinations (see Figure 2-6 and Figure 2-7); * indicates species listed as noxious weed in Alberta (Province of Alberta 2011). Nomenclature as per USDA (2015). Note: *Lepidium densiflorum* and *Stachys palustris* L. were only included in Forb analysis; 'F' represents Forb, 'G' represents Graminoid, 'I' represents Introduced, 'N' represents Native. 'X' indicates a species was present that year and '-' indicates that a species was present not present in that year.

Species Name	Year 2013	Year 2014	Code	Forb vs. Graminoid	Introduced vs. Native
Bassia scoparia (L.) A.J. Scott	X	X		F	I
Beckmannia syzigachne (Steud.) Fernald	X	-		G	N
Brassicaceae species	X	-		F	N
Bromus ciliatus L.	X	X		G	N
Calamagrostis canadensis (Michx.) P. Beauv.	-	X		G	N
Capsella bursa-pastoris (L.) Medik.	X	-		F	I
Chenopodium album L.	X	X		F	I
Cirsium arvense (L.) Scop.	X	X		F	I
Crepis tectorum L.	X	X	Creptec	F	I
Dalea purpurea Vent.	X	-		F	N
Descurainia sophia (L.) Webb ex Prantl	X	X		F	I
Elymus repens (L.) Gould	X	-		G	N
Elymus trachycaulus (Link) Gould ex Shinners subsp. *trachycaulus*	X	X	Elymtra	G	N
Equisetum arvense L.	X	X	Equiarv	F	N
Equisetum pratense Ehrh.	X	-		F	N
Erucastrum gallicum (Willd.) O.E. Schulz	X	X		F	I
Erysimum cheiranthoides L.	X	X	Erysche	F	I
Festuca idahoensis Elmer	X	-		G	N
Festuca saximontana Rydb.	X	X		G	N
Galeopsis tetrahit L.	X	X	Galetet	F	I
Geranium bicknellii Britton	X	-		F	N
Hordeum jubatum L.	X	X		G	N
Koeleria macrantha (Ledeb.) Schult.	X	X		G	N
Lactuca serriola L.	X	X		F	I
Lathyrus species	-	X		F	I
Lathyrus venosus Muhl. ex Willd.	X	-		F	N
Lepidium densiflorum Schrad.	X	-		F	No Data
Medicago sativa L.	-	X		F	I

Melilotus alba (L.) Lam.	X	X		F	I
Melilotus officinalis (L.) Lam	X	X		F	I
Melilotus sp.	X	X	Melispp.	F	I
Mentha species	-	X		F	I
Phalaris arundinacea L.	X	X	Phalaru	G	N
Phleum pratense L.	X	X		G	I
Plantago major L.	X	-		F	I
Poa palustris L.	-	X		G	N
Poa pratensis L.	X	X		G	I
Polygonum aviculare L.	X	X	Polyavi	F	I
Polygonum convolvulus L.	X	X		F	I
Polygonum lapathifolium L.	X	-		F	I
Populus balsamifera L.	X	X		Tree	N
Populus tremuloides Michx. natural regeneration	-	X		Tree	N
Potentilla norvegica L.	X	X		F	I
Salix species	X	X		Many	I
Salsola kali L.	X	X	Salskal	F	I
Sonchus species	X	X	Soncspp.	F	I
Stachys palustris L.	X	X	Stacpal	F	No Data
Tanacetum vulgare L.	X	X	Tanavul*	F	I
Taraxacum officinale F.H. Wigg.	X	X	Taraoff	F	I
Thlaspi arvense L.	X	-		F	I
Tragopogon dubius Scop.	-	X		F	I
Trifolium hybridum L.	X	X		F	I
Trifolium pratense L.	-	X		F	I
Trifolium repens L.	X	X		F	I
Trifolium species	X	X		F	I
Tripleurospermum perforatum (Mérat) M. Lainz	X	-	Tripper*	F	I
Vicia americana Muhl. ex Willd.	X	X	Viciame	F	N

Table A 3: Volunteer species found within the shelter treatment plots. Nomenclature and status in Alberta are as per USDA (2015).

Species	**Status in Alberta**
Bassia scoparia (L.) A.J. Scott	Introduced
Chenopodium album L.	Native and Introduced
Elymus trachycaulus (Link) Gould ex Shinners *subsp. trachycaulus*	Native
Erucastrum gallicum (Willd.) O.E. Schulz	Introduced
Melilotus Mill.	Introduced
Polygonum arenastrum Jord. ex Boreau	Introduced
Polygonum aviculare L.	Introduced
Populus balsamifera L.	Native
Populus tremuloides Michx.	Native
Salsola tragus L.	Introduced
Sonchus L.	Introduced

