Quantifying the Influence of Soil Prescriptions on Ecosystem Processes in Reclaimed Forests of Varying Age in a Post-Oil Sands Landscape in the Athabasca Oil Sands Region, Alberta, Canada

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

The Athabasca Oil Sands Region (AOSR) in northern Alberta, Canada contains ~4800 km^2 available for surface mining, and as of 2017 ~767 km^2 had been disturbed for oil sands operations. The Alberta government requires this land to be reclaimed back to an equivalent capacity following the closure of mining operations. This includes the reclamation of upland forests, which serve vital ecosystem functions to the region. These functions are influenced by the cover soils used while reclaiming these ecosystems as they are designed to provide sufficient water and nutrients for the vegetation being planted. There are two different cover soils typically used in reclamation, peat mineral mix (PMM) and forest floor material (FFM), while there have been studies examining the differences between them some of the results are inconsistent. This research aims to further the understanding of how differences in cover soils used can influence the moisture and nutrient regimes of reclaimed forests, and how these processes change as vegetation develops.

Seven sites in the AOSR that varied in age, cover soil, and vegetation prescription were used for this study. Differences in soil physical properties were assessed and compared to changes in volumetric water content throughout the growing season to assess their impact on water regimes. Once the relationship between soil physical properties and water regimes were established the nutrient regimes of the sites were assessed through the in situ buried bag method. Similarly, to volumetric water content, nutrient mineralization rates were compared to soil physical properties to assess their impact on the nutrient regimes of the sites. Once the relationship between soil prescription and the water and nutrient regimes were established, how vegetation development can impact these processes could be determined.

Soil texture was found to be the dominant driver of water regimes at reclaimed sites, having a greater influence than topographical variables. This led to some sites being re-vegetated

incorrectly, which can lead to increased time for vegetation to become established and a potentially longer period before sites can become certified. Furthermore, the type of cover soil and mineral layer used were found to influence soil water regimes, with prescriptions using FFM having higher infiltration rates then PMM, while fine tailings sand mineral layers were more likely to result in water limited systems than overburden material.

In contrast the impact soil prescriptions used in reclamation had on nutrient regimes was much smaller then hypothesized. The lack of differences observed between FFM and PMM suggests that five years post-revegetation any initial benefits to the nutrient regimes of the soil will no longer be present. The only parameter that seemed to influence nutrient mineralization rates was silt content, where sites with a higher silt content typically had a slight increase in N, NH_4^+, and NO_3^- mineralization. In contrast, litter mineralization rates followed a similar trend to what would typically be observed in natural boreal forests, with broadleaf sites having higher P mineralization rates while NH_4^+ and N were unrelated to vegetation type.

These findings suggested that while soil physical properties have a significant influence on the water regimes of reclaimed sites, they have little impact on nutrient regimes five years post-revegetation. Instead vegetation inputs are the dominant control on nutrient availability. However, soil water regimes drive what vegetation can become established on reclaimed sites. Therefore, when attempting to predict the nutrient regimes of a site it is important to consider the impact soil properties will have on water regimes and how that may impact vegetation colonization, which will ultimately govern the nutrient mineralization rates.

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Chapter 1: Introduction

The Athabasca Oil Sands Region (AOSR) in northern Alberta, Canada contains ~4800 km^2 of land available for surface mining, of which by 2017 ~767 km^2 had been disturbed for oil sands development (Government of Alberta, 2019). The Alberta Government requires oil companies to restore this land back to an equivalent capacity (Government of Alberta, 2017), which will require the reconstruction of endemic ecosystems and landforms at the scale of whole landscapes (Johnson & Miyanishi, 2008). This includes the reconstruction of fen peatlands and boreal forests. In this region wetlands comprise 64% of the landscape, which are predominantly peatlands, while only 23% is comprised of forests (Rooney et al. 2012). Despite boreal forests taking up a smaller proportion of the landscape they will be essential in efforts to reclaim peatlands as upland forests play a key role on the hydrogeological functions of the landscape (Devito et al. 2005). Price et al. (2010) found that it will take a ratio of 3:1 forested uplands to peatlands to adequately supply water to support fen functions in this sub-humid climate, which will result in the conversion of land from a wetland dominated landscape to one dominated by forests (Rooney et al. 2012).

There has been significant research done on the reconstruction of boreal forests throughout the AOSR, however most of these studies are conducted on isolated land units, not taking the influence they have on hydrologic functions and biogeochemistry of the larger landscape into account (Rooney et al. 2012). Disturbed boreal forests can take 10-20 years for hydrological and biogeochemical functions to return to that of a natural system (Amiro et al. 2006; Goulden et al. 2011). However, this rate of recovery is dependent on the type of disturbance and whether it was left to recover naturally or was managed (Strilesky et al. 2017). Similarly, reclamation can be dependent on techniques used while constructing these ecosystems. There have been several studies that attempted to model requirements of forest reclamation to optimize the recovery of

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these systems (Carrera-Hernandez et al. 2012; Huang et al. 2015). These different techniques can result in the reclamation of different ecosites of the AOSR (Alberta Environment, 2010). While several studies have examined how reclaimed sites develop over time (Strilesky et al. 2017, Hahn and Quideau, 2013; Pinno and Hawkes, 2015; Rowland et al., 2009), more research is still needed to assess changes to hydrological and biogeochemical process as ecosystems develop.

1.1 Cover Soils and their Influence on Hydrological and Biogeochemical Processes

Before vegetation is planted on reclaimed sites, soils that are suitable for vegetation growth must be placed. Typically, reclaimed sites consist of a mineral substrate layer that has a cover soil layered overtop. These cover soils are designed to mitigate percolation into overburden waste, and provide adequate water for vegetation over dry summer periods (Carey, 2008; Meiers et al. 2011). Cover soils generally used are peat mineral mix (PMM) comprised of harvested lowland soils mixed with a mineral substrate, and forest-floor material (FFM), which consisting of harvested upland soils mixed with a mineral substrate (Mackenzie and Naeth, 2010). Mixing harvested upland and lowland soils with the mineral substrate improves tilth and reduces the loss of organic matter due to rapid decomposition (Mackenzie, 2011). Cover soils can differ considerably in their texture, bulk density, infiltration rates, porosity, specific yield, depths, and organic matter content (ex. Leatherdale et al, 2012; Huang et al. 2015; Ketcheson & Price, 2016). Which can have a significant impact on the moisture and nutrient regimes of reclaimed sites.

Previous studies have shown that PMM typically has lower infiltration rates and higher surface runoff than FFM when used to reclaim slopes, while FFM has been associated with higher SOM and greater vegetation development (Kwak et al. 2016; Mackenzie & Naeth, 2010; Leatherdale et al., 2012). Furthermore, soil texture has been shown to significantly impact soil water regimes by increasing available water holding capacity (AWHC), which has been associated

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with increased forest productivity (Haung et al. 2011). Paedogenic processes can further the impact soil texture may have on water regimes. For example, wetting and drying cycles can increase intra-aggregate bulk density in fine textured soils leading to the separation of pore space and a more continuous inter-aggregate pore network (Horn & Smucker, 2005; Pires et al. 2008). This process can foster the development of preferential flow paths impacting infiltration rates and increasing the development of soil organic matter (Raab et al. 2012; Zhang et al. 2018). Soil organic matter (SOM) can then increase the ability of a soil to retain water further impacting moisture regimes (Rawls et al. 2003).

Several studies have examined the impact different cover soils have on nutrient regimes of reclaimed sites (ex. Quideau et al., 2017; Howell et al., 2016; Jamro et al., 2014; Mackenzie and Quideau, 2012; McMillian et al. 2007; Gringras – Hill et al., 2018; Hahn and Quideau, 2013; Farnden et al., 2013; Kwak et al., 2016). However, conflicting results have been reported by studies examining the difference soil organic amendments (PMM or FFM) have on reclaimed sites (Quideau et al., 2017; Howell et al., 2015; Jamro et al., 2014; Mackenzie and Quideau, 2012; McMillian et al. 2007). Quideau et al. (2013) further support this by showing a disconnect between organic matter composition and nutrient availability of reclaimed soils. These findings indicate that more research is needed to assess how different soil properties influence nutrient regimes and whether soils are the dominant control in reclaimed forests.

1.2 Revegetation

After placement, soils are assessed to determine what vegetation is suitable given the soil characteristics. This is done following the guidelines of the Land Capability Classification System for Forested Ecosystems (LCCS). This system is based on soil nutrient regimes, moisture regimes and other physical and chemical properties that could limit vegetation growth. Soil moisture

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regimes are assessed based on the AWHC, which is determined by measuring the difference between the field capacity and wilting point in the mineral layer of the soils. The mineral layer of the soil is comprised of all soil with <17% total organic matter and is divided into three sections: topsoil (~0-20cm), upper subsoil (~20-50cm) and lower subsoil (~50-100cm). Reclamation material that contains soil with >17% TOC is the organic layer, which is used to determine the soil nutrient regime. Once the soil nutrient and moisture regimes are determined they can be used to assess potential ecosites for the reclaimed area, which will determine what vegetation is planted (Alberta Environment, 2010, 2006; Mackenzie, 2011). For example, an ecosite determined as type D may be planted with aspen (*Populus tremuloides)*, balsam poplar (*Populus balsamifera*), black spruce (*Picea mariana*), white spruce and an assortment of shrubs (e.g. dogwood (*Cornus stolonifera*). With potential ecosites determined, re-vegetation can take place following the Guidelines for Reclamation to Forest Vegetation in the AOSR (Alberta Environment, 2010).

1.3 Vegetation Impacts on Hydrological and Biogeochemical Processes

Once sites are revegetated, tree growth may cause changes to the water regimes of the site. Strilesky et al. (2017) showed that in the first ten years following reclamation tree growth has been linked to an increase in evapotranspiration rates. Additionally, increases in root growth can impact plants access to water and nutrients, which may further vegetation development (Bockstette et al. 2017). Thus, changes in water regimes overtime due to vegetation development may impact the successional trajectory of a forest and its ability to supply water for reclaimed fens. Continued research is needed to determine how vegetation development will further alter the ecohydrological interactions of reclaimed ecosystems, as well as establish the impact vegetation development has compared to the initial soil properties of these sites.

Vegetation has also been shown to have s significant influence on nutrient regimes of boreal forests. Studies have shown that P, C:N ratios and N concentrations are significantly higher in broadleaf forests compared to coniferous forests (Flanagan and Van Cleve, 1983; Prescott et al., 2000; Jerabkova et al., 2006). Furthermore, the chemistry of SOM has been directly linked to vegetation type of the forests (Quideau et al., 2001). However, studies done in oil sands reclamation have shown that reclaimed forests can differ in their biogeochemical processes considerably from natural forests, and there can be a disconnect between SOM and nutrient availability in these reconstructed ecosystems (McMillian et al., 2007; Rowland et al., 2009; Quideau et al., 2013). Thus, further research is needed to determine the influence vegetation may have on nutrient regimes of reclaimed ecosystems.

1.4 Summary

Forested uplands are an essential component of the landscape of the AOSR and as such must be reclaimed following the closure of mining operations. These forests will be essential in the reclamation of fen peatlands as an upland to fen ratio of 3:1 is required to meet the hydrological needs of the fen. Reconstructing theses ecosystems requires the placement of a capping layer in order to provide adequate water and nutrients for vegetation establishment. These cover soils can differ in the type of organic amendment, texture, bulk density, porosity, specific yield, organic matter, infiltration rates, and nutrient availability. As such it is necessary to properly characterize soils prior to revegetation to ensure suitable species are planted given the moisture and nutrient regimes of the site. Further, once sites are revegetated, moisture and nutrient regimes of the site may be altered through the development of roots, canopy and litter inputs. This study will look at several sites throughout the AOSR to determine how differences in soils physical properties have impacted the moisture and nutrient regimes of reclaimed sites. Furthermore, the influence of

vegetation on these processes will be assessed to determine whether soils or vegetation have a greater impact on the moisture and nutrient regimes of reclaimed sites, which will impact the successional pathway of the reclaimed system.

Chapter 2: Study Site

2.1 South Bison Hill

South Bison Hill, herein referred to as SBH_P_04, is located on a former overburden deposit at Syncrude Canada Ltd. approximately 40km north of Fort McMurray, Alberta (57° 39′ N, 111° 13′ W). Construction began in stages between 1980-1996 with reclamation capping layers placed on the slopes in 1999 and on the plateau in 2001. The site is ~200 ha in size and rises 60 m in elevation, with the plateau capped with ~20 cm of PMM and underlain with ~100 cm of reworked glacial till soil (Figure 2.1A). Three test covers of varying thickness were constructed on the north facing slope, one of which was used for this study. The test cover used was capped with ~20 cm of PMM and underlain with ~80 cm of reworked glacial till soil (Figure 2.1B). In the summer of 2002, the site was seeded to barley cultivar (*Hordeum* spp.) to prevent erosion of the soil covers and, in the summer/fall of SBH_P_04 it was planted with white spruce (*Picea glauca*) and aspen (*Populus* spp.).

2.2 Cell 11A

Cell 11 A, herein referred to as C_P_06, is located within the Millennium mine lease at Suncor Energy Inc. ~40km north of Fort McMurray, Alberta (56° 89′ 43″ N, 111° 38′ 05″ W). Construction took place in C_P_06, with the site being revegetated later that year. Its situated on a south facing slope with a 15% gradient (Figure 2.1A). The surface is capped with ~25 cm of loam textured PMM with a bulk density of 1190 g ml^{-1} (Figure 2.1B), which is underlain by a coarser layer of tailings sand material with a bulk density of 1490 g ml^{-1}. Revegetation consisted primarily of jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides)*, white birch (*Betula papytifera*) and an assortment of shrubs (e.g. blueberry (*Vaccinium* spp.)).

2.3 Nikanotee Fen Watershed

Three reclaimed slopes located in a constructed watershed (Nikanotee Fen watershed, Figure 2.1A) within the Millennium mine lease at Suncor Energy Inc. approximately 40 km north of Fort McMurray, Alberta, (56° 55′ 94″ N, 111° 25′ 04″ W) were used for this study. The oldest of these slopes, herein referred to as E_P_08, was constructed in 2007 and revegetated in 2008. The site is 8.1 ha in size and has a 19% slope, and a surface capped with ~40-50 cm of PMM underlain by ~100 cm of suitable overburden material (Figure 2.1B). Revegetation consisted primarily of white spruce, aspen, white birch, and an assortment of shrubs (e.g. green alder (*Alunus viridis*)). The two other hillslopes used in this study, the south east slope (SE_P_12) and west slope (W_P_12), were constructed in 2011 and revegetated in 2012. W_P_12 is 2.4 ha and has a 13% slope, while SE_P_12 is 8.4 ha. Both sites were constructed using the same ~40-50 cm of PMM underlain with ~100 cm of suitable overburden material (Figure 2.1B). W_P_12 was revegetated as a moist-rich site, primarily comprised of white spruce and aspen. In contrast SE_P_12 was planted as a dry site primarily comprised of jack pine.

