An Alternate Indicator System for Nutrient Supply as Part of Ecosystem Function, a Component of Reclamation Success in the Athabasca Oil Sands Region

by

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Abstract

Northeastern Alberta faces the challenges of reclaiming a vast area that has been disturbed my oil sands mining, cumulatively 896 km^2 and increasing as of December 2013. The limited resources available for reclamation and expensive costs of undertaking the process necessitates that reclamation be as effective and efficient as possible in reaching its goal of establishing "equivalent land capability" to ecosystems that existed prior to disturbance. To achieve this goal, the best possible indicators of reclamation success should be used. Foliar macronutrient concentrations are commonly recommended as an indicator for soil nutrient levels in wildland ecosystems, but questions as to the validity and sensitivity of this indicator have been raised. This study measured the nutrient concentrations in three different pools of nutrients (soil, bioavailable, and foliar), on sites reclaimed using different reclamation treatments and on sites disturbed naturally by wildfire in order to determine if the current methods for indicating the success of establishment of nutrient cycling were valid and to determine if alternative systems of indicators could better identify differences between reclaimed and natural ecosites. The study was divided into two components: 1) Assessing the validity of the assumption that foliar nutrient concentrations were a useful indicator for belowground nutrient pools; and 2) Developing and demonstrating a different system for the assessment of similarity between sites' nutrient profiles.

1) A study on CNRL's Reclamation Area-1 (RA-1) compares whether trembling aspen (*Populus tremuloides* Michx.) foliar concentrations of macronutrients are an accurate representation of the bioavailable and soil nutrient pools in both natural and reclaimed forest ecosystems and if a multivariate similarity metric, rather than critical values for individual nutrients, could provide a more meaningful assessment of key differences between ecosystems. Individual macronutrient concentrations were different between treatments in the total soil

nutrient pool, but differences decreased in the soil bioavailable pool and disappeared in the foliar nutrient pool. Few significant correlations between the foliar and belowground nutrient pools were observed. Multivariate analysis, consisting of non-metric multi-dimensional scaling and multiple response permutation procedures, showed a similar level of differences between natural wildfire-impacted reference sites and the reclaimed treatment sites across all three nutrient pools. Again, few significant multivariate correlations between foliar and belowground nutrient pools were observed. Comparison of the relative similarity of different reclamation treatments to the natural reference sites showed there are differences between the reclamation treatments not reflected by individual macronutrient levels.

2) A study on Syncrude Canada's Aurora Soil Capping Study provided the opportunity to use principal components analysis to reduce data from trembling aspen and jack pine (*Pinus banksiana* Lamb.) foliar nutrient concentrations, and bioavailable and soil nutrients from both topsoil and upper subsoil to a minimum dataset that best represented the differences between a natural forest ecosystem and reclaimed ecosystems on sites capped with different topsoil materials and depths of subsoil material. The minimum dataset for this combination of site types was found to be differentiated by aspen foliar Mn, topsoil soil S, pine foliar Mn, pine foliar Ca, topsoil soil B, topsoil bioavailable Mg, topsoil soil total C, topsoil bioavailable Ca, aspen foliar Fe, upper subsoil soil B, upper subsoil bioavailable K, topsoil bioavailable Fe, topsoil bioavailable Pb, topsoil bioavailable Cu, topsoil bioavailable B, topsoil bioavailable Al, pine foliar K, and pine foliar Zn. Based on differences in these nutrients, the reclamation treatment of B/C blended subsoil (BCB) capped with forest floor-mineral mix (FFM) was most similar to the reference upland forest sites. Among treatments capped with peat-mineral mix (PM), a subsoil

BCB horizon thickness of 70 cm was most similar to the reference site, followed by the 120 cm BCB and 30 cm BCB treatments.

Preface

The following thesis is composed of original data generated and analyzed by Jeffrey Ian Hogberg, with no data having been published at the time of submission. Data from Chapter 2 "Evaluating Foliar Nutrient Concentration as an Indicator of Ecosystem Function in Natural and Reclaimed Soils in the Alberta Oil Sands Region" was presented at the following conferences: poster at the 2016 Alberta Soil Science Workshop, Grande Prairie, Alberta, Canada; oral presentation at the 2016 Soil Science Society of America Annual Meeting, Phoenix, Arizona, USA. Journal articles utilizing the data from Chapters 2 and 3 are forthcoming, pending review.

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Dedicated to all of those who have had enduring confidence in my ability to succeed at this endeavor, even when I was certain that I would be unable to – supervisors, friends, and family

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List of Abbreviations and Terms

AESRD	Alberta Environment and Sustainable Resource Development
ANOVA	Analysis of variance
Al	Aluminum
AOSR	Athabasca Oil Sands Region
ASCS	Aurora Soil Capping Study
Avg	Mean average
B	Boron
BCB	Blend of salvaged subsoil (B and C horizons) from a/b ecosites
$BaCl_2$	Barium chloride
C	Carbon
Ca	Calcium
$CaCl_2$	Calcium chloride
Cd	Cadmium
CEC	Cation exchange capacity
CEMA	Cumulative Environmental Management Association
CIF	Criteria and Indicators Framework for Oil Sands Mine Reclamation Certification
CO_2	Carbon dioxide
Cu	Copper
NPI(V)	Nutrient Profile Index (Value)
EC	Electrical conductivity
EFSI	Ecosystem Functional Similarity Index
EPEA	Alberta's *Environmental Protection and Enhancement Act*
Fe	Iron
FFM	Forest floor-mineral mix
FFM_{Fert}	Forest floor-mineral mix that has undergone fertilization treatment

H_2O	Water
HCl	Hydrochloric acid
HNO_3	Nitric acid
ICP-OES	Inductively Coupled Plasma Optical Emission Spectroscopy
K	Potassium
LCCS	Land Capability Classification System
LFH	Organic litter layer soil surface horizons (litter, fibric/fermented, humus) in natural upland forest floor, atop mineral soil In Figure 3.1, refers to a blend of upland site LFH materials and the upper 10 – 20 cm of mineral soil used to make FFM capping material; See also: FFM
LS	Lower subsoil, the lowest layer of soil material. Analogous to the C horizon of natural soils. Alternatively, the loamy sand soil textural class
Mg	Magnesium
Mn	Manganese
Mo	Molybdenum
MRPP	Multiple response permutation procedures
N	Nitrogen
Na	Sodium
NH_4^+	Ammonium
Ni	Nickel
NMS	Non-metric multidimensional scaling
NO_3^-	Nitrate
OB	Overburden
P	Phosphorus
Pb	Lead
PC	Principal Component (typically with a numerical value, indicating which principal component it represents, eg: PC1, PC2, etc.)

PCA	Principal Components Analysis
PM	Peat-mineral mix; peat mix
PM_{Fert}	Peat-mineral mix that has undergone fertilization treatment
PRS Probe	Plant Root Simulator (PRS™) Probe (Western Ag Innovations Inc., Saskatoon, SK, Canada). A specific brand of Ion Exchange Resin Probe.
Ref	See Reference
Reference	Natural reference sites used for comparison with the reclaimed sites in these studies
Rnd	Random
S	Sulphur
	Alternatively, the sand soil textural class
SAR	Sodium Absorption Ratio
SCL	Sandy clay loam soil textural class
TC	Total carbon (determined by combustion analysis)
TN	Total nitrogen (determined by combustion analysis)
TOC	Total organic carbon
TS	Topsoil, the upper layer of soil material. Analogous to the A horizon of natural soils
Trt	Treatment
US	Upper subsoil, the middle layer of soil material. Analogous to the B horizon of natural soils
Zn	Zinc

1.0 Disturbance, Reclamation, and Reintegration – The Athabasca Oil Sands Region

1.1 – Resource Extraction and Ecosystem Disturbance

With rare exceptions, extraction or harvesting of the vast majority of mineral commodities requires at least some level of disruption of tracts of wilderness and the ecosystems that form within them. These anthropogenic disturbances can range from localized removal of some portion of habitat to nearly complete desolation on a landscape scale. As the population of humanity grows and the standard of living improves globally, the demand for resources will continue to increase. Technological advances can mitigate some of the demands, but such advances may also simply change demand for one resource into demand for another. Damaged ecosystems translate not only into lost capacity to support natural biological communities, but also decreased land productivity and reduced ecosystem services provided within the disturbed area; including hydrological, geological, climatic, and nutrient cycling processes.

Open-pit mining can create an environmental disturbance that is both broad and deep, spanning a large surface area and involving excavation of large amounts of material, which must be transported and stored elsewhere. Unlike mines that tunnel deep underground to provide access to a seam, vein, or lode, with a somewhat minimized surface disturbance for the entrance, open-pit mines remove excess material from above the desired commodity, termed spoil or overburden, across a vast surface area. Open-pit mining and quarrying has been used historically for harvesting of metal ores, gemstones, building materials, coal, and many other resources. More recently, open-pit mines have been used to collect raw oil sands material for bitumen refinement; perhaps most extensively in the Athabasca oil sands region (AOSR) of Alberta. Unlike relatively rocky or barren mountainous areas where open-pit mining is more common and soil development is much more limited, the mines in the AOSR require the removal of boreal forest ecosystems, which consist of upland mixedwoods and lowland bogs and fens (Natural Regions Committee 2006). These forest ecosystems support a higher density of biomass than mountainous regions (Natural Regions Committee 2006), much of which is perennial and can take years to decades to reach maturity, which means that regeneration of a landscape to the conditions that existed prior to industrial disturbance is a slow process. According to estimates, there are over 89,000 ha of land affected by oil sands mining in the AOSR, a number that only stands to increase as more land is subject to mining (Figure 1.1) (Canadian Association of Petroleum Producers 2016). New technologies, such as steam-assisted gravity drainage, have allowed movement away from this large-scale destructive mining practice. This means that new exploration and development of resources will have a decidedly lower impact on surface environments and ecosystems. However, many open-pit mines are still in operation in the AOSR, and all disturbances, large or small-scale, leave a disturbed area that must be reintegrated with the surrounding environment at some point.

The dramatic disturbance caused by open-pit mining differs greatly from all but the most extreme of natural disturbances. The complete removal of vegetation and other organisms, soil, and even some geological strata, especially over a broad area, reduces the land's capability to that which might be seen following a volcanic eruption or glacial retreat. More common periodic disturbances, such as wildfires or windstorms, may still exhibit tremendous destructive capacity; however, they will leave legacies of previous ecosystems, in terms of both a substrate legacy, such as charcoal or decaying organic matter, and a biological legacy, in the remaining plant propagules and soil microbial community. These two legacies provide the materials and the early actors to restart the regeneration process, which is referred to in these situations as a secondary succession. Primary succession, by contrast, occurs as plants and other species slowly encroach from the edges of an area devoid of biological and substrate legacies. Open pit mining leaves such a disturbance. As evidenced by the fact that ecosystems do eventually form in areas following these kinds of disturbance, it stands to reason that, given enough time, areas impacted by open-pit mining will likewise regenerate some sort of ecosystem. However, the unmodified sub-surface parent geological materials that remain following overburden removal are highly unsuitable for plant growth compared to actual soil (Bradshaw 1997), and leaving the restoration process to nature alone is undesirable in terms of the costs of allowing the land to remain unproductive, ecologically and economically, for the length of time that it would take for the regeneration to occur, as well as the uncertainty of the trajectory of the ecosystem regeneration.