Appendix B Figures

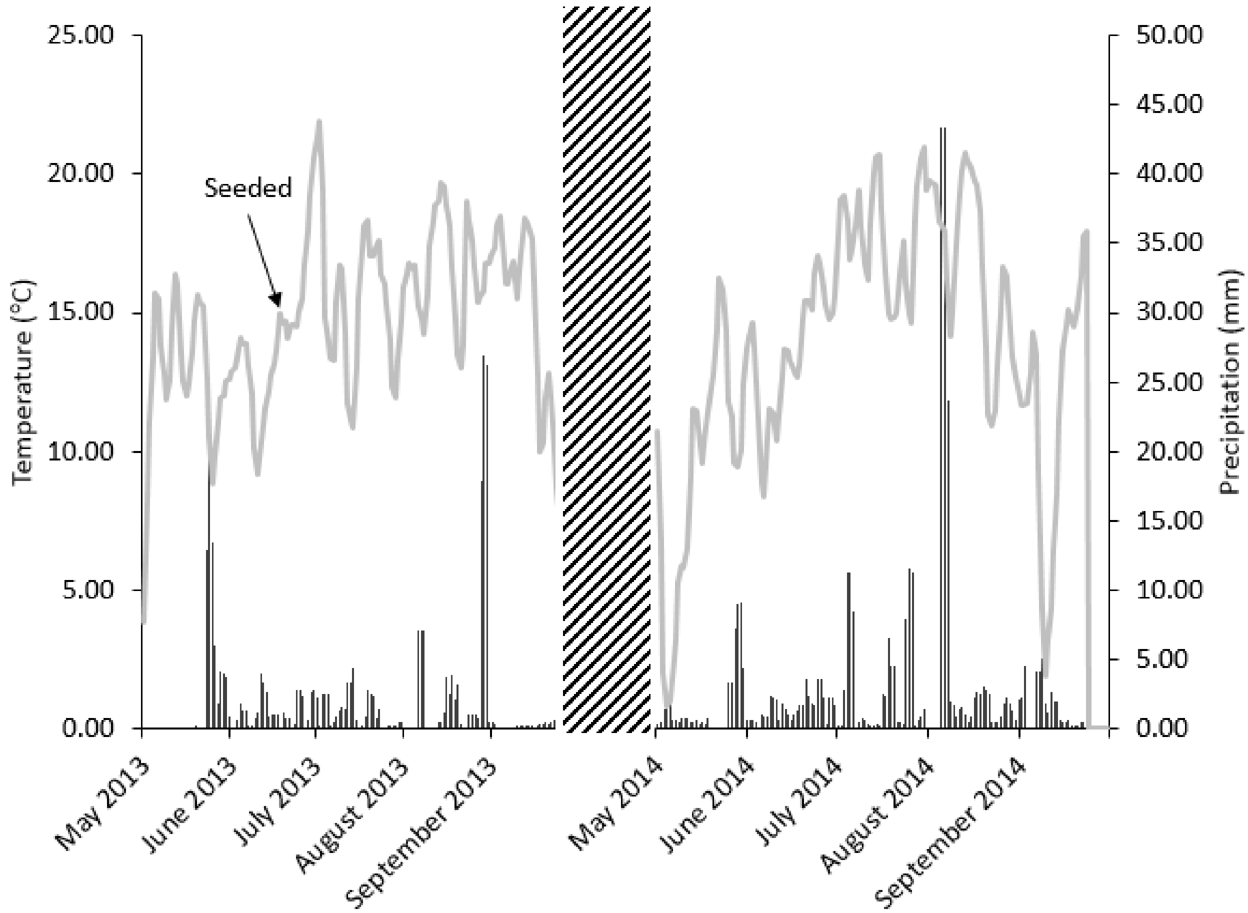

Figure B 1: Temperature and precipitation during the growing season (May 5 – September 30) in 2013 and 2014 based on data from a weather station located 30 km southwest of the mine in the township of Tomahawk, Alberta. Lines represent three-day mean temperature and bars represent the daily precipitation. Note: The arrow indicates when seeds were sown for the CWD experiment (June 24 2013 Chapter 3); hatched area represents dormant months (October 2013 – April 2014). Data accessed from Government of Canada (2016).