2.4 Sandhill Fen Watershed

Two upland hills (hummocks) located in a constructed watershed (Sandhill Fen watershed, Figure 2.1A) within Base Mine at Syncrude Canada Ltd. approximately 40km north of Fort McMurray Alberta (57° 02′ N, 111° 35′ W) were used for this study. Both sites were constructed in 2011 and revegetated in 2012. Hummock 6 (H6_F_12) is 3.7 ha in size, the surface is caped with ~20 cm of FFM from a D ecosite source area and is underlain with ~30 cm of clay till mineral soil (Figure 2.1B). Revegetation consisted primarily of aspen, balsam poplar (*Populus balsamifera*), black spruce (*Picea mariana*), white spruce and an assortment of shrubs (e.g. dogwood (*Cornus stolonifera*). Hummock 7 (H7_F_12) rises 8 m in elevation and is ~3.5 ha in size, the surface is caped with ~15 cm of FFM from a A/B ecosite source area and underlain with ~40 cm of

Pleistocene fluvial sand (Figure 2.1B). Revegetation consisted primarily of white spruce, aspen, jack pine, and an assortment of shrubs (e.g. green alder).
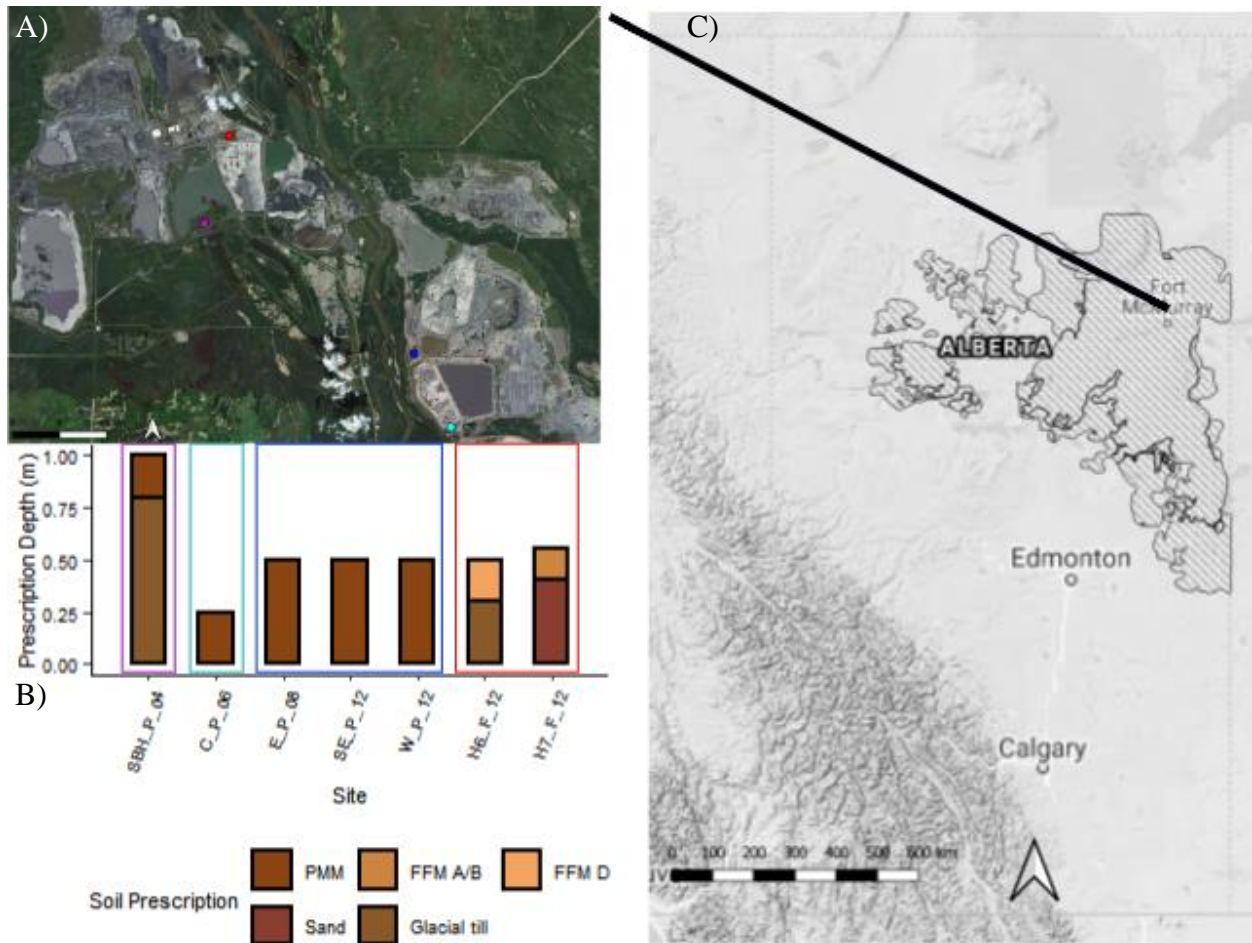
Figure 2.1: Map of study sites the AOSR. A) shows location of sites within the oilsands. B) shows prescription depths (m) for primary and secondary cover soils and the type of organic amendments used. C) shows where sites are located in Alberta, Canada.

Table 2.1: Site soil prescriptions for the primary cover soil types (PMM, FFM D, FFM A/B) and depths (m), secondary cover soil types (Glacial Till, Sand) and depths (m) and mineral substrate layer types (Overburden Material, Tailings Sand) .

Site	Primary Cover Soil		Secondary Cover Soil		Mineral Substrate Layer
	Type	Depth (m)	Type	Depth (m)	
SBH_P_04	PMM	0.2	Glacial Till	0.8	Overburden Material
C_P_06	PMM	0.25	N/A	N/A	Tailings Sand
E_P_08	PMM	0.5	N/A	N/A	Overburden Material
SE_P_12	PMM	0.5	N/A	N/A	Overburden Material
W_P_12	PMM	0.5	N/A	N/A	Overburden Material
H6_F_12	FFM D	0.2	Glacial Till	0.3	Tailings Sand
H7_F_12	FFM A/B	0.15	Sand	0.4	Tailings Sand

2.5 Climate

The climate of the Athabasca Oil Sands Region (AOSR) in northern Alberta is classified as sub-humid continental, characterised by long cold winters and short warm summers. The closest Environment Climate Change Canada (ECCC) weather station is located at the Fort McMurray Airport ~40 km south of the study sites. Thirty-year (1989 – 2018) growing season (May – August) averages indicate mean daily temperatures of +14.5 °C and total precipitation of 237.6 mm. During the 2018 growing season mean daily temperature and total precipitation were higher than the thirty-year average, with a mean daily temperature of +15.7 °C and total precipitation of 260.1 mm. However, average temperatures and total precipitation were not consistently higher throughout the growing season (Figure 2.1). In May and June higher mean temperatures were observed (+13.9 °C and +16.3 °C, respectfully) compared to the thirty-year averages (+10.0 °C and +14.9 °C, respectfully), while July and August had similar mean temperatures to the thirty-year averages. Total precipitation was higher in June and July for the 2018 growing season (114.2 mm and 116.9 mm, respectfully) compared to the thirty-year averages (75.4 mm and 77.6 mm,

respectfully) while May and August experienced lower precipitation (4.5 mm and 24.5 mm, respectfully) than the thirty-year averages (30.7 mm and 53.9 mm, respectfully).
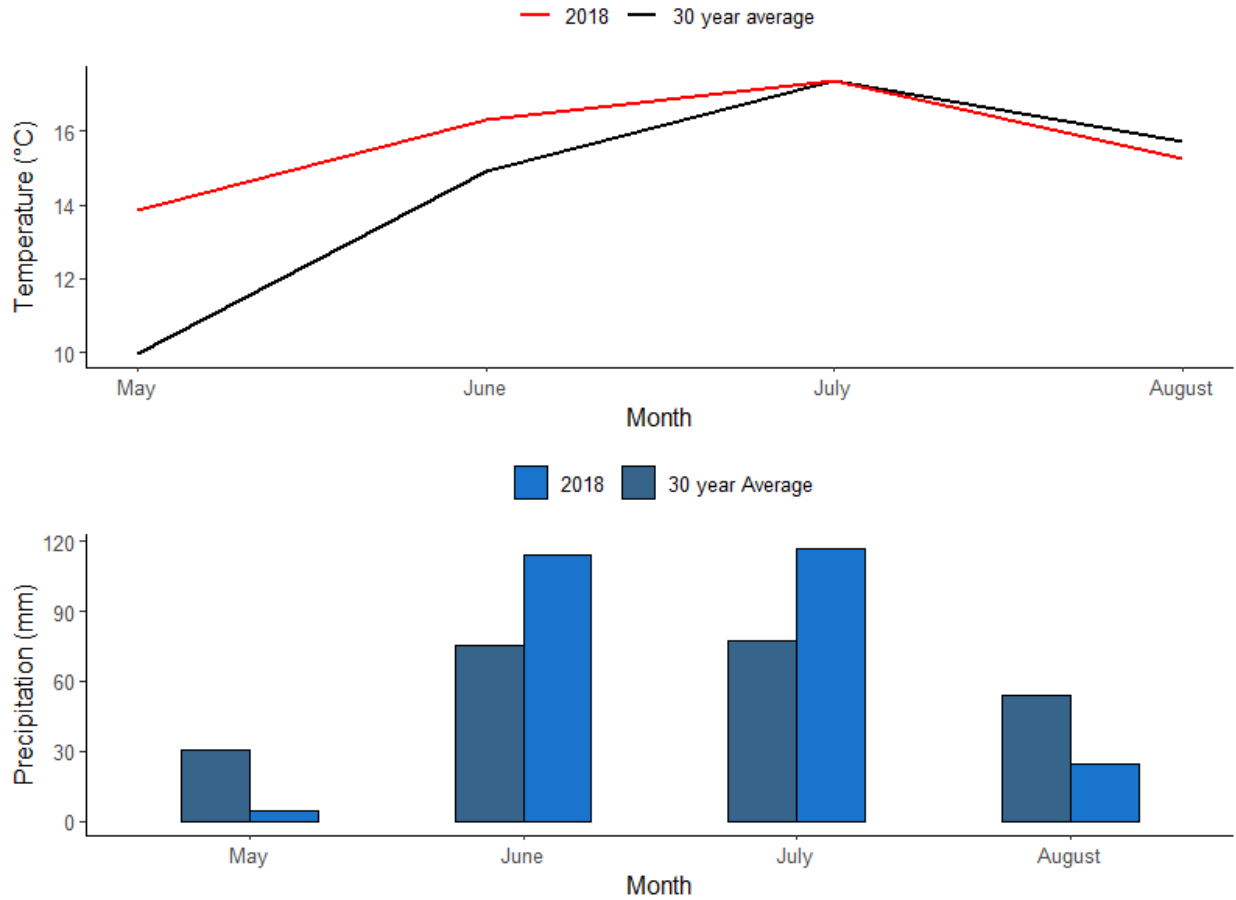
Figure 2.2: 2018 growing season temperature and precipitation compared to 30 years averages (Fort McMurray Airport, Environment and Climate Change Canada (ECCC) Weather Station).

Chapter 3: Quantifying the Effectiveness of Reclamation Cover Materials on Soil Water Regimes in a Post – Oilsands Landscape

3.1 Introduction

The Athabasca Oil Sands Region (AOSR) in northern Alberta, Canada contains ~4800 km^2 of land available for surface mining, and as of 2017 ~767 km^2 had been disturbed for oil sands operations (Government of Alberta, 2019). The Alberta Government requires this land to be reclaimed back to an equivalent capacity following the closure of mining operations (Government of Alberta 2017), which will require the reconstruction of ecosystems and landforms comprising the whole landscape (Johnson & Miyanishi, 2008). In the AOSR, this includes the reconstruction of fen peatlands and upland forests, which comprise 64% and 23% of the region (Rooney et al. 2012). Despite forests comprising a smaller proportion of the region they will be essential in efforts to reclaim peatlands after oilsands development as upland forests play a key role on the hydrological functions of the landscape and serve vital ecosystem functions to the region (Devito et al. 2005). For example, a ratio of approximately 3:1 of forested uplands to peatlands is required to adequately supply water to support fen functions in this sub-humid climate (Price et al. 2010; Rooney et al. 2012).

During forest reclamation, cover soils are placed over a mineral substrate layer to mitigate percolation into overburden and provide adequate water for vegetation during dry summer periods (Carey, 2008; Meiers et al. 2011). Two different types of cover soils are typically used for reclamation in the AOSR; peat mineral mix (PMM), which is salvaged lowlands organic soil mixed with mineral substrate, and forest floor material (FFM), which is salvaged upland boreal forest soil mixed with a mineral substrate (Mackenzie & Naeth, 2010). Whether PMM or FMM is used can have a significant impact on soils physical properties. Leatherdale et al. (2012) showed that PMM typically has lower infiltration rates and higher surface runoff than FFM when used to reclaim

slopes. While FFM has been associated with higher SOM and greater vegetation development (Kwak et al. 2016; Mackenzie & Naeth, 2010).

Cover soils can differ considerably in their texture, bulk density, infiltration rates, porosity, specific yield, depths, and organic matter content (ex. Leatherdale et al, 2012; Huang et al. 2015; Ketcheson & Price, 2016). These differences have been shown to have a significant impact on soil water regimes. Soil texture has been shown to increase soil available water holding capacity (AWHC), which can increase forest productivity (Haung et al. 2011). Furthermore, wetting and drying cycles can increase intra-aggregate bulk density in fine textured soils leading to the separation of pore space and a more continuous inter-aggregate pore network (Horn & Smucker, 2005; Pires et al. 2008). This process can foster the development of preferential flow paths impacting infiltration rates and increasing the development of soil organic matter (Raab et al. 2012; Zhang et al. 2018). Soil organic matter (SOM) can likewise impact soil water regimes by increasing the soils ability to retain water (Rawls et al. 2003).

As vegetation develops, increases in growth can further impact the water regimes of the site. Increases in tree growth have been shown to increase evapotranspiration rates in the first ten years following reclamation (Strilesky et al. 2017). Additionally, increases in root growth can impact plants access to water and nutrients, which may further vegetation development (Bockstette et al. 2017). Thus, changes in water regimes over time due to vegetation development and paedogenesis may impact the successional trajectory of a forest and its ability to supply water for reclaimed fens. Further research is needed then to determine what initial soil characteristics have the greatest influence on soil water regimes and how these regimes change over time while establishing the ecohydrological development pathway of the forest.

This study will look at several reclaimed sites in the AOSR that differ in their age since reclamation, the type of capping layer used (PMM or FFM) and resulting soil hydrophysical properties, and vegetation planted. These variables will be assessed and compared to the volumetric water content of the soil (VWC), AWHC and potential evapotranspiration rates (PET) to determine how differences in capping layers impact soil moisture regimes in reclaimed forests. Additionally, comparing results from young sites (5 years since revegetated) to older sites (≥ 8 years since revegetated) will allow for insight on how these systems change over time due to vegetation development and paedogenesis, and their interactions. These findings will permit the assessment of how soil physical properties and vegetation development may impact ecosystem reclamation success.