Actively undertaking land reclamation allows land managers to influence both the rate of ecosystem restoration and the trajectory that the ecosystem will follow as it regenerates. This will allow land managers to not only accelerate the rate at which regeneration occurs, but also allow them to specifically target the previously-existing ecosystem as an end-goal, rather than just accepting whatever ecosystem forms as regeneration occurs (Rowland et al. 2009). Land reclamation's end goal is to improve condition of disturbed land to a state where it is more economically and ecologically productive (Jacobs et al. 2015). In the context of reclaiming to a wildland ecosystem, this generally means restoring a self-sustaining, regionally appropriate ecosystem. Land reclamation can involve a number of different procedures, including: creating new topography to replace landforms that were removed during mining operations – landscape engineering; removing any potential chemical contaminants at a site – remediation; actively planting desired species of plants from the original native ecosystem and providing them with necessary nutrients for them to start growing – revegetation; and active monitoring and corrective action as necessary to ensure that any problems that arise do not derail the overall course towards reclaiming the site to its end land use. The most important part of reclamation, in terms of re-establishing an ecosystem, may well be the replacement of the soil. Soil is the foundation of any terrestrial ecosystem, providing the home and growth medium for any plants and soil microorganisms that anchor ecosystem nutrient cycling, as well as a large array of other organisms, such as micro-, meso-, and macrofauna. Soil also provides storage of both water and organic matter, as well as facilitating the latter's decomposition and recycling into accessible nutrient pools (Séré et al. 2008). As the interface between the atmosphere, hydrosphere, lithosphere, and biosphere, soil is crucial to any terrestrial ecosystem, and replacing it should be among the top priorities for any land managers implementing reclamation plans.

1.2 – Reclamation of Disturbed Ecosystems

1.2.1 – Regulatory Requirements

The regulation of natural resources extraction is generally within the purview of the provincial governments in Canada. It is generally only situations where large projects with consequences that extend beyond one province's political borders that Environment Canada, the federal regulation body, may require environmental impact assessments prior to industrial activity. The federal government also regulates any projects impacting navigable or fish-bearing waters. Provincial jurisdiction over natural resources was first granted as part of the *Natural Resources Transfer Agreements* in 1930. It was in 1963 that the Alberta government recognized that formal protection of the environment from degradation was in the public's best interests, and the *Surface Reclamation Act* was implemented (Powter et al. 2012). The act was repealed in

1973 and replaced with *Land Surface Conservation and Reclamation Act*. This act was amended and built upon until 1993, when the *Environmental Protection and Enhancement Act* (EPEA) re-codified the myriad updates to earlier legislation. Under EPEA, unlike earlier incarnations of the legislation, a regulation was finally added to formally define remediation and reclamation standards, rather than the previous vague “satisfactory condition” requirements. The *Conservation and Reclamation Regulation* formally defined the objective of land reclamation to be restoring the land to “equivalent land capability” to that which existed prior to the disturbance (Powter et al. 2012). This became the guiding principle that directs the efforts of land reclamation that occurs in Alberta. Under this principle, reclaimed land in the AOSR must also support locally common boreal species and integrate sensibly with the surrounding landscape, as well as support various activities at a similar capacity as existed prior to disturbance, though the specific activities need not necessarily be identical (Oil Sands Research and Information Network 2011; AESRD 2013). While a formal standard has now been introduced into the legislation, creating a specific objective for land reclamation, a similar problem to the earlier acts still exists; in that, nowhere within the legislation is what the standard of “equivalent land capability” actually means rigidly defined (Oil Sands Research and Information Network 2011).

1.2.2 – The Boreal Forest Ecoregion

The Boreal Forest Natural Region spans a vast area in northern Canada; as such, it contains many subregions that reflect local variations in climate and other environmental properties. Oil sands mining disturbances in the AOSR are most common within the Central Mixedwood Subregion (Natural Regions Committee 2006). Fort McMurray, the central location for oil sands mining operations in the AOSR, has an annual average (1981 – 2010) of 418.6 mm of precipitation; with a mean average daily temperature of 1 °C (Environment Canada 2016).

The highest summer temperatures occur in July and August, when average daily temperatures are, respectively, 17.1 °C and 15.4 °C.

Native tree species found on well-drained upland topographical positions include jack pine (*Pinus banksiana* Lamb.) with some trembling aspen (*Populus tremuloides* Michx.). These well-drained, sandier soils are generally of the Brunisolic soil order, according to the Canadian System of Soil Classification. Soils that have a more mesic moisture regime and finer-textures generally fall into the Luvisolic soil order, and support stands of trembling aspen and white spruce (*Picea glauca* Moensch) (Beckingham and Archibald 1996). The boreal ecoregion contains numerous interspersed wetlands, some of which are created or altered by the activity of beavers (*Castor canadensis* Kuhl). Under saturated conditions, wetlands develop thick layers of decaying peat mosses (primarily *Sphagnum* spp.). Tree species that grow in Organic and Gleysolic soils of the lowlands are black spruce (*Picea mariana* Mill.), tamarack (*Larix laricina* Du Roi), and various birch varieties (*Betula* spp.). The parent geological material of the boreal plains of northern Alberta is dominated by Cretaceous shales, derived from marine sediment (Natural Regions Committee 2006). This means that soils in the region have a legacy of elevated levels of salinity, which can impact their effectiveness in reclamation operations. The natural vegetation community, and individual species' adaptations to this material, such as rooting depths, are reflective of these limitations (Purdy et al. 2005).

In addition to the broader climatic, topographical, and parent geological variability within the boreal region, the composition of the ecological community is also influenced by periodic natural disturbances. Minor disturbances are caused by beaver activity, windfall, and pathogens. The spread of the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and the associated blue stain fungus pathogen (*Grosmannia clavigera* (Rob.-Jeffr. & R.W. Davidson) Zipfel, Z.W.

de Beer & M.J. Wingf.), increasingly a concern for the western portion of Alberta, has not yet spread as far east as the AOSR. The primary source of disturbance in the boreal forest ecoregion is fire, which can impact ecosystems on a landscape scale. Fire frequency and intensity is variable; however, it is rare for boreal landscapes to remain unburnt past 200 years of age. Only 5 – 10 % of boreal landscapes remain untouched by fire for this length of time (Johnson et al. 1995). As such, the ecological community in the boreal ecosystem is adapted to recover from these fire events and the conditions that follow them, such as altered temperature ranges, increased light due to openings in the canopy, altered soil dynamics, and the addition of charred organic matter to the soil (Zackrisson et al. 1996). In addition to these natural disturbances, resource extraction, including timber harvesting and petroleum exploration and mining, are becoming increasingly important anthropogenic disturbances in the AOSR. Unlike wildfire disturbances, there has not been sufficient opportunity for evolutionary attenuation by native species to these impacts. Recovery from these disturbances presents an opportunity for novel ecosystems to develop; however, there is some debate as to whether these novel ecosystems meet the legal requirements of "equivalent land capability" when it comes to defining reclamation success.

1.2.3 – Reclamation Practices

1.2.3.1 – Topsoil Materials

As the parent geological material that the overburden is composed of has undergone limited development compared to the soil removed from mining sites, it is generally unsuitable as a growth medium for plants and other soil organisms (Bradshaw 1997). This means that replacement topsoil must be placed at the site, on top of whatever material is being used to fill in the voids left by mining, whether overburden, tailings, or other waste or filler materials. As topsoil is a valuable resource, lease holders are required to salvage suitable topsoil materials during initial site preparation, following vegetation clearing. This is directed by provincial authorities and modified for each individual lease agreement, in order to account for and accommodate different conditions at each site. Depending on the topographic position of the topsoil prior to salvaging, it will fall into one of two broad categories. Both categories of salvaged topsoil are generally mixed with subsurface mineral soil in order to improve their suitability as topsoil under the new conditions of the sites undergoing reclamation and to spread the beneficial properties of the salvaged topsoil materials over a greater area than their limited quantities would otherwise allow.

Soils salvaged from lowland bogs, fens, and other wetlands are rich in decaying sphagnum peat. Mixing this peat-rich soil with other mineral soil creates the peat-mineral mixture (PM) reclamation topsoil. The exact ratio of peat to mineral soil is not necessarily consistent, as debate still exists as to what the optimal combination to facilitate revegetation is. Additionally, as peat has such a low bulk density compared to mineral soil, mixture of the two materials can be inconsistent, leading to patches of nearly pure mineral soil interspersed within otherwise pure sheets of peat as material is placed on the reclamation site. The peat material, being composed of decaying organic matter, is generally quite rich in C and N, though may have some deficiencies in nutrients derived from mineral sources, such as P and K. From its high organic matter content, PM also tends to have a very high water holding capacity, and most of this water will be available for plant growth (McMillan et al. 2007; MacDonald et al. 2012; MacDonald et al. 2015a; Pinno and Errington 2015). The relative abundance of peat soils in the AOSR has meant that PM is a preferred topsoil material for reclamation, especially as further development of the AOSR leads to an increased supply of material for use in reclamation.

The second topsoil material is one that has received more prominence recently. The salvage of upland forest floor topsoil, including the LFH layer and A horizon, to an approximate depth of 10 – 20 cm, generates supplies of a topsoil material that is similar to the properties of the reclamation target ecosystem's topsoil. This forest floor-mineral mix (FFM) can preserve the propagule bank that aids in the recovery of unmodified ecosystems from natural disturbances, such as fire. The properties of these soils are more similar to other upland forest soils than PM topsoil, often containing a naturally greater proportion of mineral soil components, and a nutrient balance that has a lower C and N content, but greater P and K content (McMillan et al. 2007; Mackenzie and Naeth 2010; MacDonald et al. 2015b). The LFH layer is admixed with the mineral A horizon and other mineral soil during the salvaging and storage process, so this material is no longer the equivalent of a natural, purely organic LFH. Despite this, it is not uncommon to see the name "LFH" applied to a FFM soil within industry terminology (See Figure 3.1 for an example). As upland forests can develop on different textures of soil, it is common practice to separate coarse-textured FFM from fine-textured FFM, and use these reclamation materials in similarly-textured areas, so that reclaimed ecosystems will better reintegrate with surrounding natural ecosystems. Upland forests occupy less of the newly-disturbed area in the AOSR than lowland wetlands, and topsoil salvaged from these forests is much shallower than the peat than can typically be recovered from those wetlands, meaning that the supply of FFM is more limited than that of PM (Mackenzie and Naeth 2010).

1.2.3.2 – Soil Placement

While soils can be stockpiled following salvage, it is generally more beneficial to directly place these materials on to reclamation sites (Mackenzie 2013). Soil biogeochemical conditions degrade over the course of stockpiling, and propagule viability decreases. The term progressive reclamation is used to describe the process of initiating reclamation immediately following mine decommissioning. Following any landscape engineering, such as contouring, newly salvaged subsoil and topsoil material from mine expansion elsewhere is placed and spread to desired thicknesses on reclamation sites. This practice also has the advantage of reduced handling and storage costs for salvaged soils, in addition to documented ecological benefits. This however, is an ideal situation that is not always possible due to the nature and timing of mine expansions; fresh donor sites are not always available when reclamation begins, and ready receptor sites are not always available when salvage occurs. This means that stockpiling of salvaged soil materials is more likely to be the norm than the exception for reclamation.