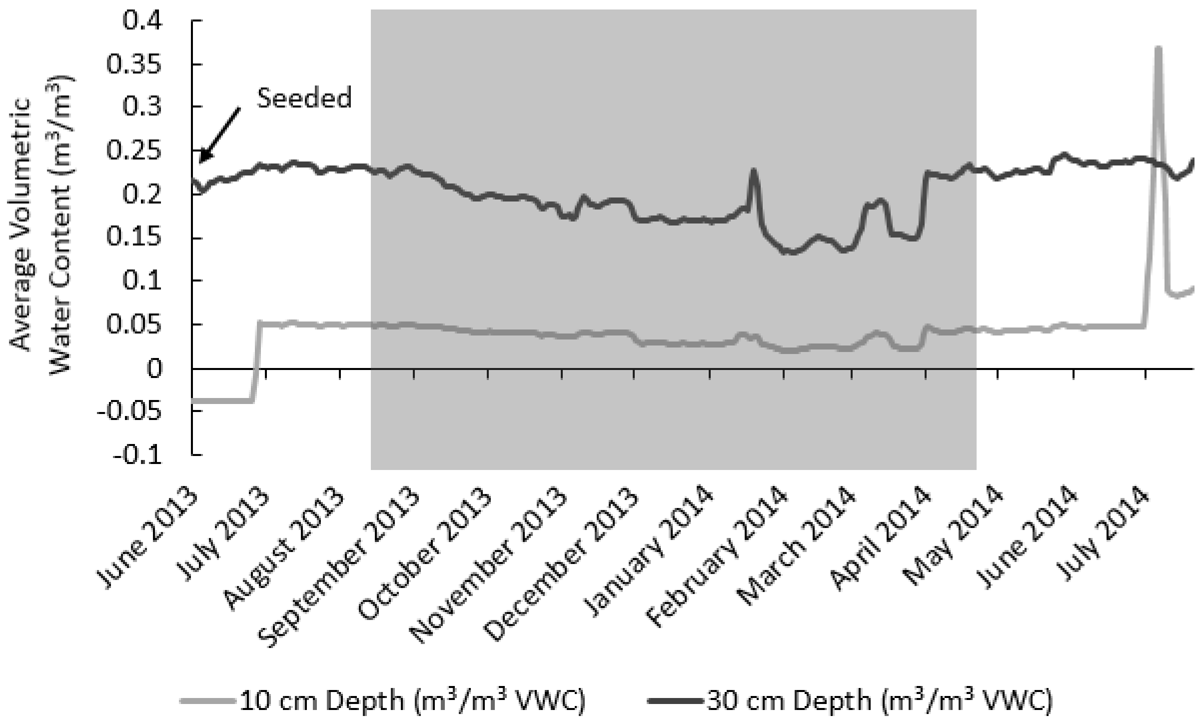

Figure B 2: Running three-day average volumetric soil water content measured using soil moisture and temperature sensors (5TM) and a EM50 datalogger at two depths (10 cm and 30 cm) in each of the six blocks in the reclamation research site at Whitewood coal mine in Wabamun, Alberta. The 10 cm depth placement is represented by the light grey line, while the 30 cm depth placement is represented by the dark grey line. Note: The arrow indicates when seeds were sown for the CWD (Chapter 3) experiment (June 24, 2013); shaded area represents dormant months (October 2013 – April 2014).

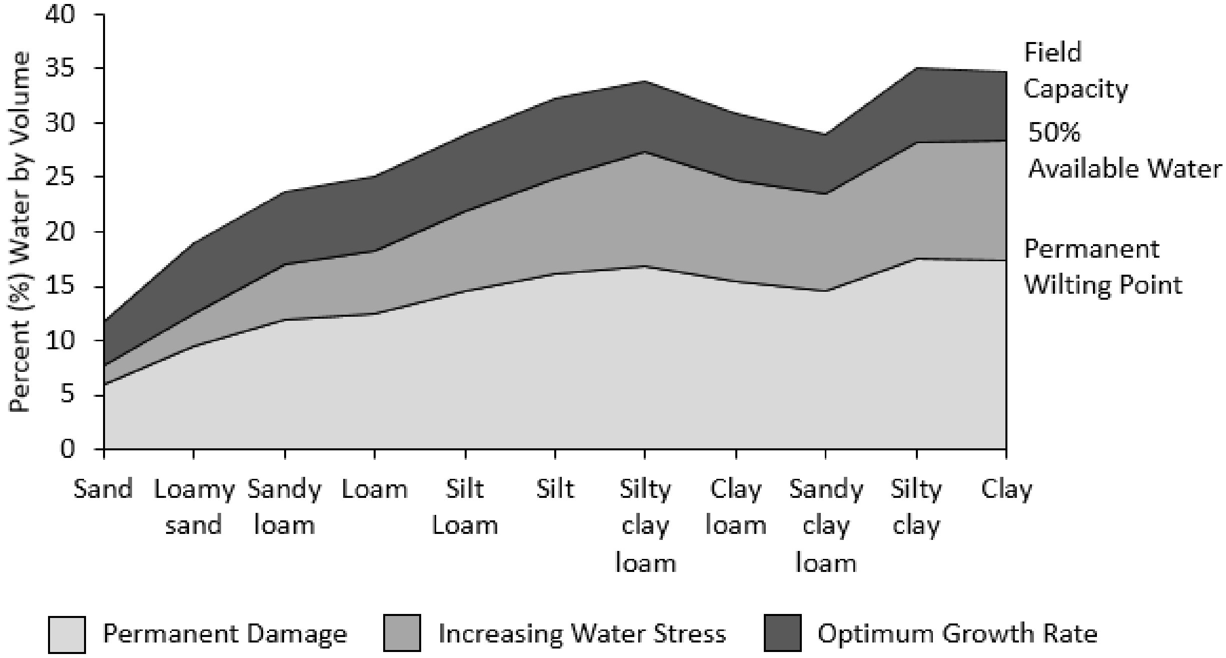

Figure B 3: Available soil moisture by soil texture expressed as percent water by volume including optimum growth range for major soil textures. Adapted from Shortt et al. (2011).