3.2 Materials and Methods

3.2.1 Meteorological Measurements

Meteorological stations were deployed at each site prior to the start of the 2018 growing season. Measurements of net radiation (NR-LITE & CNR1, Kipp and Zonen, Delft , Netherlands), ground heat flux (TCAV-L thermocouples & REBS HFT-3; Campbell Scientific Ltd., Logan, UT), air temperature (HMP45C; Vaisala, Oyj, Finland, & Hobo U23 Pro V2 datalogger; Onset Computer corporation, Bourne, MA), relative humidity (HMP45C; Vaisala, Oyj, Finland, & Hobo U23 Pro V2 data logger; Onset Computer Corporation, Bourne, MA) and precipitation (Hobo RG3-M data logger; Onset Computer Corporation, Bourne, MA) were recorded by data loggers (CR1000, CR3000, CR5000, & CRX23; Campbell Scientific Ltd, Logan, UT) and averaged for half-hour time intervals. Thermocouple wires and moisture probes (CS – 615 & CS – 650; Campbell Scientific Ltd., Logan, UT) were installed horizontally at varying depths in the cover soils to measure ground temperature and volumetric water content (VWC) respectfully (2 – 5cm depths & 10-15 cm depths). Thermocouple data from the primary capping layer was used to determine

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ground heat flux for sites without ground heat flux plates. Using data from the meteorological stations potential evapotranspiration (PET) was calculated using the Penman-Monteith method (Equation 1),

$$\lambda PET = \frac{\Delta(R_n - G) + p_a c_p \frac{(e_s - e_a)}{r_a}}{\Delta + \gamma} \tag{1}$$

where R_n is net radiation (MJ m^{-2}), G is ground heat flux (MJ m^{-2}), (e_s -e_a) is the vapour pressure deficit of air (kPa), p_a is the mean air density (kg m^{-3}), c_p is the specific heat of air (MJ kg^{-1} $^\circ$C^{-1}), Δ is the slope of the saturation vapour pressure temperature relationship (kPa $^\circ$C^{-1}), γ is the psychrometric constant (kPa $^\circ$C^{-1}), and r_a is the aerodynamic resistances (s m^{-1}), respectfully.

3.2.2 Soil Properties

Intact soil cores samples were collected using PVC pipe (10 cm diameter x 10 cm height) driven into the ground and wrapped in polyethylene film. Samples were transported back to the lab and analyzed for porosity, bulk density, and specific yield following standard methods (e.g. Freeze and Cherry 1979; Klute 1986). Organic matter (SOM) was calculated for a subset of samples using loss on ignition at 550°C for 3 hours. A second subset of samples were analyzed for texture by sieving the soil through a 2mm sieve and using a laser scattering particle size analyser (Horiba LA – 950V2) to measure particle size distribution. Infiltration rate (*f*) was measured at twelve points on the surface of each slope using a single-ring infiltrometer. Infiltrometers were installed at a minimum of 1cm depth and tests were conducted until a steady state was observed in order to account for antecedent moisture conditions (minimum five consecutive measurements within ± ~15%), the length of this process ranged from ~0.5 – 1.5 hours depending on the initial saturation of the soil (Ketcheson & Price, 2016).

Field capacity (θ_{fc}) and plant wilting point (θ_{pwp}) were derived from Equation 2, where Ψ_m is the matrix potential (J kg^{-1}), θ_s is the porosity, Ψ_c is the air entry potential (J kg^{-1}) derived from Equation 3, and b is a constant estimated from Equation 4. In Equations 3 & 4, d_g and σ_g are the geometric mean particle size (µm) and its standard deviation respectively. It is assumed that Ψ_m of θ_{fc} and θ_{pwp} is -33 J kg^{-1} and -1500 J kg^{-1} respectfully. These values provided the range for AWHC which is defined as the water held in the soil between field capacity and permanent wilting point.

$$\Psi_m = \Psi_c \left(\frac{\theta}{\theta_s}\right)^{-b} \tag{2}$$

$$\Psi_c = \frac{-5}{\sqrt{d_g}}(2\theta_s)^{-b} \tag{3}$$

$$b = \frac{10}{\sqrt{d_g}} + 0.2\sigma_g \tag{4}$$

3.2.3 Vegetation

Vegetation surveys were conducted during late July – early August 2018. Four transects extending from the bottom to top of each slope were established and three points along each were surveyed: at the bottom, middle and top of the slope. At each point the number of trees within a 5 m radius were counted with species and tree height recorded using a clinometer. From this data species diversity, abundance and frequency were determined. Additionally, fine root biomass was examined for each site following the sequential core method (Vogt & Persson, 1991). Three soil cores were collected at each site between the end of July and beginning of August. Cores were 30 cm in length and comprised of pvc pipe (10 cm diameter). Samples were soaked in water overnight, poured into buckets, and rubbed gently. Roots were then collected by pouring water through a sieve (0.2 mm). This was repeated until only rocks and organic debris were left in the soil. Live

roots were then separated from dead. Roots were considered live if they were pale in colour and free of decay and were considered dead if they were black or brown in colour and inflexible. Once roots were separated, they were oven dried for 24 hours at 70°C and weighed. Fine root biomass was calculated according the McClaugherty et al. (1982) as dry mass of living roots (gram) x 10^{-3} x 10^{8}/area of the core.

3.2.4 Statistical Analysis

All statistical analyses were performed with R (R Development Core Team, 2013). All data used for this paper were tested for normality using a Sharpiro-Wilk test. Data relating to soil hydrophysical properties and vegetation were found to be normal and as such were analyzed using a parametric test. However, meteorological data was found to be non-normally distributed, even with log-transformation being used, and was therefore analyzed using non-parametric tests. To analyze spatial differences in soil physical properties an ANOVA test was done using the function *aov* and was considered significant if $p < 0.05$. This was followed by a Tukey HSD post-hoc analysis to examine which sites shared similar soil characteristics. For the meteorological data a Kruskal Wallis ANOVA was performed using the *aov* function to examine spatial differences in the overall PET and VWC levels. A Spearman correlation analysis was then performed using the *cor* function to examine how fluctuations in VWC and PET differed among sites. Both Kruskal Wallis ANOVA and Spearman were considered significant if $p < 0.05$.

3.3 Results

3.3.1 Soil Physical Properties

The finest soil texture was in the SBH_P_04 sites, consisting of a clayey-loam (Figure 3.1). This was followed by four sites (E_P_08, SE_P_12, W_P_12 and H6_F_12) that were characterized by loam soils, of which H6_F_12 had the highest clay content and W_P_12 the highest sand. The coarsest textured sites were H7_F_12 and C_P_06, which were comprised of loamy-sand and

sandy-loam, respectfully. The SBH_P_04 site had a bulk density that was significantly lower than all other sites (0.65 g ml^{-1}, p < 0.05). Bulk density and specific yield were statistically similar for all other sites (Figure 3.2). When examining porosity, SOM and f greater differences were observed. Porosity was highest in SBH_P_04 (0.48), C_P_06 (0.46), and H7_F_12 (0.48), two of which are characterised by coarse textured soils. SE_P_12 had the lowest porosity (0.37), however it was not significantly different from W_P_12 (0.39, p = 0.99), which was constructed at the same time using the same prescription of PMM. SOM was highest in SBH_P_04 (0.31), and lowest in H7_F_12 and C_P_06 (0.07 and 0.13 respectively). Infiltration rates were highest for SBH_P_04 (1089 mm hr^{-1}) and lowest for SE_P_12 (142 mm hr^{-1}). However, an examination of the p-values shows the f for SE_P_12 is almost identical to that of W_P_12 (239 mm hr^{-1}, p = 0.99).
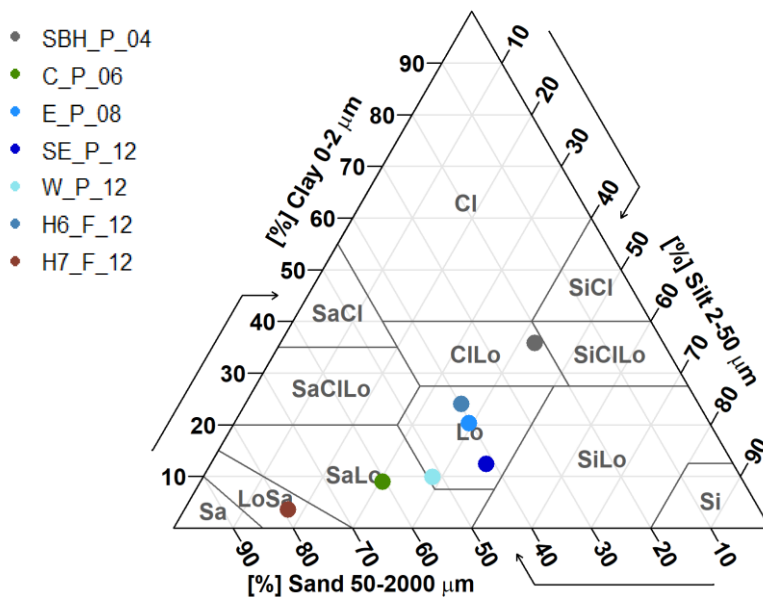
Figure 3.1: USDS soil texture plot for all sites, where the axes indicate % clay, silt and sand. Points represent the mean particle size for sites based on 10 cm soil cores (n = 12).
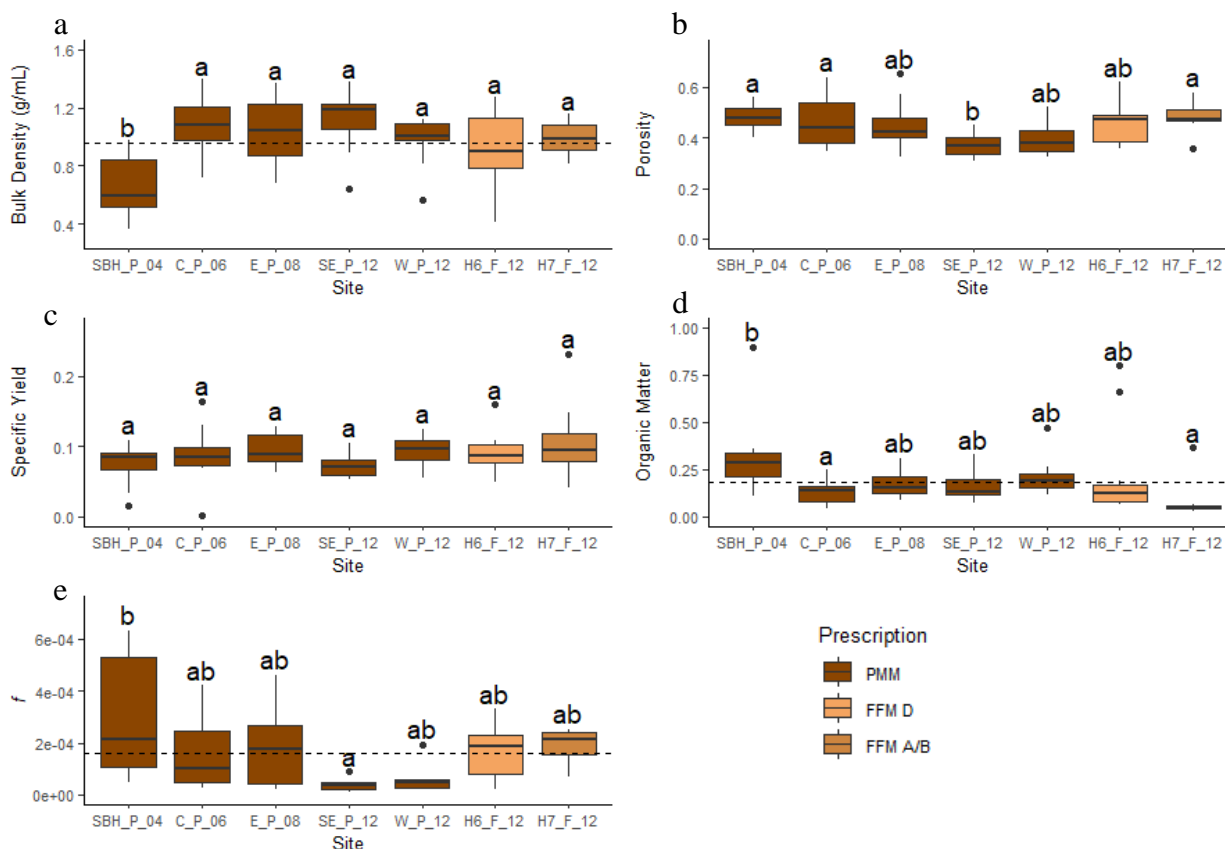
Figure 3.2: Box plots of soil bulk density (a), porosity (b), specific yield (c), organic matter (d) and infiltration (e). Horizontal line represents the mean for all sites. Tukey HSD results shown above each plot. Colours represent soil prescription used (PMM, FFM – A/B, FFM-D).