Soil depth is not consistent in natural ecosystems, due to differences in a variety of factors at different sites, including topography, climate, and vegetation. As such, a mandated standard application depth would be impractical, and capping depth requirements are determined on an individual basis for each site. The largest consideration for soil capping depth is to create a soil environment that provides the greatest advantages for plant regeneration on the site (Kessler et al. 2010). This includes properly separating the rooting zone of plants from deleterious substances that exist in deeper landform materials, and providing sufficient nutrient and water holding capacities to support the variety of appropriate local boreal species. Landforms are often constructed using waste materials from the mining process, such as overburden or tailings sand. The more detrimental these construction materials are for plant growth, the deeper the layer of capping material should be. Another consideration that should be made during soil placement is that abrupt textural differences between different soil layers could interfere with root penetration and vertical water movement (Naeth et al. 2011; Jung et al. 2014).

If topsoil material that is placed on reclamation sites is deficient in soil nutrients, amendments, such as fertilizer, can be used to improve conditions for initial plant establishment and growth. Application rates are generally greatest during the first growing season, though repeat applications may be used (Pinno et al. 2012). Ultimately, as the goal is to create a self-sustaining ecosystem, fertilization cannot be relied upon to make up for nutrient shortfalls indefinitely. Fertilization is generally accomplished using farming equipment, if the reclamation site is accessible and traversable with such machinery or with aerial application where such equipment is impractical or impossible to employ. There are concerns that fertilizer overuse can create a surplus of nutrients that invasive species will take advantage of, at the expense of the current and future desired native species community (Davis et al. 2000). Another amendment that has become increasingly common is coarse woody debris, which helps replace part of the substrate legacy that would be found on a natural ecosite. The coarse woody debris increases soil carbon and other nutrients as the material decomposes, and creates variability in surface microtopography. This increase in surface roughness creates a wider assortment of niches for plant establishment and helps limit overland water flow, which decreases erosion and increases water infiltration into the soil (Brown and Naeth 2014).

1.2.3.3 – Revegetation

In order to stabilize soil in the first growing season following placement, nurse crops, such as agricultural barley (*Hordeum vulgare*), are often planted. As with fertilizer application, site accessibility can determine whether it is aerially broadcast or seeding with standard farming equipment. Seeding occurs late in the growing season so that the annual species does not reach maturity by the time of the first killer frost. This ensures that the non-native grass species does not spread from the reclamation site or return in the following growing season, allowing native plant species to instead dominate the reclamation site.

Once the site has been adequately prepared, nursery-grown tree seedlings are planted as recommended by the provincial government's *Guidelines for Reclamation to Forest Vegetation in the Athabasca Oils Sands Region* (Alberta Environment 2010). The species planted depend on the nature of the target ecosite, and goals for the end-land use by any site lease holders. Most commonly, trembling aspen, jack pine, and white spruce are planted. Other desired species, such as shrubs may be similarly planted at this time, and seeding of understory herbaceous species may also occur. Migration of plants from surrounding areas without human assistance will also occur over development of the site, through mechanisms such as wind dispersal and animal transport. While advantageous for regeneration of native species, these can also be vectors for site invasion by non-native weedy species.

1.3 – Reclamation Challenges

1.3.1 – Salinity

Due to the salinity of the parent geological material in the region, some soils may have high salinity and sodicity issues. Elevated electrical conductivity (EC) and sodium adsorption ratios (SAR) of these soils may exceed acceptable levels permitted by regulatory guidelines. This may mean that an adjustment of plant community expectations may be necessary for reclamation occurring on saline sites (Purdy et al. 2005).

1.3.2 – Stockpiled Soil

As discussed in section 1.2.3.2, stockpiling can lead to a degradation of properties that make salvaged topsoil a valuable resource for reclamation. It is also inevitable that stockpiling of soils will occur due to differences in rates of expansion and reclamation of mining operations in the AOSR. When topsoils are stored in large heaps instead of spread relatively thinly across a landscape, their internal conditions change, including soil temperature, water content and movement through soil, gas content, redox conditions and soil structure and stability (Mackenzie 2013). Additionally, the valuable propagule banks will degrade rapidly over the course of a stockpile's lifetime, with the bank being mostly lost within the first 16 months of stockpiling. While these issues can be somewhat mitigated with use of smaller stockpiles, and planting stockpiles with desirable species, this incurs additional storage and maintenance costs.

1.3.3 – Plant Community

Traditional practices used to stabilize soil and prepare it for revegetation, nurse crops and fertilization, can interfere with the natural progression of plant community regeneration during reclamation. Fertilization creates a surplus of nutrients that provides an opportunity for invasive species to colonize the site, until the nutrient uptake rate better matches the available nutrient supply (Davis et al. 2000). As most native boreal species are adapted to lower-levels of available nutrients, and not specifically to take advantage of pulses of nutrients, weedy species are likely to outcompete them for these freely available nutrients (Pinno and Errington 2015). Additionally, if those surplus nutrients are not being readily taken up by plants, native or invasive, they may be lost from the site as runoff or as atmospheric or leaching losses of N. At best, this represents wasted effort and costs; at worse, this can cause eutrophication problems in downstream bodies of water. Nurse crops, in addition to stabilizing soils, can mitigate some of these concerns by taking up excess nutrients and converting them to organic matter that will be recycled into the soil at the site. However, nurse crops, typically introduced agricultural species, often have potent growth rates and can outcompete and exclude any native vegetation that would have otherwise colonized the site during that initial growing season. Additionally, non-native plants, whether they were planted or invaded the site, will shape the ecosite to suit their own needs as best as they can (Bayfield 1996). These introduced species may persist in the ecosystem for an extended period of time, which can alter the course of the plant community's regeneration trajectory.

1.3.4 – Scarcity of Reclamation Materials and Optimum Usage

Perhaps one of the most pressing issues plaguing oil sands reclamation is the sheer scope of the disturbance. Use of open pit mining has created a vast area of severely degraded environments and ecosystems (Figure 1.1). The amount of materials needed to reclaim these afflicted areas already imposes a huge economic cost. New oil exploration and extraction continues, creating new disturbances; though modern techniques create less of a surface disturbance, which is overall better for affected ecosystems. Paradoxically, this does exacerbate a challenge already faced by progressive reclamation of disturbed areas, as less fresh soil material that can be used for reclamation is salvaged. Furthermore, while most disturbed areas were wetlands, most reclamation efforts are focused on creating upland topography, partially because the dynamics of upland ecosystems are easier to re-create, and partly because upland forests are more economically valuable for lease holders than lowland wetlands. This means that, ideally, upland reconstructed soils would be capped with soil materials taken from upland positions, such as FFM. However, this material was already in more limited supply than PM due the relatively smaller size of the areas it was harvested from and shallower depths that it can be salvaged from, compared to PM. As this material is already valuable and in limited supply, it is important that it is being used as efficiently as possible during reclamation, so that its beneficial properties can be applied to as much reclamation area as is sustainable. Even though PM is in greater supply, there still is a limit to how much of this material is available too, and so it likewise must be used efficiently. The same applies to clean subsoil materials that are used in capping of landforms. There is no consensus for what represents the optimum and minimum soil capping depths for materials, research is ongoing as to the best depths to establish sustainable soil moisture and nutrient regimes (Howell 2015). Shallower application depths would allow material to be used across a greater surface area, but may not provide adequate buffer zones from deleterious substances in some waste materials used to construct landforms, or provide enough storage capacity for water and soil nutrients.

1.4 – Challenges in Determining Reclamation Success

1.4.1 – Complexity

With these potential pitfalls in mind, it is critical that any reclamation declared successful not encounter unforeseen problems in the future that would render such a proclamation invalid. Certainly, land managers have a vested interest in having their reclamation declared successful, as they are responsible for upkeep of the land until certification is earned and it can be transferred back to the care of the province and any timber-rights leaseholders. However, the difficulty with declaring a reclamation operation successful is that natural systems are inherently complex. There are many different dynamics to consider, including: nutrient cycling, hydrology, vegetation community, microbial community, faunal community, soil biogeochemistry, and many other considerations. All of these pieces form part of the greater ecosystem equation individually, but they are themselves intertwined in complex ways. This means that changes to one aspect of an ecosystem do not exist in isolation, and will have ripple effects throughout the entire ecosystem. Additionally, there are not necessarily clear boundaries where one ecosystem's influence halts and another ecosystem's begins. While in theory it sounds straightforward enough to define boundaries in terms of the lease agreement, as part of the requirement that reclaimed sites integrate sensibly with the surrounding landscape, considerations must be made for how changes inside the boundaries affect the area outside of the boundary, and for how changes outside of those boundaries likewise affect the site within.

In nature, these systems may take decades or longer to reach a state of ecosystem equilibrium, with several smaller equilibrium steps occurring throughout the process (Figure 1.2). This raises the question of when exactly it would be acceptable to declare a system as reclaimed, and how this stage being reached would be recognized. Furthermore, a return to a functioning, self-sustaining natural ecosystem does not necessarily guarantee a return to an identical ecosystem to that which existed prior to disturbance (Figure 1.3). Does this mean that such a reclaimed ecosystem does not satisfy the regulatory requirements of "equivalent land capability"? These two key questions are both complicated, and require information that is precise, reliable, and quantifiable if they are to be properly answered.

1.4.2 – A Tangible Definition of Success

As mentioned in Section 1.2.1, legislation in Alberta has generally not mandated specific values by which reclamation success can be measured. This is because not only would it be impractical to try to consider the range of possibilities that can be seen throughout the highly variable natural landscape, it is also unclear as to whether specific values for properties such as nutrient concentrations have the same direct implications in wildland ecosystems as they do for managed ecosystems, such as those found in agricultural operations. The Land Capability Classification System (LCCS) attempted to provide a tool for assessment of site productivity based on soil nutrient and moisture regimes (Cumulative Environmental Management Association [CEMA] 2006); however, calculations used by the LCCS oversimplified the complexity of soil health, which led to a lack of ability to clearly distinguish between successful and unsuccessful reclamations. Furthermore, the LCCS was biased towards nutrient and moisture demands of commercial timber species, while other ecosystem components were neglected by the simplification of information. This exemplifies the challenges of creating a quantification system for assessing successful development of ecosystems across such a vast and highly variable area.

1.4.3 – Criteria and Indicators

Building off of the LCCS, CEMA worked with researchers, industrial partners, and regulatory authorities to develop the *Criterion and Indicators Framework for Oil Sands Mine Reclamation Certification* (CIF), which attempted to expand to incorporate other aspects of ecosystem function as part of the determination of reclamation success; the CIF is incorporated into the reclamation policy of regulatory authorities in Alberta (Alberta Environment and Sustainable Resource Development [AESRD] 2013). This framework collected and codified many techniques that were already being used or examined for potential use in measuring and monitoring of reclamation success throughout the AOSR. Many proposed indicators listed within are still undergoing evaluation to determine how they would be implemented as part of a formal policy.

An indicator, simply, is some sort of information that can be used as a proxy for another type of information. In an ecological context, indicators are generally properties of an ecosystem that are easy to measure and compare that have strong correlations with different properties that might be more broadly informative as to the condition of an ecosystem, but are themselves more complicated to measure and compare. National Research Council (2000) list three key features of good indicators; a good indicator: quantifies information and makes the importance of said information readily apparent; simplifies complex information and makes communication of that information easier; and is simpler and more cost-effective to assess and compare than the processes or information that it represents. Alone, the information provided by indicators does not necessarily achieve goals, which must also be clearly defined. Combined with appropriate policies and strategies, however, indicators can be means by which ecological objectives are achieved (Failing and Gregory 2003; AESRD 2013). For instance, indicators can be powerful tools for developing and implementing strategies for managed ecosystems, when paired with specific thresholds and clear options for decision-making based on whether or not these thresholds are met (Failing and Gregory 2003). Without clear thresholds and specific guidelines for corrective action if thresholds are not being met, indicators become little more than an inventory of properties. The CIF itself lists three considerations used to select indicators that would be used in the CIF: Effectiveness, soundness, and practicality. These three categories are further broken down into eleven sub-categories total.

For effectiveness, there are two sub-categories: Usefulness/definitiveness and distinctness in acceptable/unacceptable conditions. These categories align with Failing and Gregory (2003)'s guideline that the indicator is linked to the overall end goal by attaching specific conditions to the indicator, and ensuring that those conditions are relevant to the property that the indicator is acting as a proxy for. Failure to take this step often leads to long lists of properties that provide somewhat useful pieces of information, but do not actually lead to any meaningful conclusion (Failing and Gregory 2003).

The soundness category is divided into five sub-categories. The first of these requires a strong scientific backing for the indicator, or wide acceptance of the principles. Evidence-based policy is useful in that it can be changed in light of new findings. However, wide acceptance of older methods can be difficult to overcome, even with strong evidence. The next categories assess whether or not the indicator is measurable, that measurement is replicable, and whether a common measurement method exists for the indicator. As part of having specific values linked to failure conditions, those values must be measurable for all sites undergoing measurement. The importance of having a common measurement technique is also important within a management context, to ensure that the data being used to judge the success or failure of a site is in a standardized form and easily understood by all parties involved (National Research Council 2000; Failing and Gregory 2003). The final sub-category for soundness is responsiveness of the measured indicator. A large amount of statistical background noise that occurs during monitoring can make it difficult to assess whether or not a result is meaningful.

The four sub-categories of practicality relate more to the challenges of implementing a potentially useful indicator. The subcategories of technical feasibility, functioning with existing techniques and technology, and cost-effectiveness ensure that the indicator is readily-usable to those who would benefit from it. It is an important part of the management context to consider whether there would be barriers to widespread adoption of a standardized indicator, such as cost or gaps in technical expertise (Failing and Gregory 2003). The final sub-category simply asks whether or not the indicator is specifically tied with the management end-goal of receiving certification of reclamation under Alberta's *Environmental Protection and Enhancement Act* (EPEA). Again, this aligns with Failing and Gregory (2003)'s recommendation that a specific endpoint be in mind when designing and selecting indicators.

1.4.4 – Problems with Current Indicators for Nutrient Cycling

The first objective listed by the CIF is that "Reclaimed landscapes are established that support natural ecosystem functions". This goal focuses mainly on the construction of reclaimed landforms and the materials used in their construction. Objective 2 of the CIF is stated as

“Natural ecosystem functions are established on the reclaimed landscape”. This objective narrows the focus of indicators mainly to a polygon level, focusing on individual ecosites, rather than landscape as a whole. For the criterion of “Nutrient cycling is established on the reclaimed ecosystem” the trend in Foliar Nutrients is the major indicator (AESRD 2013). Soil Macronutrients is also listed as an indicator, but this indicator only focuses on soil N and P, and no other nutrients. When used in situations that control for their shortcomings (National Research Council 2000), foliar nutrient concentrations can be good indicators. For example, foliar nutrient concentrations are commonly used in agricultural ecosystems to monitor the health of crop species (Smethurst 2000). The problems with using this indicator as the sole representation of nutrient cycling on a reclamation site, however, are numerous.

First and foremost, nutrient cycling refers to movement of nutrients from one nutrient pool to another, and yet, only one nutrient pool is being monitored under the scope of this indicator for most nutrients. It is tempting to want to use foliar tissue as an indicator, as it is generally easier to collect than soil sample; however, in doing so, important relationships between soil and plant nutrient pools are oversimplified. There are ecosystems where this simplification is not as problematic. For agricultural systems, foliar nutrient concentrations can be easier to measure than rates of biomass accumulation, and well-characterized relationships between nutrient concentration and biomass production exist for many species (de Mello Prado and Caione 2012). Any potential nutrient deficiencies can be used to trigger corrective actions when detected. Communication of the exact nature of deficiencies becomes much easier if values are quantified, as opposed to diagnosing symptoms of potential deficiencies, but requires a well-defined nutrient requirement for that plant species (Ballard and Carter 1986; Walworth and Sumner 1988; de Mello Prado and Caione 2012). Unlike in carefully managed agricultural ecosystems, however, it is unclear whether pre-defined "critical levels" for nutrients are as meaningful in wildland ecosystems.

Unlike in managed agricultural systems, wildland ecosystems consist of numerous plants from multiple species, all competing to acquire nutrients from the same supply found in the soil (Chang et al. 1996; Staples et al. 1999; Pinno and Bélanger 2009; Hu et al. 2014). Rather than a consistent planting density, the plant community will vary depending on availability of nutrients and other resources, such as water and light. Different microsites could have conditions that support entirely different plant communities. This means that the relative competitive ability of target species may vary within each site, and certainly across the landscape. This means that what may be an acceptable level of a nutrient's concentration for one stand would instead indicate a deficiency in another. Additionally, tree species nutrient demands differ with the age of the stand (Wang and Klinka 1997). It is easy to see that maintaining a database of acceptable nutrient concentration values that will have the flexibility to account for these temporal and spatial differences will quickly become unwieldly. Additionally, there would almost certainly be difficulties in ensuring that the appropriate set of reference critical values is being used to judge each individual site. Simplifying the information, such as by examining only selected nutrients or using a broader rule that does not account for site-to-site differences means a loss of discrimination power for the indicator.

1.4.5 – Ecosystem Similarity

Nutrient cycling is only one aspect of ecosystem function, and this same problem will be encountered with any other potential aspects of ecosystem function that would be useful to examine as indicators of reclamation success. Ultimately, the solution to this problem may lie within the guiding principle of land reclamation in the AOSR itself, "equivalent land capability".

Primarily, the equivalent land capability of a reclaimed site is measured in its productive capacity. While resource output is certainly important from an economic standpoint, the concept of equivalent land capability can also be expanded to include the underlying functions that drive an ecosystem. Assuredly, the value of ecosystem services provided by forests, such as hydrological services, biodiversity conservation, and carbon sequestration, extend well beyond merely the value of harvestable timber that they can produce (Ciccarese et al. 2012).

If the goal of reclamation is to establish a self-sufficient ecosystem that integrates smoothly with surrounding, undisturbed areas in the landscape, it follows that such an ecosystem would benefit greatly from having soil, water, and plant community properties that are functionally equivalent to ecosystems that already maintain self-sufficiency under the specific environmental conditions of the region in which the reclamation is occurring (MacDonald et al 2015a). This is somewhat more difficult to assess, primarily due to the complex nature of interactions between biotic and abiotic components within an ecosystem, but also due to the fact that there is less reliable data available for such properties than for the potential yields of well-characterized economic commodities. However, use of a similarity index comparing a reclaimed site to a natural site that has characteristics of the desired end land-use of the reclamation not only overcomes these gaps in knowledge, but also can account for local variations in site conditions that might affect the reliability of general values. Critical foliar nutrient concentration values for wildland ecosystems are already mainly derived from observed average concentrations rather than specific survival or yield thresholds as would be used in agricultural ecosystems (Ballard and Carter 1986; Smethurst 2000). This same principle of "biological benchmarking" could theoretically also be applied to other aspects of ecosystem function. Different analysis techniques can also evaluate relative differences between sites for numerous different properties at the same time and highlight which variables primarily drive differences between these sites (Masto et al. 2008; Rowland et al. 2009; Mukhopadhyay et al. 2014; Howell et al. 2017).

1.5 – Research Outline

Nutrient cycling is an important aspect of the underlying ecosystem processes that drive the return to "equivalent land capability" on sites undergoing reclamation. Previous evidence suggests that foliar nutrient concentrations from one or two species of trees may not be the most reliable indicators for nutrient cycling in belowground nutrient pools (Ballard and Carter 1986; Rorison 1987; Chang et al. 1996; Hu et al. 2014). The belowground nutrient pools include readily bioavailable nutrients in the soil solution and nutrients adsorbed to the soil particles themselves. There is value in examining both of these belowground nutrient pools, as the bioavailable nutrient pool represents a short term nutrient supply for a site, while the soil nutrient pool represents a nutrient supply for the long term (Mengel et al. 1990; Smethurst 2000; Röing et al. 2006). There are other pools of nutrients that exist in wildland ecosystems; perhaps most importantly, the decomposing organic litter surface layer. However, this forest floor litter layer takes 5 to 80 years to develop on reclaimed sites (Preston et al. 2000). Sorenson et al. (2011) found that within the AOSR, reclaimed sites that were 33 years old had yet to reach forest floor layer thickness comparable to natural sites. At the time of sampling, both studied reclamation sites were no more than 3 years old, so sufficient time to establish a litter layer had not yet passed. My research intentions are to contribute to the existing measures of nutrient cycling in oil sands mine reclamation and the approach taken for indicators of said nutrient cycling. As part of this, I will specifically demonstrate that the recommended indicator for nutrient cycling in the AOSR, the concentrations of only select foliar nutrients, is inadequate at properly discerning between successful and unsuccessful ongoing reclamation treatment progression. Much of the underlying rationale for the use of this indicator is rooted in agricultural methods and philosophy, which aim to maximize productivity of a soil, in a situation where output is removed from, rather than recycled back into the ecosystem. Instead of this system, I propose alternative systems of measurement that measure success in terms of outcomes that will help re-establish natural diverse and functional ecosystems found naturally in the region. My first study examines the correlation between nutrient pools for major macronutrients in reclaimed soils on Canadian Natural Resources Ltd.'s (CNRL) Horizon Project and nearby soils disturbed by wildfires. This study examines macronutrients individually and also uses multivariate data analysis to compare data for all nutrients in all pools simultaneously in order to examine one approach to a similarity index. My second study builds on the multivariate approach used to assess sites in the first study; assessing sites at Syncrude Canada's Aurora Soil Capping Study (ASCS) and nearby wildfire-disturbed sites in order to reduce the massive amount of information generated by assessing many nutrients in many nutrient pools, determining a minimum dataset that would clearly show the most crucial differences between natural and reclaimed sites. Ultimately, this research could contribute to the development of new methodologies for assessing reclamation success within the AOSR, improving upon both the LCCS and CIF and approaching a more holistic Ecosystem Functional Similarity Index.