3.3.2 Soil Water Regimes

VWC was significantly different for all sites (p <0.05), with coarse textured sites (C_P_06 & H7_F_12) having consistently lower VWC than fine textured sites reclaimed at similar times (SBH_P_04 & H6_F_12) (Figure 3.3). Furthermore, differences in VWC rates in sites that all consisted of loam (E_P_08, SE_P_12, W_P_12, and H6_F_12) were much less extreme than between H6_F_12 and H7_F_12. Similarly, AWHC was significantly lower in sites with coarser textured soils (Table 3.1). Older sites had higher AWHC than younger sites that consisted of similarly textured soils, apart from SE_P_12 (0.13). The upper and lower limits to AWHC (θ_{FC} and $\theta_{PWP,}$ respectively) were also higher in finer textured soils and at older sites, aside from H6_F_12 (0.27 and 0.15 respectively). Additionally, it was observed that older sites (SBH_P_04,

C_P_06, E_P_08) had more days when shallower moisture contents fell below θ_{PWP}, apart from SE_P_12. The deeper values at these sites also fell below θ_{FC} more consistently than the younger sites, with some younger sites having values above θ_{FC} everyday of the growing seasons (W_P_12, H7_F_12). The E_P_08 site was the only site to have its deeper VWC fall below θ_{PWP} for the majority of the growing season.
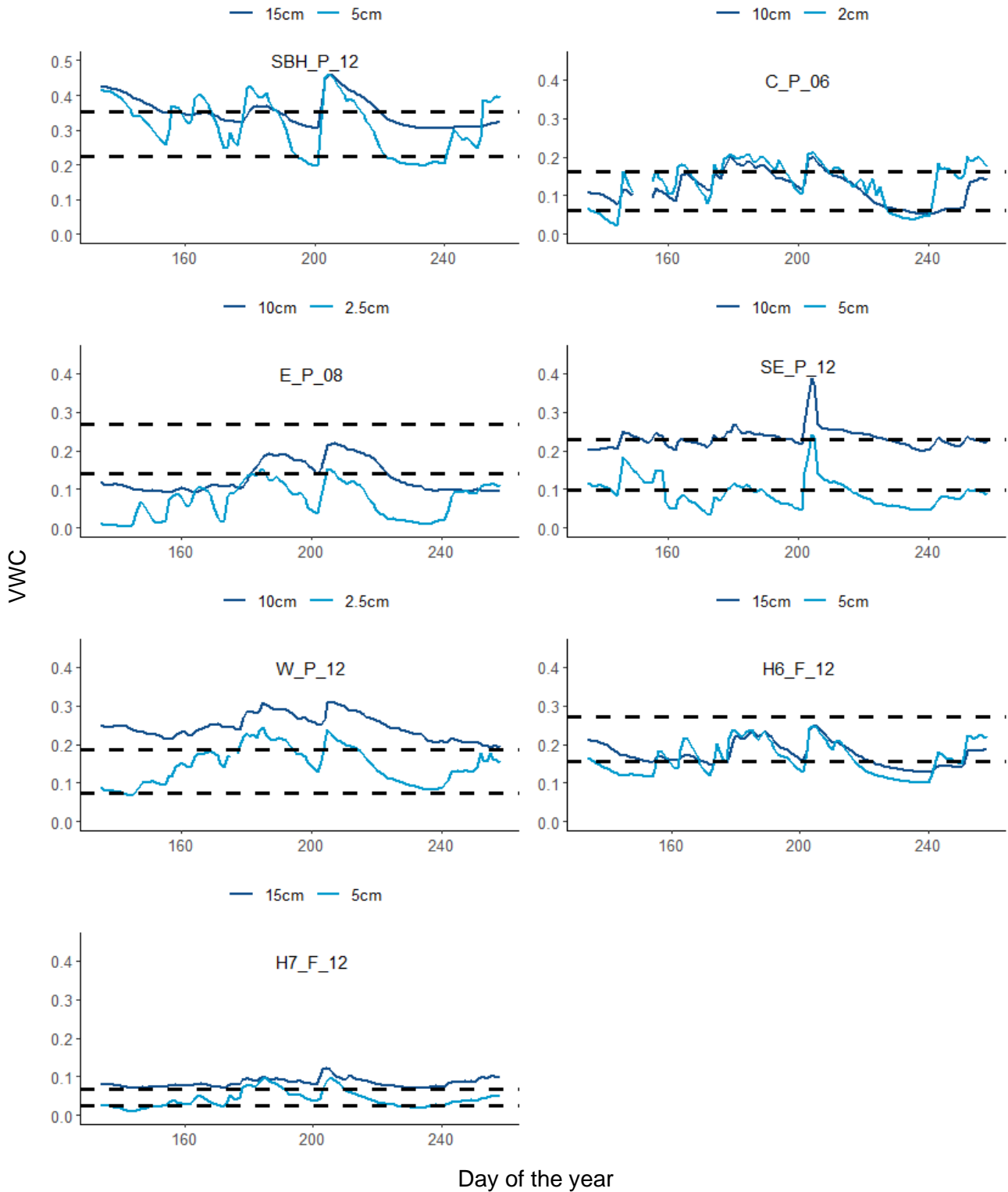
Figure 3.3: Growing season volumetric water content (%) for all study sites. Horizontal lines represent the upper and lower limits of AWHC (θ_{FC} and θ_{PWP}).

Table 3.1: Soil hydrophysical properties. Where θ_{FC} is the field capacity, θ_{PWP} is the plant wilting point, and AWHC is the soil available water holding capacity.

Site	θ_{FC}	θ_{PWP}	AWHC
SBH_P_04	0.35	0.22	0.13
C_P_06	0.16	0.06	0.10
E_P_08	0.27	0.15	0.13
SE_P_12	0.23	0.10	0.13
W_P_12	0.18	0.07	0.11
H6_F_12	0.27	0.15	0.12
H7_F_12	0.07	0.02	0.04

Highest daily PET was observed at C_P_06 (2.94 mm/day), along with the highest total PET for the growing season (362 mm). PET rates at this site were significantly higher than all other sites ($p < 0.05$) with the exception of H6_F_12. This was likely due to C_P_06 having a lower ground heat flux then sites with similar net radiation, temperature and relative humidity (0.0076 MJ m^{-2} 30min^{-1} vs >0.02 MJ m^{-2} 30min^{-1}). PET was also significantly lower at SBH_P_04 and W_P_12 compared to H6_F_12 ($p = 0.0002$ and $p = 0.0001$ respectfully), again likely due to its low ground heat flux compared to sites with similar net radiation (0.0050 MJ m^{-2} 30min^{-1}). Total PET for all sites ranged between 249 and 362 mm per day with W_P_12 experiencing the least and the C_P_06 experiencing the most. The only sites where PET exceed precipitation were C_P_06, E_P_08, SE_P_12, which are in close proximity to one another (Table 3.2). However, PET was only ~53 mm greater in SE_P_12 than W_P_12. Fluctuations in PET throughout the growing season did not show any significant differences ($p > 0.05$). From day 202 – 205 all sites showed low PET rates likely due to increased cloud cover over those days resulting in low net radiation (~0.05 MJ m^{-2} 30min^{-1})
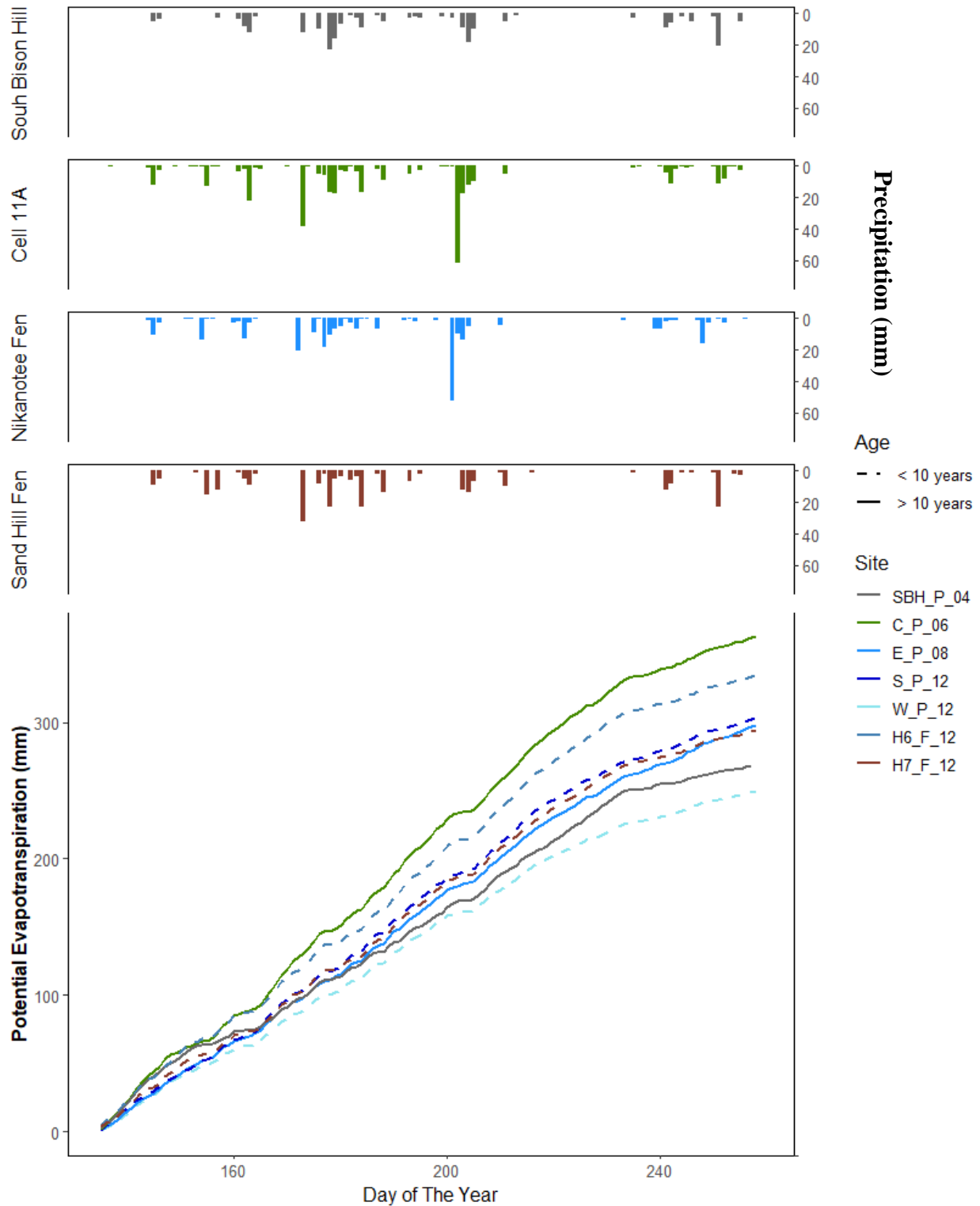
Figure 3.4: Cumulative total daily PET (bottom) throughout the 2018 growing season for all sites, based on the Penman-Montieth equation. Daily precipitation (top) for all sites throughout the 2018 growing season.

Table 3.2: Microclimate parameters during the 2018 growing season at all sites.

Site	Total PET (mm)	Total Precipitation (mm)	Average RH (%)	Average Air Temperature (°C)	Max Air Temperature (°C)	Min Air Temperature (°C)
SBH_P_04	254	328	70	15	37	-6.14
C_P_06	362	346	64	16	42	-6.35
E_P_08	297	273	63	16	36	-4.8
SE_P_12	302	273	61	18	43	-4.5
W_P_12	249	273	63	18	43	-4.5
H6_F_12	333	358	65	16	35	-2.5
H7_F_12	293	378	63	16	36	-2.3

3.3.3 Vegetation Development

Tree growth was highest in the SBH_P_04 site, with trees averaging heights >10 metres (Table 3.3). Further, SBH_P_04 had the tallest tree measured (18.7 m) and the highest FRB (1196 kg ha^{-1}). In the 2012 sites ones reclaimed using FFM (H6_F_12, H7_F_12) had the greatest, tree growth (1.5 m, 1.4 m), density (5613 stems ha^{-1}, 4138 stems ha^{-1}) and FRB 483 kg ha^{-1}, 276 kg ha^{-1}). Overall, sites dominated by broadleaf species showed greater growth, density and FRB than coniferous sites, and even in coniferous dominated sites the tallest trees were broadleaf apart from the C_P_06 sites, which only had coniferous species present.

Species diversity had often changed considerably from what was initially planted at the sites. C_P_06 was planted with predominantly jack pine with some broadleaf species (white birch) but is now solely dominated by jack pine (Figure 3.5). Furthermore, SE_P_12 and W_P_12 were initially planted as *ecosite-d* and *ecosite-a/b,* respectfully. However, tree surveys done in 2018 showed both sites having similar species composition despite being planted differently initially.

Table 3.3: Mean and maximum tree height, density and fine root biomass (FRB) measurements from the 2018 growing season

Site	Mean Tree Height (m)	Max Tree Height (m)	Density (stems ha^{-1})	FRB (kg ha^{-1})
SBH_P_04	13.6	18.7 (Aspen)	6215	1196
C_P_06	3.4	3.6 (Jack Pine)	2560	230
E_P_08	1.6	10.7 (Aspen)	14565	535
SE_P_12	1.0	3.6 (Poplar)	1167	104
W_P_12	1.1	2.6 (Aspen)	1984	123
H6_F_12	1.5	3.3 (Poplar)	5613	483
H7_F_12	1.4	4.0 (White Birch)	4138	276
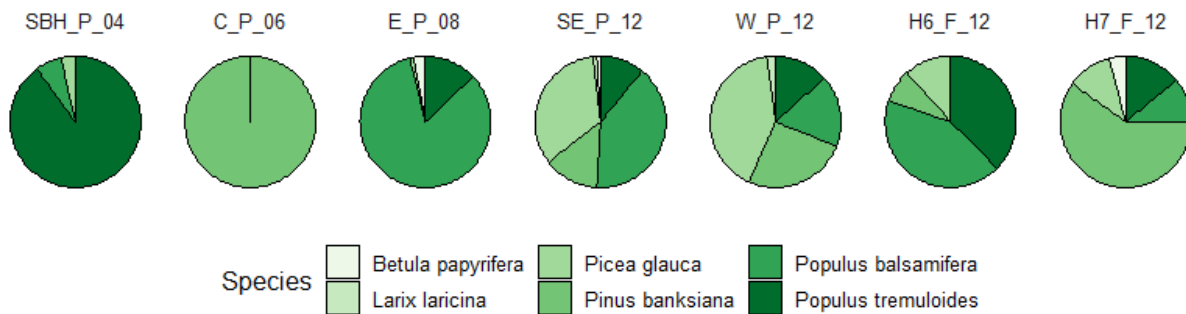
Figure 3.5: Species diversity at all study sites based on surveys conducted during the 2018 growing season.

3.4 Discussion

3.4.1 Reclamation Prescriptions Impact on Water Regimes

Soil texture played a predominant role on the moisture regimes of all sites. Finer textured soils experienced consistently greater VWC and AWHC regardless of the aspect or slope of the sites. This suggests that soil physical properties may play a greater role in determining moisture regimes than slope or aspect (Devito et al. 2005; Leatherdale et al. 2012; Jung et al. 2014; Gringras - Hill et al. 2018). However, further research is needed to assess the influence of topography on moisture

regimes of reclaimed sites. Furthermore, soil texture was found to influence the distribution of fine roots with sites that have coarser textured soils having a smaller FRB than sites with more fine textured soils. Jung et al (2014), similarly found soil texture had a major impact on water availability, fine root distribution and nutrient availability.

In addition to soil texture, site age had a significant impact on water regimes of reclaimed soils. Older sites were shown to have higher porosity and infiltration rates than younger sites with similar initial soil characteristics (Hussein and Adey, 1998, Pires et al, E_P_08; Adeli et al. 2019). However, it is important to note that these sites were characterized by finer textured soils, which can lead to an increase in intra-aggregate bulk density and a separation of pore space. Thus, if sites had been comprised of coarser textured soils the differences may have been less significant (Horn & Smucker, 2005). This is can be seen when comparing data from C_P_06 and E_P_08. Although E_P_08 was the younger site, its infiltration rates are similar to C_P_06, which is likely due to its high clay content resulting in a separation of pore space as sites age effectively increasing porosity and infiltration rates (Hussein and Adey, 1988; Horn & Smucker, 2005; Pires et al. 2008; Wu et al. 2017; Adeli et al. 2019). Similarly, Ketcheson & Price (2016) showed infiltration rates in 2014 for the 2007 (E_P_08) and 2011 (W_P_12) sites were 195 mm/hr and 35 mm/hr, respectively, while surveys from 2018 showed 660 mm/hr and 240 mm/hr, respectively. However, increases in infiltrations rates may diminish over time as the E_P_08 site only increased by ~3 times the amount in four years, while W_P_12 increased by ~7 times the amount in four years. Furthermore, increases in infiltrations rates may only be present at shallower depths as previous research has shown that greater weathering occurs in the top portions of soil reducing petroleum hydrocarbons and increasing infiltration rates (Neil and Si, 2018; Neil and Si, 2019).

Infiltration rates, porosity, and SOM were also higher in FFM than PMM sites of the same age. Additionally, the fine textured FFM site (H6_F_12) AWHC was higher than in the similarly textured PMM sites of the same age (SE_P_12, W_P_12), which may be due to its higher clay content (Leatherdale et al. 2012; Jung et al. 2014). It was also the only one of the three sites where VWC in either moisture probe (5 cm & 15 cm) never reached θ_{FC}. This is likely due to H6_F_12 being underlain with tailings sand instead of overburden material, which is prone to low VWC (Naeth et al. 2011; Duan et al. 2015). Further, less variability in fluctuations of VWC at H7_F_12 may be due to the high SOM, which can increase the soils ability to retain water (Rawls et al. 2003). However, it may also be due to layering of coarse textured soils, which has been shown to increase field capacity beyond what would be estimated based on average soil textures (Zettl et al., 2011).

3.4.2 Water Regimes Impact on Vegetation Development

Five years after planting, SE_P_12 and W_P_12 have similar species diversity with SE_P_12 having a greater abundance of broadleaf species and W_P_12 a greater abundance of conifers, despite originally being planted as ecosites a/b and d, respectively (Daly et al. 2012). This change in diversity from what was originally planted is likely due to SE_P_12 having a greater slope for which it was categorized as a dry ecosite (Alberta Environment, 2006; Daly et al. 2012). However, its higher AWHC is resulting in more broadleaf species and a causing a change from ecosite a/b to ecosite d (Gringras – Hill et al. 2018; Pinno and Hawks, 2015). In the older sites (age $\geq$ 10 years) there was a decrease in species richness with sites becoming dominated by either coniferous or broadleaf species, depending on there AWHC. This was inline with previous research showing that site can take up to 20 years to stabilize following reclamation (Peltzer et al. 2000; Hunt et al. 2003).

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Although VWC and AWHC have a strong influence on vegetation establishment (Gringras – Hill 2018), studies have shown that it often is not the limiting factor for growth with nutrient availability having a greater impact (Kwak et al. 2016; Mackenzie & Naeth, 2010). This may be the reason for greater growth at FFM sites than PMM sites reclaimed the same year, as previous studies have shown the type of cover soil can have a significant impact on nutrient regimes of reclaimed sites (Jamro et al., 2014; Kwak et al. 2016). Furthermore, at the PMM sites despite SE_P_12 having higher AWHC it had lower mean tree height, density and FRB than W_P_12 . This could be due to the broadleaf species being planted at the W_P_12 initially allowing them longer to grow over the broadleaf trees at SE_P_12, as PMM can take longer for non-planted species to become established (Gringras – Hill et al. 2018).

At the drier FFM site (H7_F_12) there was lower mean tree height, density and FRB than at H6_F_12. There are several reasons this may be occurring, the first is that deciduous sites typically have greater nitrogen availability and are fast growing (Jerabkova et al., 2006; Pinno and Hawks, 2015). However, further research is needed to apply these findings to reclaimed sites as biogeochemical processes can vary considerably from natural and reclaimed forests (Quideau et al. 2013). The second possibility is that these are water limited sites, which is leading to increased growth at H6_F_12. This aligns with previous research that suggests FFM sites are typically less nutrient limited than PMM sites (Jamro et al., 2014; Kwak et al., 2016). Further, as previously discussed the FFM sites have lower water retention (Gringras – Hill et al. 2018), high infiltration rates, and low VWC relative to AWHC. These factors in addition to the FFM sites being underlain with tailings sand, which has been shown to cause water limitation (Naeth et al. 2011; Duan et al. 2015), suggests that the FFM sites in this study are likely water limited opposed to nutrient limited.

FRB allows a further analysis of site limitations when considering optimality theory, which is that trees should keep roots alive until the efficiency of resource acquisition is maximized (Espeleta & Donovan, 2002). Therefore sites that are nutrient or water limited will keep roots alive instead of recycling them in order to maximize resource acquisition. This has been observed in several studies in oil-sands reclamation (e.g. Naeth et al. 2011; Jung et al. 2014). However, results from this study suggest that broadleaf sites are typically more water limited than coniferous dominated sites. This is seen when comparing FRB in broadleaf sites, which had ~1100 kg ha^{-1} difference between young and old sites, while coniferous sites only had a ~40 kg ha^{-1} difference. Furthermore, root development in this study was unlikely to be restricted by bulk density as the bulk density for all sites fell within the ideal bulk density for root growth given the soil texture (Arshad et al., 1997). Overall this suggests that broadleaf sites have a higher water demand and are more likely to be water limited than coniferous sites (Stephenson 1998; Zha et al. 2010).

3.4.3 Vegetation Impacts on Water Regimes

As trees begin to take root and vegetation begins to develop the impact biotic factors have on the water regimes of reclaimed sites can be observed. At older sites ($\geq$ 10 years post reclamation) VWC decreased below θ_{PWP} more often than at younger sites (<10 years post reclamation), which may be due to increased uptake of water by vegetation leading to increased evapotranspiration rates at older sites (Strilesky et al., 2017; Chasmer et al., 2018). Further, when factoring for age broadleaf sites experienced more days where VWC at both depths fell below θ_{PWP}, this is likely due to greater vegetation development at broadleaf sites and a higher water demand (Stephenson, 1998; Zha et al., 2010; Strilesky et al., 2017; Chasmer et al., 2018). This is further supported when examining differences in FRB between broadleaf and coniferous sites. According to optimality theory (Espeleta & Donovan, 2002), sites that are nutrient or water limited will keep roots alive

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instead of recycling them in order to maximize resource acquisition. This has been observed in several studies in oil-sands reclamation (e.g. Naeth et al. 2011; Jung et al. 2014). In this study broadleaf sites showed ~1100 kg ha^{-1} difference in FRB between young and old sites while coniferous sites only had a ~40 kg ha^{-1} difference, suggesting broadleaf sites have a higher water demand and are more likely to be water limited than coniferous sites (Stephenson 1998; Zha et al. 2010). Additionally, root development may increase infiltration rates at reclaimed sites, increasing storage into groundwater and decreasing available water for plants (Wu et al., 2017), however further research is needed to apply these findings to boreal forest reclamation.

3.4.4 Influence of Soil vs Vegetation on Water Regimes

Immediately following revegetation of reclaimed sites soil prescriptions will govern water regimes. Soil texture will have a significant influence on AWHC and VWC in young ecosystems (Jung et al., 2014). However, these parameters may also be impacted by layering of soils (Zettl et al., 2011). Additionally, SOM and the presence of petroleum hydrocarbons may increase the soils ability to retain water and impact infiltration rates (Rawls et al. 2003; Neil and Si, 2018; Neil and Si, 2019). As soils age and paedogenesis beings to occur it is likely that fine textured sites will see significant increases in infiltration rates due to increased intra – aggregate bulk density and a separation of pore space (Hussein and Adey, 1998; Pires et al., 2008; Adey et al., 2019). Weathering may further increase infiltrations rates at shallow depths by reducing petroleum hydrocarbons (Neil and Si, 2018; Neil and Si, 2019). Furthermore, infiltration rates may be increased due to root growth, which was shown to occur more quickly in broadleaf dominated sites (Wu et al., 2017). Similarly, broadleaf dominated sites were shown to have increased tree growth and density. This can lead to increased water demands shown by VWC dropping below θ_{PWP} at sites dominated by broadleaf species, which is inline with previous studies showing increased

evapotranspiration rates at sites with greater vegetation development (Strilesky et al., 2017; Chasmer et al., 2018). Overall, as vegetation develops and paedogenesis beings to occur at it would be expected that infiltrations rate will increase and VWC will decrease, particularly at fine-textured broadleaf dominated sites.

3.4.5 Implications for Ecosystem Reclamation

Reclamation of whole ecosystems is becoming more common in the AOSR (Ketcheson et al. 2016), making the findings of this research important to discuss in the context of ecosystem reclamation. That is, ecosystem reclamation is planned with certain ecosystem functions expected, which will be essential to maintain self-sustaining at the landscape and mine closure scale. Thus, initial soil prescriptions that may be inline with desired planting designs need to be assessed over time to ensure that these approaches keep in step with the evolving moisture requirements of that vegetation. Finer textured soils show increasing infiltration rates over time due to paedogenesis and increased root growth resulting in an increase in preferential flow paths, which has the potential to make them more suitable uplands for fen reclamation (Horn & Smucker, 2005; Wu et al. 2017). Increased infiltration rates can increase percolation and ground water storage following precipitation events (Ketcheson & Price, 2016; Wu et al 2017), which are expected to increase under future climate scenarios (Keshta et al 2012). However, layering of soil and weathering may result in increased infiltrations rates only occurring in the top portion of the soil profile keeping more water in the rooting zone (Zettl et al., 2011; Neil and Si, 2018; Neil and Si, 2019). Furthermore, results from this study show that finer textured soils had a higher abundance of broadleaf species, which have been shown to increase in ET as trees develop, although this increase plateaus after approximately ten years (Chasmer et al., 2018; Strilesky et al. 2017). Future research

is needed to assess how infiltration into the groundwater will change as ecosystems develop and if increased infiltration in finer textured soils is offset by increased ET rates.

3.5 Conclusions

Differences in soil physical properties, particularly particle size, were shown to have a strong influence on VWC and AWHC. These parameters were found to largely govern vegetation diversity at sites, regardless of how sites were initially revegetated. This has led to sites being planted incorrectly due to too much emphasis on other parameters, resulting in an increased time for vegetation to become established, and a potentially longer period before sites can be certified. Further, particle size was shown to have a significant influence on infiltration rates with fine textured sites showing an increase in infiltration rates over time due to root development and pedogenesis. This finding was contrary to what was initially expected, as infiltration rates were expected to be higher at coarse textured sites due to lower water retention. Further research is needed to determine if the increase in infiltration would offset the increase in ET at fine textured sites that are dominated by broadleaf species, and if increases in infiltrations rates are only occurring at shallow depths. Furthermore, both the soil cover and the underlying mineral layer were shown to have a significant influence on soil water regimes. FFM was shown to have higher infiltration rates than PMM, likely due to PMM's higher water retention. These findings also suggest that a mineral substrate layer comprised of tailings sand material is more likely to create a water limited system than one constructed with overburden material, which is more likely to be nutrient limited.

When constructing future ecosystems considering the effects of soil texture, cover soil material, and mineral substrate material will be essential in predicting the moisture regimes of the site and determining the vegetation that must be planted. Such planning will allow for the best

usage of these materials and ensure constructed ecosystems will function as intended. For standalone forests, where the primary concern is tree growth, a combination of FFM and overburden material will be best suited that uses either coarse or fine textured soils depending if the goal is for a coniferous or broadleaf forest respectively. While when constructing upland forests to support fen ecosystems, where the primary goal is to support the hydrological requirements of the fen, a combination of coarse textured PMM and tailings sand will likely be most suitable. This will allow for a coniferous dominated site that is less water limited and has initially high infiltration rates. Water regimes at this site would be less variable overtime, allowing for a better prediction of water availability for the fen.

Chapter 4: Assessing the Effectiveness of Reclamation Cover Materials on the Recovery of Soil Nutrient Cycling Functions in a Post-Oil Sands Landscape

4.1 Introduction

In the AOSR, industrial development for bituminous oil sand through in-situ recovery and open pit mining has resulted in the disturbance of ~767 km^2 as of 2017. (Rooney et al., 2012; Government of Alberta, 2019) Alberta Government requires this land to be reclaimed back to an equivalent land capacity following the closure of mining operations (Government of Alberta 2000), which will require the reconstruction of native ecosystems and landforms of whole landscapes (Johnson & Miyanishi, 2008). This includes the reconstruction of the two dominant landforms of the region, fen peatlands that comprise 64% of the landscape, and upland forests, which comprise 23% (Rooney et al., 2012). While upland forests comprise a smaller proportion of the region, they play an essential role in the hydrological functions of the landscape (Devito et al. 2005). This makes upland forests essential for the reclamation of fen peatlands, for instance, Price et al. (2010) showed that it would take an upland to peatland ratio of 3:1. This will result in the conversion of land from a peatland dominated landscape to one dominated by forests (Rooney et al., 2012).

A key component in forest reclamation is understanding nutrient cycling during early succession as several studies have shown that increased nutrient availability may increase productivity in these novel ecosystems (Yan et al. 2012; Farnden et al. 2013; Pokharel et al. 2016). Nitrogen (N) is the predominant limiting nutrient within the boreal forests in the AOSR, particularly in reconstructed oilsands that lack the native N inputs of natural forests (Cheng et al. 2011; Bradshaw et al. 1987). Furthermore, different forms of N are preferentially taken up by vegetation. While NO_3^- - N is the preferred form of N for aspen (*Populus tremuloides*), jack pine (*Pinus banksiana*) and other coniferous species have shown an inability to take up NO_3^- - N

(Landhausser et al., 2010; Hangs et al., 2003). Similarly, phosphorus (P) has been shown to be a limiting nutrient on reclaimed sites, particularly at sites reclaimed using PMM (Pinno et al. 2012; Quideau et al. 2017). By understanding how nutrient cycling changes during early succession, how differences in soils and vegetation can impact nutrient availability and the feedback this has on vegetation development may be determined.

Differences in soils used while constructing these ecosystems can have a significant impact on the nutrient regimes of reclaimed forests. During reclamation, cover soils are placed over a mineral substrate layer to mitigate percolation into overburden and provide water and nutrients for vegetation development (Carey, 2008; Meiers et al. 2011; Rowland et al., 2009). Depending on the type of organic amendment used, these cover soils can differ in their texture, depths, and organic matter content (ex. Leatherdale et al., 2012; Haung et al., 2015; Quideau et al., 2017; Gringras – Hill et al., 2018). Common cover soils used in the AOSR are PMM, which is salvaged lowlands organic soil mixed with mineral substrate, and FFM which is salvaged upland boreal forest soil mixed with a mineral substrate (Mackenzie and Naeth, 2010). Several studies have examined the impact of using different cover soils on nutrient regimes of reclaimed sites (Quideau et al., 2017; Howell et al., 2015; Jamro et al., 2014; Mackenzie and Quideau, 2012; McMillian et al. 2007; Gringras – Hill et al., 2018; Hahn and Quideau, 2013; Farnden et al., 2013; Kwak et al., 2016). Jung et al. (2014) demonstrated that changes in soil texture interfaces can influence the distribution of nutrients. However, studies examining the different effects of organic amendments (PMM or FFM) on nutrient regimes have shown conflicting results (Quideau et al., 2017; Howell et al., 2015; Jamro et al., 2014; Mackenzie and Quideau, 2012; McMillian et al. 2007). Further, Quideau et al. (2013) found there was a disconnect between organic matter composition and nutrient availability of reconstructed soils. Due to these inconsistent findings, more research is

needed to assess how different cover soils impact nutrient regimes and whether soils are the dominant control in reclaimed forests.