Tables and Figures

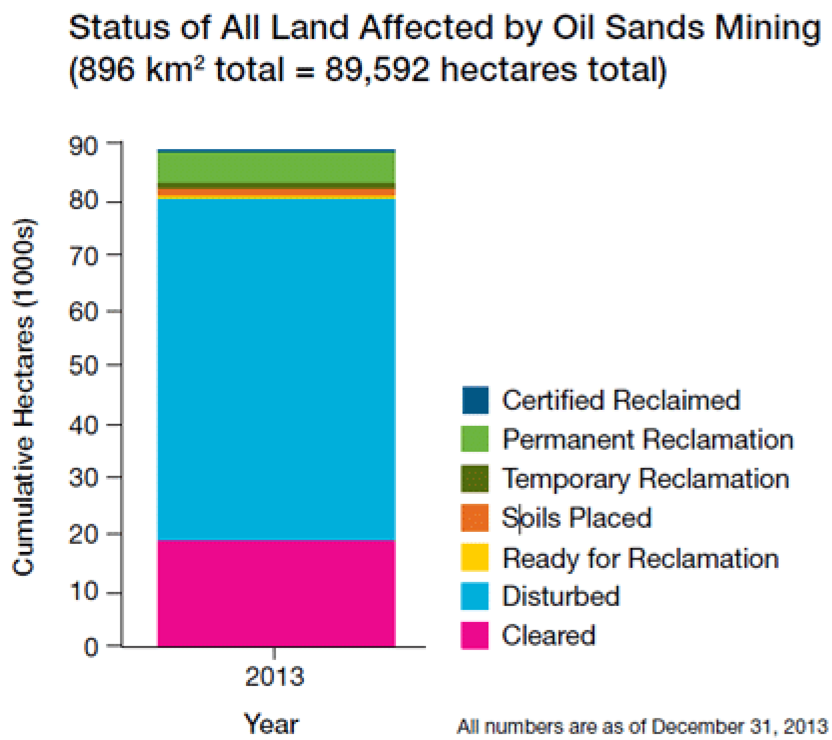


Figure 1.1 – Extent of land disturbed for oil sands mining in Alberta as of December 31, 2013 is estimated to be 896 km^2. From Canadian Association of Petroleum Producers (2016).

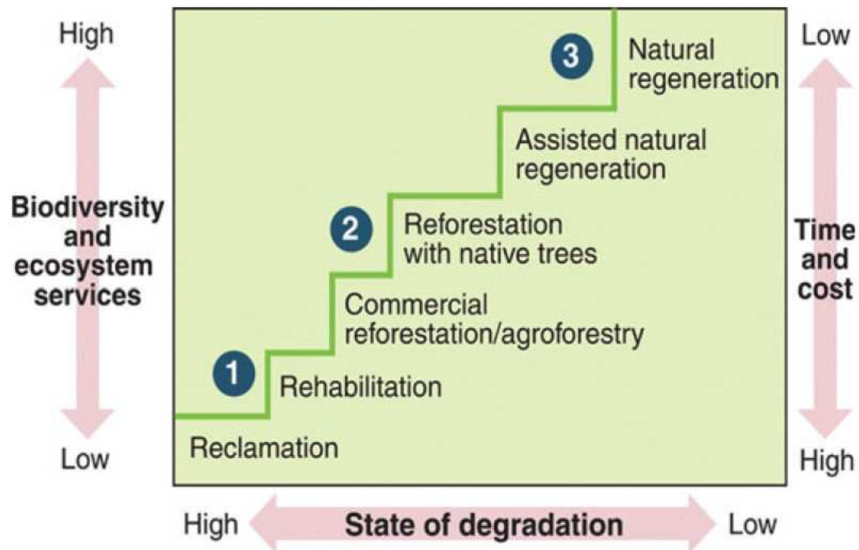


Fig. 1 The restoration staircase. Depending on the state of degradation of an initially forested ecosystem, a range of management approaches can at least partially restore levels of biodiversity and ecosystem services given adequate time (years) and financial investment (capital, infrastructure, and labor). Outcomes of particular restoration approaches are (1) restoration of soil fertility for agricultural or forestry use; (2) production of timber and non-timber forest products; or (3) recovery of biodiversity and ecosystem services. *Source:* Chazdon (2008)

Figure 1.2 – The restoration staircase, representing different stages of reclamation from different stages of degradation and simplified representation of the relative effort it takes to move from degraded states to reclaimed ones. From Ciccarese et al. (2012).

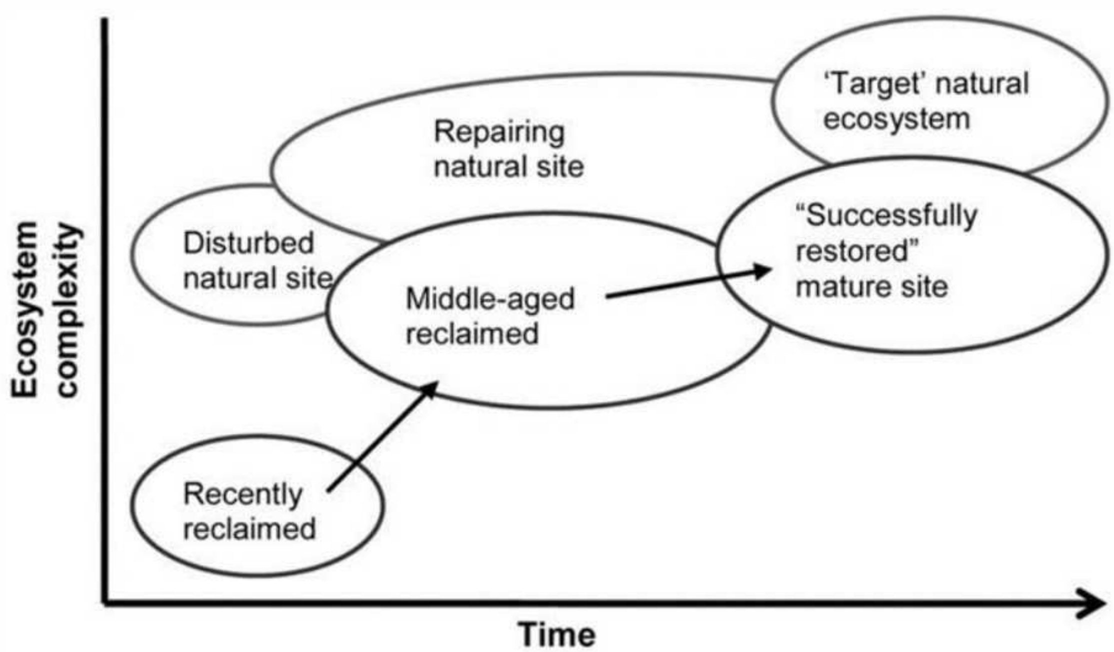


Figure 1.3 – Model representation of ecosystem restoration from consequences of natural and anthropogenic disturbances. Arrows represent a potential restoration trajectory. Note the relatively increased severity of anthropogenic disturbances on ecosystem complexity. From Rowland et al. (2009).

2.0 Evaluating Foliar Nutrient Concentration as an Indicator of Ecosystem Function in Natural and Reclaimed Soils in the Alberta Oil Sands Region

2.1 – Introduction

The use of indicators to represent a more complex process or interaction is a common practice in ecology, due to the scale and complexity of many ecological systems. Alone, indicators can be useful for monitoring these more complex processes and alert those responsible for the monitoring to potential problems and trends (National Research Council 2000). Combined with appropriate policies and strategies, and practical reference values, indicators can be means for achieving ecological objectives (Failing and Gregory 2003; Alberta Environment and Sustainable Resource Development [AESRD] 2013). National Research Council (2000) described three key features of good indicators: The first feature of a good indicator is that information is quantified and its importance is made apparent; the second is that complex information is simplified and communication of that information is made easier by use of that indictor; finally, the indicator is simpler and more cost-effective to measure and monitor than the processes or information that it represents.

Re-establishing natural, self-sustaining ecosystem functions on reclaimed landscapes is one of the major objectives of oil sands reclamation in Alberta. The relevant criterion for nutrient availability on reclaimed sites is "nutrient cycling is established on the reclaimed site" (AESRD 2013). However, nutrients do not remain idle in one easily measured pool until transferring in an orderly, consistent manner to another easily measured pool (Asher 1978; Mengel et al. 1990; Röing et al. 2006). Three nutrient pools that are generally of interest for monitoring soil-plant relations and nutrient cycling are soil nutrients, bioavailable nutrients available in the soil solution, and nutrients contained in vegetation, which is usually represented by foliar nutrients (Ballard and Carter 1986; Röing et al. 2006; Vanguelova and Pitman 2009; Lafleur et al. 2013). It is difficult and impractical to observe the movement of nutrients between these pools directly, especially across an entire ecosystem, but it may be possible to use the concentrations in each pool as an indicator of cycling.

Foliar tissue is generally easier to collect than soil samples, making foliage a potentially useful indicator of other ecosystem functions, such as nutrient cycling. Foliar nutrient concentrations are used to confirm visible nutrient deficiencies (Ballard and Carter 1986; Walworth and Sumner 1988; de Mello Prado and Caione 2012), but this requires a well-defined nutrient requirement for that plant. Measuring foliar nutrient concentrations quantifies the amounts of each nutrient within foliar tissue, which allows comparison to pre-determined “critical levels” of those nutrients within foliar tissue. Communication of the exact nature of deficiencies becomes much easier with these quantified values. These values can be used as thresholds to trigger corrective actions in managed systems. When used in situations that control for their shortcomings (National Research Council 2000), foliar nutrient concentrations are good indicators. Hence, foliar nutrient concentrations are commonly used in agricultural ecosystems to monitor the health of crop species (Smethurst 2000). In these monocultures, where well-characterized relationships between nutrient concentrations and biomass production are established, foliar nutrient concentration is much simpler to measure than rates of biomass production through the growing season and allow for earlier intervention when deficiencies are discovered (de Mello Prado and Caione 2012).

Foliar nutrient concentrations are also used as indicators in wildland ecosystems to monitor the health of species of interest (Ballard and Carter 1986; Wang and Klinka 1997; Chen et al.

1998; Gower et al. 2000; Brockley 2001; Gonzalez et al. 2010). Even when determined using field studies, many critical values for tree species are derived from studies on managed tree plantations, rather than natural ecosystems (Hansen 1994). Additionally, years of agricultural research have generated a more robust database for crop nutritional needs (de Mello Prado and Caione 2012), while wildland ecosystems often rely on average values for nutrient concentrations, rather than species-specific critical levels (Ballard and Carter 1986; Smethurst 2000). Many critical nutrient values are based on greenhouse studies, which may not reflect other limitations faced in a wildland ecosystem, nor will they be sensitive to spatially-linked variability (Ballard and Carter 1986; Rorison 1987). Additionally, as wildland ecosystems are not managed monocultures like tree plantations, there will be competition for all resources by different species (Chang et al. 1996; Staples et al. 1999; Pinno and Bélanger 2009; Hu et al. 2014). Nutrients being transferred from soil nutrient pools would end up in the tissue of multiple plant species, and at different rates; meaning that nutrients in species not being monitored would be unaccounted for (Chang et al. 1996; Hu et al. 2014). Luxury uptake of some nutrients can occur, providing the plant with an emergency stockpile in case of sudden drops in nutrient availability, but nutrient uptake is driven by demands of the plants (Chapin 1980; Rorison 1987; Hu et al. 2014; Sardans et al. 2016). Additionally, nutrients may be translocated from one portion of plant tissue to another, such as the movement of nutrients from old tissue to support the growth of new tissue (Morrow and Timmer 1981). While changes in nutrient availability would eventually affect all plant tissues, there is still the question of whether or not specific differences in foliar concentrations of individual nutrients can be directly attributed to differences in availability of nutrients in the soil and not to other processes or physiological differences.