Several studies have shown the impact vegetation can have on the nutrient regimes of boreal forests. Broadleaf forests have been shown to have higher P, C:N ratios and N concentration compared to coniferous forests (Flanagan and Van Cleve, 1983; Prescott et al., 2000; Jerabkova et al., 2006). Furthermore, Quideau et al. (2001) showed that there is a clear link between vegetation type and chemistry of the resulting SOM. However, studies done in oil sands reclamation have shown that reclaimed forest can differ in their biogeochemical processes considerably from natural forests, and there can be a disconnect between SOM and nutrient availability in these novel ecosystems (McMillian et al., 2007; Rowland et al., 2009; Quideau et al., 2013). More research is needed then to determine how vegetation may impact nutrient regimes and if it is one of the dominant controls on nutrient cycling in reclaimed sites.

The aim of this study is to examine several reclaimed sites in the AOSR that differ in the age since reclamation, the type of cover soil used (PMM or FMM) and resulting soil properties, and vegetation planted. Soil and litter extractable nitrate-nitrogen (NO_3^- - N), ammonium-nitrogen (NH_4^+- N), and phosphorus (P) will be examined along with mineralization rates during peak growing season to determine how soil properties influence the nutrient regimes of reclaimed sites, and whether litter or soil contributes more to nutrient availability. Furthermore, comparing results from young sites (5 years since revegetated) and older sites ($\geq$ 10 years since revegetated) will provide insight on how the nutrient regimes of these sites change as vegetation develops. The objectives of this study are to determine how differences in cover soils will influence nutrient regimes of reclaimed forests of varying ages and to determine whether litter or soil is the main contributor to nutrient availability in these ecosystems. It is hypothesized that differences in cover

soils will have a significant impact on nutrient regimes and that litter will have a greater contribution to nutrient mineralization rates than soil. These findings will permit the assessment of how soil properties and vegetation may influence nutrient regimes of reclaimed sites and the impact this has on reclamation success.

4.2 Materials and Methods

4.2.1 Soil Properties

Twelve intact soil cores were collected at each site using PVC pipe (10 cm diameter x 10 cm length) driven into the ground. Once removed, cores were wrapped in polyethylene film and stored for transportation to the Hydrometeorology Lab, University of Waterloo where they were analyzed for porosity, bulk density, and specific yield following standard methods (c.f. Freeze and Cherry 1979; Klute 1986). Additionally, a subset of soils was analyzed for SOM by loss on ignition at 550°C for 3 hours (Dean, 1974). A second subset of samples were analyzed for texture by sieving the soil through a 2 mm sieve and using a laser scattering particle size analyser (Horiba LA – 950V2, Kyoto, Japan) to measure particle size distribution. In the field, single-ring infiltrometers were used to measure infiltration rate (f) at twelve points along each slope. Six infiltrometers were installed in each site at a minimum of 1cm depth and tests were conducted until a steady state was observed in order to account for antecedent moisture conditions (minimum five consecutive measurements within $\pm$ ~15%), the length of this process ranged from ~0.5 – 1.5 hours depending on the initial saturation of the soil (Ketcheson & Price, 2016).

4.2.2 Vegetation

Tree surveys were conducted along four transects running vertically through the slopes between late June – early August 2018 (Figure 3 – 1). At each point the number of trees within a 5 m radius were counted with species and tree height recorded using a clinometer (Suunto, Vantaa, Finland). To measure root development, fine root biomass (FRB) was determined for each site using the

sequential core method, where three cores were collected using PVC pipe (10 cm diameter x 30 cm length) between the end of July and beginning of August (Vogt & Persson, 1991). Roots were collected by soaking samples in tap water over night, rubbing them gently to separate the roots from the soil and pouring the water through a sieve (0.2 mm). This was repeated until only rocks and organic debris were left in the soil. Live and dead roots were then determined and separated from one another. Roots were considered live if they were pale in colour and free of decay and were considered dead if they were black or brown in colour and inflexible. Live roots were then oven dried at 70 °C for 24 hours and weighed. FRB was calculated as dry mass of living roots (g) x 10^{-3} x 10^8/area (m^2) of the core (McClaugherty et al. 1982).

4.2.3 Biogeochemical Sampling

Net rates of nitrogen (N) and phosphorus (P) mineralization in soil and litter were determined through in-situ buried-bag incubation conducted over a three-week period from June – July 2018 (Hart et al. 1994; Macrae et al. 2013). Subsamples were extracted in 50 ml of distilled-deionized water for analysis of soluble reactive phosphorus (SRP) and nitrate (NO_3^- - N), while a second subsample was extracted in 50 ml of KCl for ammonia (NH_4^+- N). All filtered extractions were analyzed using colorimetric analysis at the Biogeochemistry Lab at the University of Waterloo (Bran Luebbe AA3, Seal Analytical, Seattle, U.S.A., Methods G-102-93 (NH_4^+- N), G-109-94 (NO_3^- - N), and G-103-93 (SRP)). Net ammonification was calculated as NH_4^+- N accumulated after 3 weeks minus NH_4^+- N at the beginning, net nitrification rate as NO_3^- - N at the end minus NO_3^- - N at the beginning, net N mineralization rate as inorganic N (NH_4^+- N + NO_3^- - N) at the end minus inorganic N at the beginning, and net P mineralization rate as P at the end minis P at the beginning.

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Biomass samples, comprised of the foliage of living trees, were collected in July 2018 and were frozen in July 2018 to transport back to the University of Waterloo to analyze for C:N:P ratios. Samples were thawed at room temperature and dried at 80 °C for 24 hours before being ground. Three subsamples from each site were analyzed for C and N using EA-IRMS (Thermo Scientific, Waltham, United States) at the Environmental Isotope Laboratory, University of Waterloo. A second group of subsamples were digested (Parkinson and Allen, 1975) and analyzed for P using ICP analysis (Optima 8000 ICP-OES, Perkin Elmer, Waltham, United States) at the Centre for Cold Regions and Water Science, Wilfrid Laurier University.

4.2.4 Statistical Analysis

All statistical analysis was performed with R (R Development Core Team, 2013). Data was tested for normality using a Sharpiro – Wilk test. All biogeochemical data was found to be non-normally distributed and therefor was analyzed using non-parametric test. Kruskal – Wallis test were performed to compare nutrient regimes in soils and litter to each other. Separate Kruskal – Wallis tests were then performed on soil mineralization rates and extractable nutrient concentrations, followed by a Dunn post-hoc analysis to examine which sites shared similar soil nutrient concentration and mineralization rates. The same tests were then preformed on litter nutrient mineralization rates and extractable nutrient concentrations. A principal component analysis (PCA) was performed on soil mineralization rates to determine how soil physical properties influenced P, NH_4^+- N, and NO_3^- - N mineralization. Finally, a second PCA was on litter mineralization rates to examine the relationship between macronutrient concentrations, vegetation and litter P, NH_4^+- N, and NO_3^- - N mineralization. Kruskal – Wallis tests were considered significant in $p < 0.05$.

4.3 Results

4.3.1 Soil Physical Properties

Soil texture was finest at SBH_P_04, consisting of a clay-loam (Table 3.1). This was followed by four sites (E_P_08, SE_P_12, W_P_12 and H6_F_12) that were characterized by loam soils, of which H6_F_12 and W_P_12 had the highest clay and sand content, respectively. W_P_12 and C_P_06 were the coarsest textured sites comprised of loamy-sand and sandy-loam respectively. SOM was highest at SBH_P_04 (0.31) followed by E_P_08, SE_P_12, W_P_12 and H6_F_12 (0.17, 0.16, 0.21 and 0.22, respectively). Coarse textured sites (C_P_06 and H7_F_12) had the lowest SOM (0.13 and 0.07 respectively). Bulk density was significantly lower at SBH_P_04 (0.65 g ml^{-1}), which also saw the highest porosity and infiltration rates (0.48 and 1089 mm hr^{-1} respectively). The opposite trend was observed at 2012, which had the highest bulk density (1.12 g ml^{-1}) while having the lowest porosity (0.37) and infiltration rates (142 mm hr^{-1}) (Table 3.1).

Table 4.1: Bulk density, porosity, infiltration rates, organic matter and soil texture at reclaimed sites in 2018

Site	Bulk Density (g ml^{-1})	Porosity	Infiltration Rates (mm hr^{-1})	Organic Matter	Soil Texture		
					% Sand	% Silt	% Clay
SBH_P_04	0.65	0.48	1089	0.31	21.6	42.7	35.7
C_P_06	1.08	0.46	585	0.13	60.5	30.7	8.8
E_P_08	1.04	0.45	658	0.17	40.4	39.4	20.2
SE_P_12	1.12	0.37	142	0.16	41.4	46.3	12.3
W_P_12	0.98	0.39	239	0.21	51.6	38.5	9.9
H6_F_12	0.91	0.46	607	0.22	39.8	36.2	24.0
H7_F_12	0.99	0.48	681	0.07	79.0	17.5	3.5

4.3.2 Vegetation Development

SBH_P_04 had the highest tree growth with the tallest tree measured (18.7m) and heights averaging >10 metres (Table 4.2). Further, SBH_P_04 had the highest FRB (1196 kg ha^{-1}) by over 600 kg ha^{-1}. H6_F_12 and H7_F_12 had greater tree growth (1.5 m, 1.4 m), density (5613 stems

ha^{-1}, 4138 stems ha^{-1}) and FRB (483 kg ha^{-1}, 276 kg ha^{-1}) than SE_P_12 and W_P_12. The tallest trees at all sites were broadleaf, apart from C_P_06, which only had coniferous species. Furthermore, tree height, density and FRB were greater at broadleaf dominated sites than at coniferous dominated sites.

Species diversity changed considerably from what was initially planted at some sites. At C_P_06, the planting prescription was predominantly jack pine with some broadleaf species (white birch). However, the site is now solely dominated by jack pine (Table 4.2). This change in species composition over time was also found at SE_P_12 and W_P_12. SE_P_12 and W_P_12 were initially classified as different ecosites (ecosite-d and ecosite a/b, respectively) and planted accordingly, however tree surveys done in 2018 showed both sites having a similar mixed composition. Overall, younger sites were regularly found to be comprised of mixed vegetation, while older sites were dominated by either coniferous or broadleaf species, regardless of initial planting prescription (Table 4.2).

Table 4.2: Mean and maximum tree height, density, fine root biomass (FRB) and species composition from the 2018 growing season.

Site	Mean Tree Height (m)	Max Tree Height (m)	Density (stems ha^{-1})	FRB (kg ha^{-1})	Species Composition (%)	
					Coniferous	Broadleaf
SBH_P_04	13.6	18.7 (Aspen)	6215	1196	4	96
C_P_06	3.4	3.6 (Jack Pine)	2560	230	100	0
E_P_08	1.6	10.7 (Aspen)	14565	535	1	99
SE_P_12	1.0	3.6 (Poplar)	1167	104	48	52
W_P_12	1.1	2.6 (Aspen)	1984	123	69	31
H6_F_12	1.5	3.3 (Poplar)	5613	483	20	80
H7_F_12	1.4	4.0 (White Birch)	4138	276	71	29

4.3.3 Macronutrient Concentrations & Extractable N & P

Figure 4.1 shows average extractable nitrogen (N), ammonium (NH_4^+- N) and phosphorus (P) were significantly higher in litter than in soil in June and July ($p < 0.05$). In contrast, extractable nitrate (NO_3^- - N) was higher in soil than litter ($p < 0.05$), although it only comprised ~23% of total extractable N. Extractable N, NH_4^+- N and P was not significantly different between any sites for soil, while NO_3^- - N was significantly higher in H6_F_12 and C_P_06 (1018 mg g-1 and 1318 mg g-1 respectively), followed by SE_P_12 and W_P_12, with E_P_08 having the lowest extractable NO_3^- - N (15 mg g-1). Litter extractable NO_3^- - N was low for all sites with E_P_08 and SBH_P_04 having 0 mg g-1. Extractable N and NH_4^+- N was highest at H6_F_12 and H7_F_12 (60456 mg g-1 and 51550 mg g-1, respectively), with the two oldest sites (SBH_P_04 & C_P_06) having similar N (9875 mg g-1 and 7198 mg g-1 respectively), and E_P_08 having the lowest (4794 mg g-1). Litter extractable P was greatest in SBH_P_04 (37 mg g-1) and lowest in C_P_06 (0.98 mg g-1) with remaining sites showing no statistical difference between them.
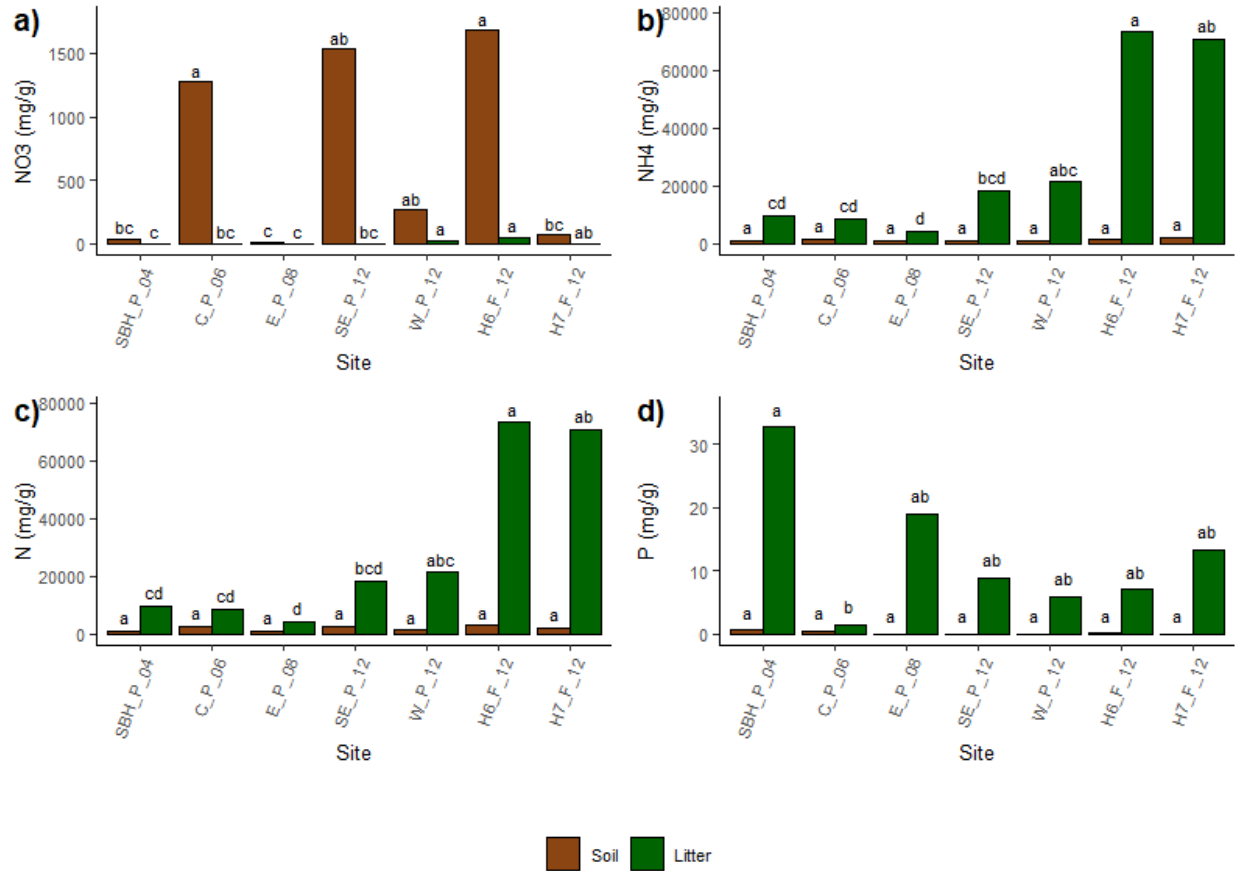
Figure 4.1: Average soil and litter extractable NO_3^- - N (a), NH_4^+ - N (b), N (c) and P (d) between June – July 2018. Results from a Dunn post – hoc analysis are displayed as letters above bars, separate post – hoc analysis was performed on soil and litter.

Carbon (C) and N ratios were highest at C_P_06 and H7_F_12 (55 mg g^{-1} and 51 mg g^{-1} respectively), which had a greater abundance of coniferous species, and were lowest at broadleaf dominated sites (SBH_P_04, E_P_08, and H6_F_12). SE_P_12 and W_P_12 had C:N ratios in between values representative of coniferous and broadleaf sites (38 mg g^{-1} and 31 mg g^{-1}, respectively), but were closer to broadleaf than coniferous sites (Table 4.3). C:P ratios were lowest at E_P_08 and SBH_P_04 (112 mg g^{-1} and 135 mg g^{-1} respectively) while being highest at C_P_06 (251 mg g^{-1}). Among the sites reclaimed in 2012, the broadleaf dominated site (H6_F_12) had the lowest C:P ratio (159 mg g^{-1}), while SE_P_12, W_P_12, and H7_F_12 had similar ratios (182 mg

g^{-1}, 192 mg g^{-1}, and 188 mg g^{-1}, respectively). N:P ratios were similar across sites, ranging from 4 – 5.

Table 4.3: Plant biomass C, N, and P concentrations at reclaimed sites during the 2018 growing season.

Site	Macronutrient Concentrations			Nutrient Ratios		
	C	N	P	C:N	C:P	N:P
	(mg/g)			ratio		
SBH_P_04	484.9	17.7	3.6	28	135	5
C_P_06	535.5	9.9	2.1	55	251	5
E_P_08	483.8	15.8	4.3	31	112	4
SE_P_12	505.4	13.3	2.8	38	182	5
W_P_12	510. 5	13.7	2.7	37	192	5
H6_F_12	517.5	16.0	3.3	32	159	5
H7_F_12	520.3	10.2	2.8	51	188	4

4.3.4 Nutrient Mineralization Rates

Similar to total extractable nutrients, litter had higher mineralization rates for NH4+- N, N and P than soil while NO3- - N was lower ($p < 0.05$). There was no significant difference between total inorganic NH4+- N mineralization rates in the litter across sites, while NO3- - N mineralization differed in H6_F_12 (-20627 mg g-1) and H7_F_12 (-7870 mg g-1) (Figure 4.2). Litter P mineralization rates were highest in SE_P_12 (8.17 mg g-1) and lowest in H7_F_12 (-12.8 mg g-1), which experienced a net immobilization, the remaining sites experienced no significant difference between one another. Soil NH4+- N and N mineralization was highest at C_P_06 (785 mg g-1 and 2063 mg g-1 respectively) and lowest at SBH_P_04 (-548 mg g-1 and -1406 mg g-1). NH4+- N at W_P_12 and C_P_06 were statistically different from SBH_P_04 ($p = 0.01$ and 0.007 respectively) while all other sites were statistically similar ($p > 0.05$), despite differences in soil physical properties. For N mineralization there were not clear links between soil properties and mineralization rates. With the young, coarse textured, FFM site (H7_F_12) being statistically similar to the old, fine textured, PMM site (SBH_P_04, $p = 0.13$). Similarly, SE_P_12 and

W_P_12 were statically similar to C_P_06 (p = 0.14 and p = 0.12, respectively) despite differences in age and soil properties. NO3- - N mineralization occurred the most in C_P_06 and HE_P_12 (2592 mg g-1 and 1046 mg g-1) followed by H6_F_12 and W_P_12 (938 mg g-1 and 134.27 mg g-1) showing no clear links between mineralization rates and soil properties, all other sites were statistically similar.
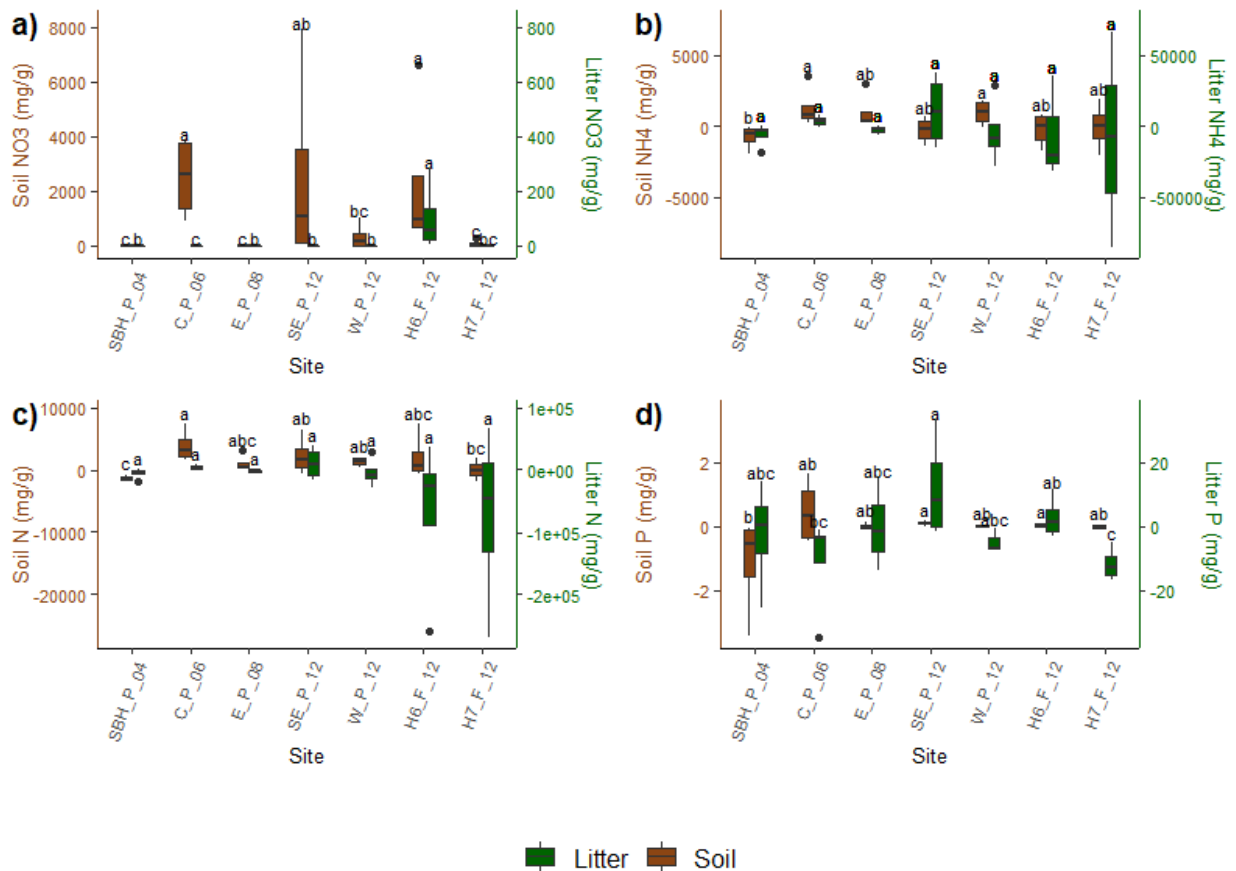
Figure 4.2: Soil and litter NO_3^- - N (a), NH_4^+- N (b), N (c) and P (d) mineralization rates over a three week period from June – July 2018. Results from a Dunn post – hoc analysis are displayed as letters above bars, separate post – hoc analysis was performed on soil and litter.

Results from the principal component analysis (PCA) showed that soil NH_4^+- N and NO_3^- - N were related to one another as well as higher silt content and temperature (Figure 4.3a). The sites most characterized by N and NH_4^+- N mineralization rates were mixed sites (SE_P_12 and W_P_12). P mineralization rates were closely related to increases in temperature and sand content and were

typically higher in sites dominated by coniferous species. In contrast to coniferous and mixed sites, broadleaf sites were not characterized by increased nutrient mineralization rates, and instead were more related to high soil moisture and clay content. A second PCA comparing nutrient mineralization rates in the litter (Figure 4.3b) showed that coniferous sites were characterized by high C:P and C:N ratios and were inversely related to P and NO_3^- - N mineralization rates. In contrast broadleaf sites were characterized by high P and NO_3^- - N mineralization and lower C:P and C:N ratios. NH_4^+- N mineralization was slightly related to NO_3^- - N mineralization but was independent of C:P and C:N ratios and P mineralization rates. Mixed sites were not strongly associated with any parameter.

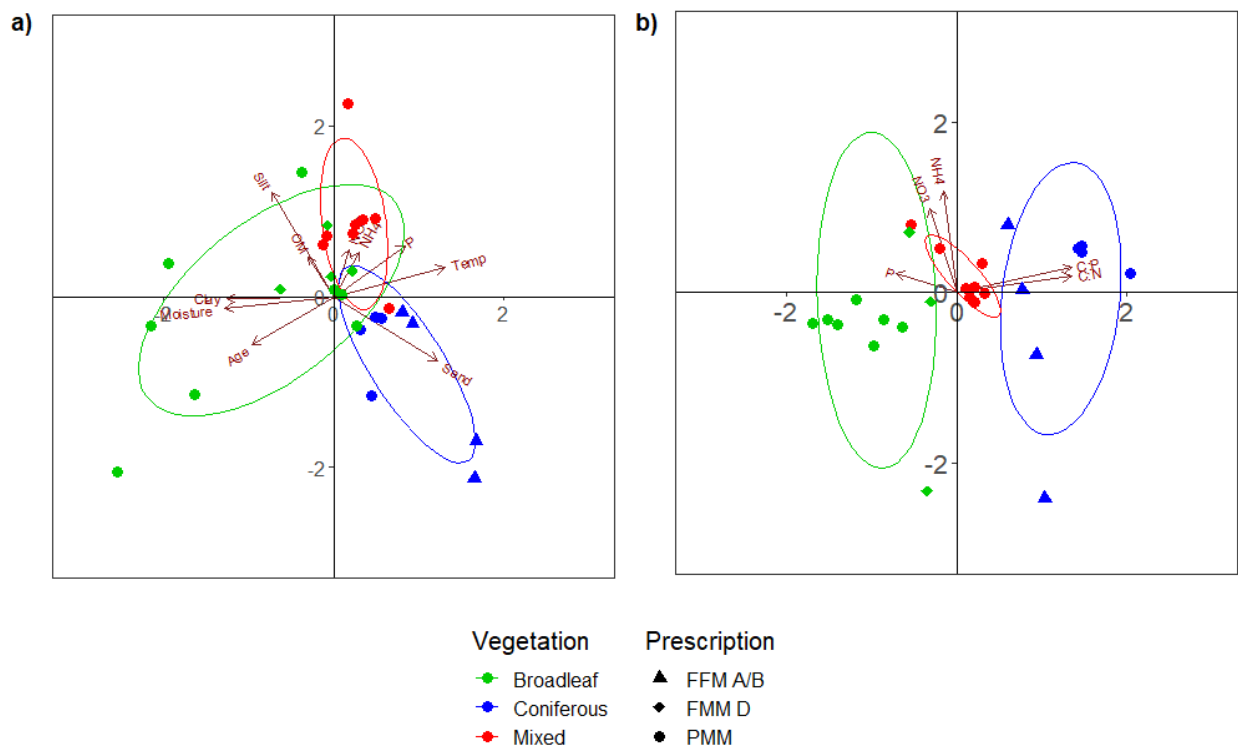**Figure 4.3:** Principal component analysis of soil (a) and litter (b) mineralization rates over a three-week period from June – July 2018.

4.4 Discussion

4.4.1 Reclamation Prescriptions Influence on Nutrient Cycling

Result of this study suggest that soil mineralization rates and extractable nutrients were not impacted by the use of FFM or PMM, contrary to several recent studies comparing the different cover soils (Kwak et al. 2016, Howell et al. 2016, Gringras – Hill et al. 2018). However, Quideau et al. (2017) established that when N mineralization rates are expressed on a soil weight basis, studies have often reported conflicting results. Furthermore, N release has been shown to decrease in FFM after only 25 weeks, while PMM can maintain consistent mineralization rates for over 45 weeks due to the slower decomposition of its organic matter (MacKenzie and Quideau, 2012; Quideau et al., 2017). These findings suggest that any initial benefit in mineralization at FFM sites may only be during the first couple years following placement. The timing of this study may have also affected the differences observed between PMM and FFM. Soil nutrient concentrations have been shown to be relatively low during the summer and high in the fall in reclaimed sites (Jamro et al. 2014). Further, contrary to previous studies, there was no observed difference in extractable P in PMM soils based on soil nutrient regimes, suggesting that PMM soils may not be P limited as previously thought (Pinno et al. 2012; Quideau et al. 2017; Mackenzie and Naeth, 2010), however further research is needed to confirm this finding. Differences in soil nutrient regimes may be driven more by physical characteristics independent of the type of organic amendment used in the cover soil used (Farnden et al. 2013)

From the soil properties measured, nutrient mineralization rates were most closely linked to increases in soil temperature and changes to soil texture (Jung et al., 2014). Contrary to what has been found in natural environments, SOM had no significant influence on nutrient availability at reclaimed sites (Chaer et al., 2009). Similarly, Quideau et al. (2013) found that there was a disconnect between SOM composition and nutrient availability in reconstructed soils, although

this may be due to the presence of recalcitrant SOM at reclaimed sites (Larney and Angers, 2011; Quideau et al., 2017). Contrasting soil mineralization rates were found in the oldest sites studied (SBH_P_04, and C_P_06), where SBH_P_04 had a net immobilization for all nutrients, while C_P_06 had one of the statistically highest mineralization rates across all nutrients. These findings may be the result of biotic differences between sites, as SBH_P_04 is characterized by a high abundance of broadleaf species, while C_P_06 is dominated by conifers at a much lower density (Jerabkova et al. 2006; Quideau et al. 2013). However, similar findings were not observed in soil mineralization rates of 2012 sites that differed in their species composition and density. Biotic differences that can drive mineralization rates may be the result of differences in soil prescriptions used when sites were initially revegetated, as species composition can be driven by abiotic factors that soil physical properties can influence (Prentice et al., 2020; Pinno and Hawks, 2015). However, differences in abiotic factors themselves may also drive biogeochemical processes (Brockett et al. 2012; Quideau et al. 2017; Prescott et al. 2000).