The soil nutrient pool includes all nutrients present in soil, including nutrients in soil organic matter and nutrients bonded or adsorbed to the surface of soil particles. While these nutrients are not as readily bioavailable as those in the soil solution pool, bonds holding these ions in place can be released slowly over time (Mengel et al. 1990; Röing et al. 2006). This means that the soil nutrient pool represents the long-term nutrient supply for a site, which is important for ecosystems that contain many long-lived perennial plants, such as forests. However, the soil environment is heterogeneous, meaning it is inconsistent even between very close sampling locations (Ballard and Carter 1986; Rorison 1987). Direct sampling of the soil itself to determine nutrient concentrations can have a potentially high cost in terms of time and effort, so there is an incentive to measure an indicator, such as foliar nutrient concentrations, instead.

The bioavailable nutrient pool, on the other hand, consists of nutrients that are dissolved in the water that occupies the pore space within soils and is where plants and other organisms acquire soluble nutrients (Lafleur et al. 2013). Despite the importance of the bioavailable pool to soil fertility and plant productivity, it had traditionally not been used as an indicator prior to the advent of ion exchange resins (Smethurst 2000; Qian and Schoenau 2002; Johnson et al. 2005). Unlike ion extraction methods, ion-exchange membranes take the dynamics of nutrient movement from soils to plants into consideration, including integration of soil water and temperature conditions, providing a more accurate picture of what proportion of nutrients are readily available for uptake by plants under in situ conditions (Qian and Schoenau 2002; Johnson et al. 2005). As plant nutrients are acquired from this nutrient pool, any change in the availability of nutrients from the bioavailable pool would ultimately be reflected by changes in the concentration of nutrients in plant tissues (Smethurst 2000).

Given plants’ capacity to adapt to the highly variable availability of nutrients in the soil (Chapin 1980), it makes sense to determine comparable values from an ecosystem that is most similar, spatially and temporally, to the ecosystem that is the end goal of the reclamation operation (MacDonald et al. 2012). By using natural reference sites that experience similar seasonal fluctuations in temperature, water and nutrient availability, and interactions with local organisms as the reclamation sites, these factors’ influence on the concentrations of soil, bioavailable, and foliar nutrients can be accounted for. This provides a practical set of target values that incorporate the suboptimal field conditions into the expected concentrations, as opposed to reference values that are derived from plants in optimal growth conditions. An emerging philosophy in land reclamation advocates expanding on this idea of comparison to natural benchmark sites to include a simultaneous assessment of multiple properties and assessment of overall similarity (MacDonald et al. 2012; Howell et al. 2017). The use of analysis techniques capable of calculating the influence of multiple variables at the same time can reveal underlying relationships that would not be possible to discover when using conventional techniques that only compare values for individual nutrients.

The specific research questions of this study were: (1) Do foliar nutrient concentrations of plant macronutrients provide an accurate representation of the bioavailable and soil nutrient pools? (2) Is this foliar nutrient indicator consistent and sensitive enough for use in different soil types and treatments, including both natural and reclaimed ecosystems? (3) Can a similarity metric and multivariate analysis, rather than critical values for individual nutrients, provide a more meaningful assessment of key differences between ecosystems?

2.2 – Methods

2.2.1 – Study Area and Experimental Design

The study was conducted at an oil sands mine north of Fort McMurray, Alberta, Canada (57° 21' 7" N 111° 49' 49" W). This region features a natural ecosystem composed primarily of boreal mixedwood forests with major upland tree species of trembling aspen (*Populus tremuloides* Michx.) and white spruce (*Picea glauca* (Moench) Voss) on fine-textured soils, with jack pine (*Pinus banksiana* Lamb.) dominating on coarse-textured soils, and major lowland species of black spruce (*Picea mariana* (Mill.)), birch (*Betula* spp. L.), and tamarack (*Larix laricina* (Du Roi) K. Koch) (Natural Regions Committee 2006). Medium- and fine-textured upland soils are dominated by Gray Luvisols, coarse-textured upland soils are dominated by Brunisols, and poorly-drained lowland soils are dominated by Organic soil types (Natural Regions Committee 2006). Upland areas are generally dominated by aspen and spruce “d” ecosites (Beckingham and Archibald 1996). This region has a continental climate type, with mean temperatures of 16.8 °C for July and -18.8 °C for January. Mean annual precipitation for the area is 455 mm (Environment Canada 2016).

The primary research site was a reclaimed overburden dump constructed with saline/sodic overburden materials and capped in 2011 with 1.5 m of clean (non-saline) subsoil/overburden and 0.5 m of either upland forest-derived forest floor-mineral mix (FFM) or lowland-derived peat-mineral mix (PM) (Table 2.1). These are common reclamation soil materials in the region, as soil is completely removed during mining operations and must be replaced as part of the reclamation process (MacDonald et al. 2012; MacDonald et al. 2015a; Mackenzie and Naeth 2010). PM is generally quite rich in carbon (C) and nitrogen (N), and has a high water holding capacity, which helps facilitate plant establishment and growth (Table 2.1)

(McMillan et al. 2007; MacDonald et al. 2012; MacDonald et al. 2015a; Pinno and Errington 2015). However, FFM has more similar soil properties to the soil of the target ecosystem, and the potential to recover viable propagules and microorganisms from salvaged upland forest organic layers (McMillan et al. 2007; Mackenzie and Naeth 2010; MacDonald et al. 2015b).

The reclamation materials were placed in four roughly equal 20 ha patches covering the whole reclamation area, with one patch of each material undergoing fertilization treatments in June 2011 and June 2012 (FFM_{Fert}; PM_{Fert}). Fertilizer (29.9-9.1-9.1-9.1 NPKS) was applied aerially at 100 kg N ha^{-1} each year. Fertilization is a common, yet not universal treatment, used to offset potential nutrient deficiencies in reclamation soil materials (Rowland et al. 2009). The site was seeded with barley (*Hordeum vulgare*) following the 2011 fertilization to control erosion. Natural sites used as reference for comparison were nearby aspen-dominated forest sites burnt by the 2011 Richardson wildfire, which affected 576,000 ha of area in northeastern Alberta from May to August of that year and was characterized by intensities greater than 10,000 kW m^{-1} and peak spread rates greater than 30 km per day (Pinno and Errington 2016). As these reference sites were disturbed during the 2011 growing season, their regeneration from the disturbed state would begin at the same time as growth of planted and naturally-recovering trees and other plants on the reclamation area, allowing a comparison of recovering sites that are at approximately the same state of maturity.

Sampling was conducted between May and August 2014 within six 10 m-radius circular plots per treatment (seven plots from PM_{Fert}; five plots from the reference sites). From each plot, four surface soil cores (88.7 cm^3) were collected at 0 – 15 cm depth to determine soil nutrients. Four pairs of Plant Root Simulator (PRS™) probes (Western Ag Innovations Inc., Saskatoon, SK, Canada) were installed at a 10 cm depth in each plot, and left in place for six weeks during the peak growing season between June – August (Qian and Schoenau, 2002; Johnson, et al. 2005) as a measure of bioavailable nutrients. Foliar tissue was collected at each plot as a total of 50 leaves harvested from 5 – 10 aspen seedlings at the height of the growing season in August.

2.2.2 – Nutrient Concentration Analysis

Soil samples were air dried then sieved to 2 mm and ground. Ground soil samples were analyzed for C, N, phosphorus (P), sulphur (S), calcium (Ca), magnesium (Mg), manganese (Mn), aluminum (Al), iron (Fe), potassium (K), and sodium (Na). The procedure for the metal analysis was a slightly modified version of Lafleur et al.'s (2013) methods, adjusted to work with fine-textured soils, as outlined by Hendershot and Duquette (1986). A sample of 0.5 g of each soil was extracted with 30.0 mL of 0.1 M $BaCl_2$ with an end-over-end shaker at 15 rpm for 2 h, followed by centrifugation for 15 min at 700 g (290 mm rotor diameter). The supernatant was filtered through Whatman No. 41 filter paper and analyzed with a Varian AA240FS Sequential Atomic Spectrometer (Varian Medical Systems, Palo Alto, CA, USA), using Atomic Absorption Spectroscopy to determine Ca, Mg, Mn, Al, and Fe, and using Atomic Emission Spectroscopy for K and Na. The extraction procedure was repeated on the same soil samples, using 0.1 M HCl to extract ions from highly soluble materials, such as calcites. Extraction was repeated once again, using 1 M HNO_3, to extract ions from more recalcitrant materials. Totals of each ion extracted by each treatment were summed together to arrive at the entire soil nutrient pool. Bray-extractable P was determined via the Kelowna Method (Van Lierop 1988; Kalra and Maynard 1991). Soil C, N, and S were determined by dry total combustion on a LECO C/N Analyzer (LECO Corporation, 141 St. Joseph, MI, USA). Soil pH was measured from saturated pastes using water and 0.01 M $CaCl_2$ (Kalra and Maynard 1991).

For bioavailable nutrient analysis, the PRS probes were removed from the soil, washed with distilled water and sent to Western Ag. Innovations for extraction and analysis of cations and anions. Colourimetry was used to determine samples' NO_3^- and NH_4^+ concentrations; using a FIALab 2600 automated flow injection analysis system (FIAlab Instruments Inc., Bellevue, WA, USA). Inductively-coupled plasma optical emission spectrometry (ICP-OES) with an Optima 8300 inductively-coupled plasma optical emission spectrometry system (Perkin Elmer Inc., Woodbridge, ON, Canada) was used to determine P, K, S, Ca, Mg, Mn, Al, Fe, copper (Cu), zinc (Zn), boron (B), cadmium (Cd) and lead (Pb) (Western Ag Innovations 2015).

Foliar tissue was oven dried at 40 °C then ground. A 0.03 g portion from each foliar tissue sample was extracted with 5.0 mL of 1 M HNO_3, and digested in a Mars Microwave Digestion System (CEM Corporation, Matthews, NC, USA) (Baker and Suhr 1982). Digested samples were filtered through Whatman No. 42 filter paper and diluted with 25 mL of deionized water. Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-OES) using an iCap 6000 Series ICP (ThermoFisher Scientific Inc., Waltham, MA, USA) was used to determine concentrations of Ca, Cu, Fe, K, Mg, Mn, Na, P, and S. Nitrogen concentration was determined via combustion using an ECS 4010 CHNSO Analyzer (Costech Analytical Technologies, Inc., Valencia, CA, USA).

2.2.3 – Statistical Analysis

2.2.3.1 – Univariate Statistics

A one-way Analysis of Variance (ANOVA) was performed on each nutrient within each pool, followed by a Tukey's Honest Significant Difference test (Tukey 1949). For datasets that contained non-normally distributed data, as determined by the Shapiro-Wilk test (Shapiro and

Wilk 1965), a Kruskal-Wallis test was used instead of ANOVA (Kruskal 1952). All four reclamation treatments and the natural reference treatment were compared to each other using R (R Core Development Team 2015-16) and the Agricolae package. As not all data was normally distributed, Spearman rank correlation between foliar nutrients, soil, and bioavailable nutrient pools was calculated with the PerformanceAnalytics and Hmisc packages in R (Spearman 1904). For this analysis of individual nutrients, only the six macronutrients, N, P, K, S, Ca, and Mg, were analyzed (Marschner 2002).