Several studies have shown the impact soils can have on moisture regimes and temperatures of reclaimed sites (Prentice et al., 2020; Pinno and Hawks, 2015). These climatic variables have been shown to be significant drivers of soil nutrient regimes and can be difficult to account for in field-based experiments (Prescott et al., 2000; Klinka et al. 1996). Low ground temperatures may be resulting in low soil mineralization at SBH_P_04, however high temperatures did not necessarily result in greater mineralization rates at other sites. Higher NH_4^+- N mineralization rates are likely the result of a combination of high temperatures and soil moisture (Hemstock et al., 2010; McMillan et al., 2007). Shown by the high NH_4^+- N mineralization of C_P_06, and W_P_12, which were not limited by low temperatures or VWC. In contrast, NO_3^- - N and P mineralization rates did not follow the same trend. High P and NO_3^- - N mineralization

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rates were found in sites with a south facing aspect (C_P_06, H6_F_12) or low tree density (SE_P_12, W_P_12). This may be due to more solar radiation reaching the ground surface heating incubation bags, increasing mineralization rates, although climate results did not show any significantly higher ground or air temperature at these sites. The influence of tree density is one of the many ways vegetation may impact the nutrient regimes of the site (Prescott et al. 2000; Prescott and Vesterdal, 2005; Jerabkova et al. 2006), although abiotic factors tend to play a greater role in the nutrient concentrations of the soils themselves (Lamarche et al. 2004; Ponge et al. 2011).

4.4.2 Vegetation Impacts on Nutrient Cycling

The influence of vegetation on nutrient availability in boreal forests soils has been well documented (Prescott et al. 2000; Prescott and Vesterdal, 2005; Jerabkova et al. 2006). However, recent studies have shown that reclaimed forest soils can differ in their biogeochemical processes considerably from natural forests (Quideau et al. 2013; Rowland et al., 2009, McMillian et al., 2007). Although in this study it was found that abiotic controls had a greater influence on nutrient regimes in the soil (Lamarche et al. 2004; Ponge et al. 2011), nutrient regimes within the litter followed similar trends with what would be expected in boreal forests. That is, broadleaf stands did not have a significantly higher N or NH_4^+- N mineralization despite a higher total N concentration, and lowest C:N ratios of forests types (Jerabkova et al. 2006). Furthermore, highest P concentrations and C:P ratios were observed at broadleaf sites, likely due to high P – input of aspen litter, which corresponded to higher P mineralization rates (Flanagan and Van Cleve, 1983; Jerabkova et al. 2006). These findings suggest that despite soil nutrient regimes not reflecting that of natural boreal forests, vegetation inputs will remain similar to natural forests.

Despite the type of organic amendment used showing no significant impact on soil nutrient regimes, sites reclaimed using FFM had higher extractable N and NH_4^+- N in the litter and were

the only sites that showed any NO_3^- - N mineralization. This may be due to initially high nutrient mineralization rates resulting in nutrients becoming stockpiled in plant biomass, similarly to what has been observed in nursery seedlings that were fertilized prior to transplanting at reclaimed sites (Pokharel & Chang, 2016). Further, seedlings that were fertilized prior to transplantation only typically see higher N concentrations in the first year. This may explain why higher extractable nutrients were not reflected in the macronutrient concentrations of biomass samples collected during the 2018 growing season. However, increased extractable N at these sites may be reflective of higher deposition occurring in this watershed, although similar trends were only observed in the soils of one of these sites (Bytnerowicz et al., 2010; Hemsley et al., 2012).

Sites reclaimed in 2012 had higher litter extractable N and NH_4^+- N than older sites, regardless of the mineralization rates and macronutrient concentrations observed. This may be due to increased root development at older sites, resulting in greater access and uptake of available nutrients (Espeleta and Donovan, 2002; Jung et al., 2014; Naeth et al., 2011). This is supported by the low mineralization rates at SBH_P_04 and E_P_08, which had the highest FRB and tree density of all sites studied. However, C_P_06 had lower extractable NH_4^+- N than H6_F_12 and H7_F_12, despite having higher NH_4^+- N mineralization rates and lower FRB. Furthermore, the use of coarse textured soils and fine tailings as a secondary cover soil would most likely result in a water limited system over a nutrient limited one (Prentice et al. 2020; Naeth et al. 2011; Duan et al. 2015). Despite this, C_P_06 is likely nitrogen limited as suggested by its high C:N and low N:P ratios.

4.4.3 Nutrient availability in Soil vs Litter
Higher concentration of extractable N, NH_4^+- N and P in the litter of reclaimed sites than the soil suggests that in these novel ecosystems, vegetation contributes more to nutrient availability than

soil after only five years since revegetation. This aligns with typical observations of natural boreal forests, where vegetation can significantly influence nutrient availability (Jerabkova et al. 2006; Tan and Chang, 2007). Soils only contributed more to nutrient cycling in NO_3^- - N mineralization and availability, which has been shown to be an important source of N for some boreal forest species (Landhausser et al., 2010). Although it has been suggested that high NO_3^- - N availability in reclaimed sites may be due to their close proximity to industrial emissions (Quideau et al. 2013; Bytnerowicz et al., 2010; Hemsley, 2012), this was only reflected in the litter of two sites, and to a much lesser extent than the soils (). These results suggest that higher NO_3^- - N availability may not solely be due to atmospheric inputs at reclaimed sites, and NO_3^- - N mineralization in the soil may still be significant contributor to nutrient availability. Furthermore, low P mineralization rates in the litter of coniferous sites suggest that soils may still play significant role in providing P to these sites (Flanagan and Van Cleve, 1983). Overall, results suggest that soils are likely the predominant source of NO_3^- - N at reclaimed sites regardless of vegetation, and may be a potentially important source of P at coniferous sites. In contrast litter is a major contributor of N and NH_4^+- N at reclaimed sites regardless of vegetation, and an important source of P at broadleaf sites.

4.5 Conclusion

This study tested the impact soil prescriptions used in reclamation have on the nutrient mineralization rates of reclaimed forests. Contrary to what was hypothesized, differences in soil prescriptions used in reclamation had a little impact on the nutrient regimes. The observed lack of differences in nutrient regimes between FFM and PMM suggests that in as little as five years post-revegetation, any initial benefits of amendment type to the nutrient regimes of the soil may no longer be present. However, higher silt content did have a slight impact on N, NH_4^+- N, and NO_3^-

- N mineralization rates. Further, NH_4^+- N soil mineralization was greatest in sites that were neither limited by VWC or ground temperature while NO_3^- - N, and P soil mineralization was greatest at sites with low tree density and south facing aspects. In contrast to soil mineralization rates, litter mineralization rates followed a similar trend to what would be expected in natural boreal forests. That is, sites with a greater abundance of broadleaf species had higher P mineralization rates in the litter and lower C:P ratios, while NH_4^+- N, and N were unrelated to vegetation type. This finding was unexpected as previous studies have shown that reclaimed forests can differ in their biogeochemical processes from natural forests considerably. As hypothesized, litter was found to be a greater contributor to nutrient availability than soil, apart from NO_3^- - N, which was only mineralized in the litter of FFM sites.

Future reclamation projects will need to consider soils impact on nutrient regimes immediately following revegetation, on the long-term impacts to NO_3^- - N mineralization, and P mineralization at coniferous sites and the resulting effects on plant successional pathways. However, once vegetation becomes established litter will drive N and NH_4^+- N availability. These findings suggest that when assessing reclaimed ecosystems, mineralization rates of litter may prove to be a better for benchmark for ecosystem reclamation then mineralization rates of soil. However, further research is needed to determine how increases in litter depth may impact nutrient mineralization rates, and what litter depth would be required to sufficiently supply the ecosystem with enough available nutrients to meet vegetation demands in broadleaf and coniferous sites.

Chapter 5: Conclusion & Limitations

When reclaiming forest ecosystems, differences in the soil prescriptions used was found to directly impact the water regimes of the sites. Soil texture had a particularly strong influence on VWC, AWHC, and infiltration rates. Further, forest floor material (FFM) sites were found to have higher infiltration rates than peat mineral mix (PMM) sites likely due to PMM's higher water retention. These findings also suggest that sites reclaimed using a mineral substrate layer comprised of tailings sand material were more likely to create a water limited system than one constructed with overburden material. However, this study was limited by variability in the depth of soil moisture probes between sites, which may have impacted VWC measurements during the growing season. Furthermore, infiltrometers and the collection of soil samples at varying depths would have allowed for the characterisation of soil hydrophysical properties throughout the soil profile opposed to the top 10 cm.

Despites the significant impact of soil prescriptions on water regimes, the impact on nutrient regimes was smaller than hypothesized, with a lack of differences observed between FFM and PMM. This suggests that in as little as five years post – revegetation any initial benefits to the nutrient regimes of the soil will no longer be present. Further, although higher silt content did have a slight impact on N, NH_4^+ and NO_3^- mineralization rates this was insignificant compared to nutrient inputs from litter. Extractable P, N and NH_4^+ were higher in litter than in soil, while NO_3^- was only mineralized in the litter of FFM sites. More research is needed to determine what caused NO_3^- mineralization to occur only in the litter of these sites. Overall, nutrient mineralization rates in the litter followed similar patterns to what is observed in natural boreal forests, where P mineralization was higher at broadleaf sites while N and NH_4^+ were unaffected by vegetation type. This research was limited by samples only being collected during peak growing season, as previous

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studies have shown that nutrient concentrations are relatively low during the summer and higher in the fall. Additionally, due to this being a field-based study the impact soils have on nutrient mineralization could have went unobserved due to microclimatic differences having a greater control on nutrient mineralization.

When constructing future ecosystems, consideration of the long-term impact soil prescriptions have on water and nutrient regimes will be essential for successful reclamation. Although soil prescriptions had a smaller direct impact on nutrient regimes then hypothesized, their influence on water regimes will govern vegetation establishment and thereby influence the nutrient regimes. For the reclamation of standalone forests, where the primary concern is tree growth, the use of fine textured FFM and overburden material will likely be best suited. This will result in a broadleaf site that is unlikely to be water limited and where litter will provide sufficient N and P inputs to the system five years post-revegetation. While when constructing upland forests to support the hydrological requirements of fen ecosystems, a combination of coarse textured FFM and tailings sand will likely be most suitable. This will allow for a coniferous dominated site that has initially high infiltration rates and where water regimes will be less variable overtime. However, the upland of this ecosystem would likely suffer from low water availability and low P inputs, which may prove detrimental overtime. To avoid this PMM may be used in place of FFM, which will likely result in a less water limited upland, although infiltration rates may initially be lower.

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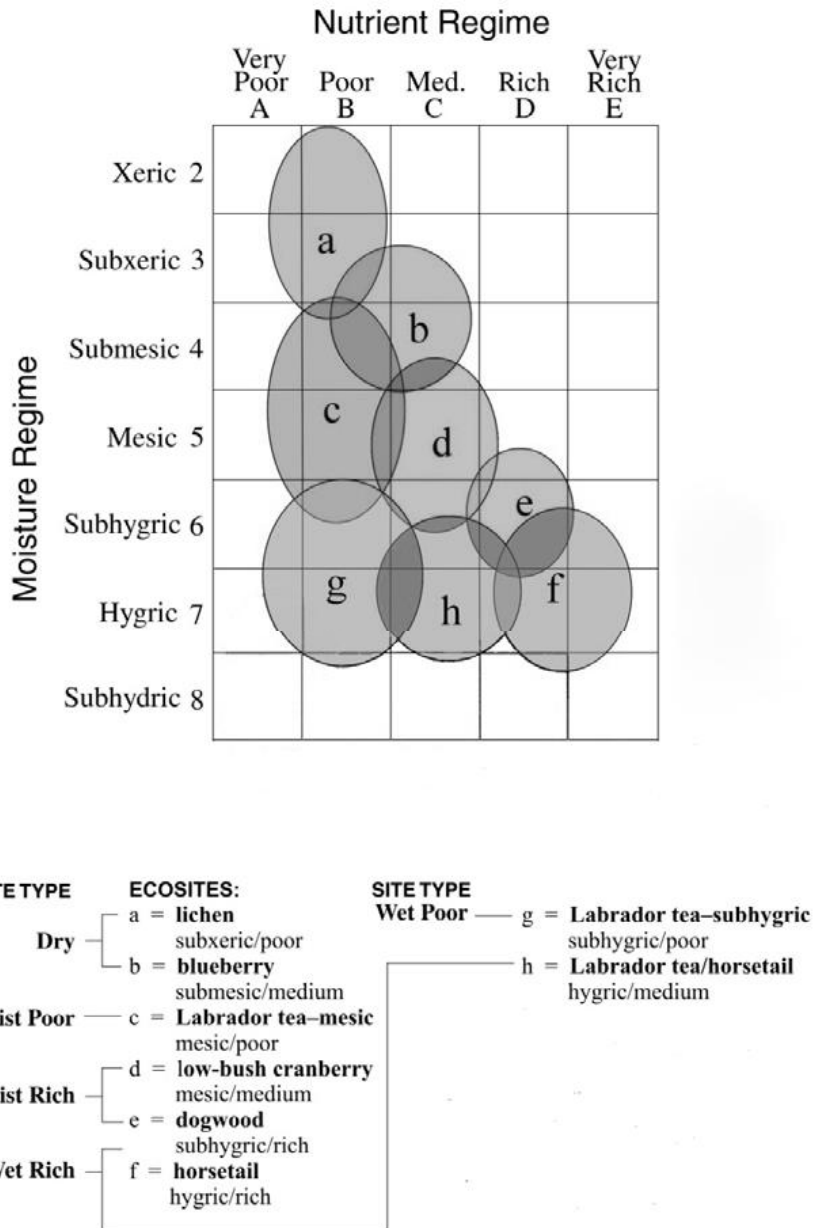
Appendix

Figure A 1: Site types and ecosites of boreal forests in relation to soil moisture and nutrient regimes. Source: Straker and Donald, 2010.