2.2.3.2 – Multivariate Statistics

A Non-Metric Multidimensional Scaling (NMS) ordination was performed on the data with a Bray-Curtis (Sørensen) dissimilarity matrix in order to compare the entire nutrient profile between treatments, rather than just individual nutrients in isolation. Non-metric multidimensional scaling (NMS) was selected for this multivariate analysis because it does not require the assumption of normal or linear data distribution, and can compute a large dataset with a large range of variation (Rowland et al. 2009) NMS analysis has been previously used to evaluate trends in ecological nutrient distribution data in the AOSR (Howell et al. 2017). Analysis was performed using the "slow and thorough" setting in PC-ORD v 6.19 (MjM Software Design, Gleneden Beach, 163 Oregon, USA), which computes 500 iterations per analysis. No transformations were applied to the data prior to analysis. A secondary matrix was also employed to compare for correlations with other factors observed at those same plots (Table 2.2). In cases where only a one-dimensional solution was found, skewing variables were removed from the main matrix until a satisfactory two or three-dimensional solution could be determined. The bioavailable nutrient pool was skewed by Ca and Mg, while the foliar nutrient pool was skewed by Mn and Fe. These nutrients were removed from the respective primary matrices, but kept in the secondary matrices. Multiple Response Permutation Procedures (MRPP) were used to assess statistical differences between treatment groupings (McCune et al. 2002).

To assess nutrient profile correlation between foliar and belowground nutrient pools, a Mantel Test was performed for each reclamation treatment and the reference sites. Each comparison's test statistic (*r*) was evaluated with the randomization test option, rather than Mantel's asymptotic approximation, as there were fewer than 40 plots analyzed in each dataset (Peck 2010). The Mantel Test measures response redundancy between two distance matrices constructed from multiple response matrices, essentially, correlation of points' relative positions (Peck 2010). It is the only commonly available tool used to determine similarity of such matrices (Peck 2010). The more similar the matrices' distance measures, the greater correlation there is between overall plot placements in the relative ordination space of the two matrices (Brosofke et al. 2001).

2.2.3.3 – Functional Similarity Index

The *T* value of the *MRPP* pairwise comparison of nutrient profiles between two groups of samples was used as a measure of similarity. When this comparison is between a reclaimed and reference site, it could potentially be used as an alternative to critical value comparisons for individual nutrients. As NMS occurs in a non-Euclidean analysis space, there are no absolute values that can be used as criteria for success, as can be done in normal analysis of individual nutrients (McCune et al. 2002). As such, differences in the *T* values were used to determine the relative differences between reclaimed sites in their degree of similarity to the natural reference sites. In addition to comparing individual nutrient pools for similarity, an overall comparison of all three nutrient pools combined was also performed.

$$\left(\frac{\left|(T_{a:Ref}) - (T_{b:Ref})\right|}{T_{a:Ref}}\right) * 100\%$$

Where: $T_{a:Ref}$ is the pairwise comparison *MRPP T* between the reference sites and the reclamation treatment most similar to the reference sites for the given nutrient pool

$T_{b:Ref}$ is the pairwise comparison *MRPP T* between the reference sites and the listed reclamation treatment for the given nutrient pool

Similarly, the relative impact of fertilization on the similarity of the different soil preparations to the reference sites was evaluated using the pairwise *MRPP T* values comparing the reference sites and the reclaimed treatments.

$$\left(\frac{(T_{U:Ref}) - (T_{F:Ref})}{T_{U:Ref}}\right) * 100\%$$

Where: $T_{U:Ref}$ is the pairwise comparison *MRPP T* between the reference sites and the listed unfertilized reclamation treatment (ie: FFM or PM) for the given nutrient pool

$T_{F:Ref}$ is the pairwise comparison *MRPP T* between the reference sites and the listed fertilized reclamation treatment (ie: FFM_{Fert} or PM_{Fert}) for the given nutrient pool

2.3 – Results

2.3.1 – Individual Nutrients in Different Pools

The soil nutrient pool differed significantly among treatments, with reference sites having lower N, S, Ca, and Mg concentrations compared to reclamation treatments (Figure 2.1, column 1). Sulphur concentration was higher in PM treatments than in reference and FFM soils. Fertilization did not significantly increase concentrations of nutrients relative to unfertilized PM or FFM (Figure 2.1, column 1). Phosphorus did not differ significantly among treatments. Potassium only differed between FFM_{Fert} and reference sites. (Figure 2.1, column 1). In the bioavailable nutrient pool, the reference sites had higher P and K concentrations than PM, and lower S, Ca, and Mg. (Figure 2.1, column 2). The reclamation treatments differed from each other only in the concentration of S (PM > FFM) and K (PM < FFM) (Figure 2.1, column 2). Fertilization did not significantly impact bioavailable nutrient concentrations for either PM or FFM (Figure 2.1, column 2). Foliar nutrient concentrations only differed significantly among treatments for S and Mg, with foliar Mg significantly lower in reference sites than in reclamation treatments, aside from the FFM_{Fert} treatment, and S higher in the PM treatments (Figure 2.1, column 3). All other foliar nutrient concentrations (N, P, K, Ca) were similar among all treatments.

Foliar nutrient concentrations showed an inconsistent correlation to the other nutrient pools (Table 2.3). Foliar N was only positively related to bioavailable N in PM_{Fert}, and soil N in FFM_{Fert} and PM. Foliar P was correlated to soil P only in the reference sites. Foliar K was correlated to soil K only in FFM_{Fert}. Foliar S, Ca, and Mg were never significantly correlated with the other pools. Out of sixty opportunities for potential correlations to occur between foliar and soil or bioavailable pools across the various treatments, only five significant correlations were observed.

2.3.2 – Multivariate Nutrient Profile Analysis

Multivariate analysis of the entire nutrient profile, rather than individual nutrients, revealed that the four reclamation treatments were more similar to each other in terms of soil nutrient profile than to the reference sites (Figure 2.2A), with FFM the most similar to the reference sites (Figure 2.2A; Table 2.4; Table 2.5). Fertilization decreased similarity to the reference sites, with the loss of similarity being more pronounced in PM than FFM (Table 2.5). For the bioavailable nutrient pool, all four reclamation treatments were again found to be more similar to each other than to the reference sites, with FFM_{Fert} most similar to the reference sites (Figure 2.2B; Table 2.4; Table 2.5). Fertilization increased similarity to reference sites for both cover soil types (Table 2.5). As with the soil and bioavailable nutrient pools, foliar nutrient pools of reclaimed sites were more similar to each other than to nutrient pools of reference sites (Figure 2.2C; Table 2.4; Table 2.5). However, unlike in the soil and bioavailable pools, FFM treatments were not inherently more similar to reference sites than PM. Fertilization increased the similarity of FFM to the reference nutrient profile. When all three nutrient pools were analyzed together in a single NMS ordination, the FFM treatments were more similar to the reference sites than the PM treatments (Figure 2.2D; Table 2.5). Fertilization decreased FFM's similarity to the reference sites, but had a minimal impact on PM's similarity to the reference sites. Overall, as was the case with each individual nutrient pool, the reclaimed treatments were much more similar to each other than to the reference sites.

The only significant multivariate correlation between foliar and soil nutrients identified with the Mantel Test was identified in the reference sites, with the bioavailable nutrient pool. There were no significant positive correlations for foliar nutrient pools to either other nutrient pool for any of the reclamation treatments (Table 2.7).

2.3.3 – Functional Similarity Index

FFM was most similar to reference sites in the soil pool, and in the combined ordination (Table 2.5). FFM_{Fert} was most similar to reference sites for both bioavailable and foliar nutrient pools. The difference between FFM_{Fert} and the other treatments was quite pronounced in the bioavailable nutrient pool, with a relative decrease in similarity to the reference sites of 31.53 %

between it and FFM, the second-most similar treatment. The combined analysis of all pools shows less overall difference between the most and least similar treatments than any individual treatment, with only an 8.31 % decrease in similarity between FFM and PM_{Fert}, the most and least similar reclaimed sites, respectively, to the reference site (Table 2.5).

Fertilization appears to greatly increase the similarity of FFM to the reference sites in the bioavailable and foliar nutrient pools (23.97 % and 19.53 %, respectively), but this increased similarity does not translate to an overall increase in similarity in the combined analysis (Table 2.6). PM does not see as much benefit from fertilization, as the only increase in relative similarity to reference sites in an individual nutrient pool, a gain of only 0.96 %, occurs in the bioavailable pool (Table 2.6). In both soil treatments, fertilization decreases the relative similarity of the soil pool to the reference sites (Table 2.6). Comparing the effect of fertilization on the similarity of reclamation treatments to natural reference sites shows a similarity decrease of 2.54 % for FFM and a similarity increase of 0.24 % for PM (Table 2.6).

2.4 – Discussion

2.4.1 – Criteria and Indicators

Concentrations of nutrients differed between treatments in the soil nutrient pool, but those differences decreased or disappeared completely in the bioavailable and foliar nutrient concentration pools. This decrease in differences moving from soil to foliage is likely because plants do not accumulate excess nutrients in their leaves once their basic requirements have been met (Ingestad and Åagren 1988), instead, they will grow more biomass if more of the limiting resources become available. For example, Munson and Timmer (1995) found that in jack pine, crown biomass increased in response to fertilization and vegetation control treatments while concentrations of foliar nutrients remained relatively unchanged. Furthermore, the increase in bioavailable N availability that resulted from the treatments was not reflected by the foliar N concentration changes observed in that study. While there is no biomass information for our study, this pattern of foliar N concentration remaining unchanged when fertilizer was added to a site mirrored our results.

Of the six macronutrients analyzed individually in the three nutrient pools, only N, P, and K were ever correlated between the foliar pool and either of the belowground pools and then only sporadically (Table 2.3). As foliar nutrient concentrations are recommended as indicators of overall site fertility, a direct positive correlation would have to be assumed between foliar nutrient concentrations and other nutrient pools (Brockley 2001; AESRD 2013). A further implication is that such correlations among the foliar nutrient pool, the bioavailable pool, and the soil nutrient pool should be consistent across all treatments and for all nutrients. While some studies have suggested that there is generally a correlation between foliar and soil nutrients (Boerner et al. 1984; Wang and Klinka 1997; Ordoñez et al. 2009), our results demonstrated that this assumption cannot be applied to all nutrients in all species of plants in all locations (Table 2.3). Gonzalez et al. (2010)'s results showed that individual tree species were unequally flexible in nutrient use efficiency, and these efficiencies are variable under different conditions. Lack of correlation between foliar and soil pools for S, Ca, and Mg may imply that trees are selectively not taking up abundant nutrients from the ecosystem and are regulating the content in their tissues.

2.4.2 – Multivariate Nutrient Profiles and Interconnected Ecological Processes

Even if multivariate analysis of nutrient concentrations is used, correlation between foliar and soil nutrient pools remains poor. The NMS ordinations suggest that the reclamation treatments do not have foliar nutrient concentration profiles that are at all similar to those found in reference sites. The Mantel Test showed only a significant positive correlation between foliar and bioavailable nutrient pools in the reference sites (Table 2.7). It is possible that equilibrium has been reached between the established biological community and the nutrient supply rates at the reference sites, while the lack of a firmly established vegetation community at the reclaimed sites means there is still an opportunity for some plants to capitalize on unused nutrients (Davis et al. 2000). Regardless of whether or not univariate or multivariate analyses are used, foliar nutrient concentration is not a reliable indicator of soil nutrient pools.

Management practices, such as fertilization, can disrupt the steady-state nutritional equilibrium that arises on a site (Davis et al. 2000). This large influx of nutrients into the available pool is often of greater benefit to rapidly-growing species than to slower-growing perennial species like trees (Chang et al. 1996; Davis et al. 2000). While the greatest impact on the relationship of nutrients between pools would occur during the growing season when fertilizer was first added, this effect could potentially persist for many years onward (Haynes and Naidu 1998). The addition of fertilizer increased the similarity of reclaimed sites to reference sites for the short-term bioavailable nutrient pool, but not the long-term soil nutrient pool (Table 2.6). In cases where a nutrient's supply may already exceed demand, such as adding N fertilizer to an already N-rich PM, luxury uptake of N may be encouraged, but only if trees can outcompete other plants for the excess N (Hu et al. 2014). Depending on the scarcity of other resources, not just soil nutrients, but also other commodities such as water and light, and the relative competitive ability of each individual plant, competition for growth and dominance at different locations could vary greatly (Staples et al. 1999; Pinno and Bélanger 2009). It is interesting that the addition of fertilizer led to a lower concentration of foliar P when applied to

FFM. It is possible that the addition of nutrients released the aspen from a nutrient limitation, which triggered the production of more biomass, until another limitation was reached (Vitousek et al. 2010). That foliar P concentration is strongly correlated with soil P in the reference sites, but not in the reclaimed sites, may be because the P is not as readily available in the intermediary bioavailable nutrient pool at the reclaimed sites (Figure 2.1). When deficient in P, plants will invest a greater proportion of their P supply into roots instead of leaves (Freeden et al. 1989). It is possible that reference site trees have more proportionate P distribution throughout all of their tissues than trees growing on the reclaimed sites, leading to a more direct relationship between P taken up from soil and foliar P concentration. This also underscores the importance of distinguishing between and examining both the soil and bioavailable nutrient supplies.

2.4.3 – Local Natural Reference Sites and a Functional Similarity Index

In lieu of specific nutrient requirements, average foliar nutrient concentrations can be used as critical values, such as those determined by Paré et al. (2013) (Table A.1). However, Paré et al. (2013)'s data comes from mature trees, while aspen measured in this study were, at most, three-year old seedlings. In white spruce, Wang and Klinka (1997) observed a negative correlation between stand age and foliar N, P, and K concentrations. This reinforces the need to properly account for differences in trees' ages and environments when interpreting the importance of foliar nutrient concentrations. This is one advantage of using a local site for reference values. Regeneration of aspen on natural reference sites began during the same growing season reclamation plots were established. This allows for a more relevant comparison between plots than using average values amalgamated from several studies across Canada.

One possible way to interpret these results is to reconsider the desired endpoint of an indicator criterion and the means by which it can be achieved (Failing and Gregory 2003). Foliar nutrient concentrations showed almost no differences between any of the reclaimed treatments and the reference sites (Figure 2.1). This implies that, because trees at the reference sites were healthy, trees on the reclaimed sites are also taking in proper amounts of nutrients required for healthy growth and that the endpoint of equivalent land capability is being met according to the standard of tree foliar nutrient concentration critical values. However, soil nutrient concentrations, which represent the long-term supply of nutrients, and bioavailable nutrients, which represent the short-term supply of nutrients, had much greater differences between reference and reclaimed sites. This suggests that because plants regulate their internal nutrient concentrations, once trees have acquired adequate amounts of nutrients for growth, foliar nutrient concentration does not change proportionately with changes in soil nutrient concentrations (Munson and Timmer 1995). Due to the lack of immediate feedback from one nutrient pool to another, using the pools to indicate the status of the others, especially over a broad area, is unlikely to provide accurate, useful information. As Ballard and Carter (1986) noted, correlation between pools is often highly variable, depending on a number of spatial and temporal factors, including stand age and specific environmental factors that vary from site to site, including relative nutrient abundance. Just as they cautioned that correlation with soil nutrients might be too low for diagnosing potential nutrient deficiencies in plants, we caution that the correlation is also too low to diagnose soil nutrient status from foliar nutrient concentrations. This applies for individual nutrients as well as entire nutrient pools.

Rather than assessing reclaimed sites in the context of critical values for individual nutrients, the use of multivariate statistics allows for comparison of reclaimed sites to natural reference sites that would have overall capability for nutrient cycling similar to that of the pre-disturbance ecosystems. Multivariate examination of the different nutrient pools indicated that the lack of similarity between reference and reclamation sites observed in the soil nutrient pool does not decrease in the bioavailable or foliar nutrient pools, nor does it decrease when all pools are considered together (Figure 2.2; Table 2.4). This implies that the endpoint of equivalent land capability is not necessarily being achieved for any individual pool within the nutrient cycle, according to the means of nutrient pool similarity. This interpretation however, is limited by the fact that only one type of natural ecosystem was considered during the analysis, while the reclaimed ecosystems may exhibit greater similarity towards other ecosystem types that exist within the region but were not considered in this study. Nevertheless, as this reference site is adjacent to the reclaimed sites, some expectation of similarity in the reclaimed ecosystem as it matures is not unreasonable.

Future use of a Functional Similarity Index system to evaluate the progress of reclamation could incorporate multiple reference ecosystem types in order to make comparisons between the type of ecosystem that is targeted as the end goal, and the types of ecosystem that a reclaimed site is currently most similar to. Understanding the different properties between the desired ecosystem and the most current most similar ecosystem could help to guide management actions. Additionally, the use of a single value from a similarity index, such as the *T* value of the *MRPP* pairwise comparisons, simplifies the vast amount of information and makes it easier to evaluate the overall impact of management actions. This single value incorporates both direct and indirect impacts of each action taken into the overall similarity score. This statistic could be used to evaluate the return on investment of different reclamation treatments in comparison with each other. For instance, in the combined analysis, addition of fertilizer to FFM decreased similarity to the reference sites by 2.54% and addition of fertilizer to PM only increased the relative similarity to the reference sites by only 0.24 %, which implies that fertilization may not ultimately be worthwhile for either soil material (Table 2.6). There was increase in similarity to the reference sites of 23.97 % for bioavailable FFM, compared to a decrease in similarity to the reference sites of 12.51% in soil PM (Table 2.6), meaning that if bioavailable nutrient supply is a priority, it may be worthwhile to fertilize FFM.

2.5 – Conclusion

Our results provide evidence that the current recommended use of foliar nutrient concentrations as an indicator of soil nutrient availability and cycling is inadvisable in reclaimed and natural forest ecosystems. Alone, foliar nutrient concentrations remain useful for the diagnosis of severe nutrient deficiencies within trees, but do not reveal the cause of the deficiency. Critical values can provide a simple standard for the complex information that makes up plant nutrition, but it is unclear if those values hold any actual meaning in wildland ecosystems (Ballard and Carter 1986), especially given variability and competition for resources in those ecosystems (Chang et al. 1996; Hu et al. 2014). The use of a multivariate approach based in similarity to existing ecosystems instead of a univariate approach rooted in critical values to this monitoring is suggested, as our results demonstrate that a multivariate approach may provide an alternative means of assessing the relationship between foliar and soil nutrient pools and assessing the outcomes of management actions. This indicator system quantifies differences between sites, simplifies a vast array of nutrient data into individual data points, and provides one overall picture of each nutrient pool that incorporates any potential relationships that may exist between nutrients within pools. Further refinement of these multivariate analysis techniques may lead to the development of a reliable Functional Similarity Index system that can be used to efficiently assess multiple aspects of nutrient cycles within ecosystems without requiring substantial changes in field data collection methodologies.

Tables and Figures

Table 2.1 – Mean basic soil characteristics for reclaimed and benchmark plots in the 0-15 cm depth at study site and natural reference sites. Reporting means with standard error in brackets. Treatments marked with the same letter are not significantly from each other for that property. TC = Total Carbon; TN = Total Nitrogen; CEC = Cation Exchange Capacity

	TC (%)	TN (%)	pH ($CaCl_2$)	CEC (cmol kg^{-1})
FFM	2.61 (0.72) b	0.12 (0.03) b	7.10 (0.21) a	66.90 (20.39) b
FFM_{Fert}	4.97 (2.48) ab	0.22 (0.08) ab	6.74 (0.42) ab	63.04 (4.77) b
PM	8.07 (4.85) a	0.25 (0.18) ab	5.82 (1.17) bc	99.82 (15.48) a
PM_{Fert}	9.13 (3.80) a	0.33 (0.16) a	6.35 (0.59) ab	113.69 (22.79) a
Reference	1.07 (0.32) b	0.07 (0.004) b	5.07 (0.78) c	23.50 (4.15) c

Table 2.2 – Variables used in the main and secondary matrices for Nonmetric Multidimensional Scaling (NMS)

Matrix	Variables Included
Soil Nutrients Main Matrix	From 0-15 cm depth soil cores: Al, Ca, Fe, K, Mg, Mn, Na (all cmol kg^{-1}); Total N, Total S (%); Total P (mg kg^{-1})
Bioavailable Nutrients Main Matrix	From PRS probes (all µg 10 cm^{-2} 6 wk^{-1}): Al, B, Ca*, Cd, Cu, Fe, K, Mg*, Mn, Total N ($NH_4^+ + NO_3^-$), P, Pb, S, Zn
Foliar Nutrients Main Matrix	From foliar tissue (all mg g^{-1}): Ca, Cu, Fe*, K, Mg, Mn*, N, Na, P, S
Secondary Matrix (used for all nutrient pools)	All nutrients from main matrices, plus: soil pH (from both H_2O and $CaCl_2$ methods); Soil carbon (%); Total soil cation exchange capacity (sum of analyzed cations, cmol kg^{-1}); All soil variables, including pH, C, and CEC from the 15-30, 30-45, and 50+ cm depth soil cores (units as above); Separate bioavailable NH_4^+ and NO_3^- (µg 10 cm^{-2} 6 wk^{-1})

*Nutrient removed from main matrix due to skewing of NMS output

Table 2.3 – Spearman's correlation scores for foliar nutrient concentrations to soil and bioavailable nutrient concentrations within each reclamation treatment at study site and natural reference sites. Significant results bolded. * = $p < 0.1$; ** = $p < 0.05$; *** = $p < 0.01$

	FFM		FFM_{Fert}		PM		PM_{Fert}		Reference	
Nutrient	Soil	Bio-available	Soil	Bio-available	Soil	Bio-available	Soil	Bio-available	Soil	Bio-available
N	0.66	0.66	**0.89****	0.66	**0.93*****	-0.26	0.40	**1.00*****	0.00	-0.40
P	0.26	0.31	-0.20	-0.43	-0.60	-0.60	0.50	0.00	**1.00*****	0.20
K	-0.66	0.12	**0.83****	0.49	-0.12	0.43	-0.49	0.50	0.10	0.50
S	-0.39	-0.54	-0.03	0.49	-0.14	0.20	0.25	0.57	0.21	-0.70
Ca	0.54	-0.09	-0.09	0.66	0.31	-0.09	0.54	0.25	0.00	0.70
Mg	-0.09	-0.31	-0.70	0.46	-0.60	0.49	-0.04	0.57	0.20	0.30

Table 2.4 – Pairwise MRPP statistics comparing nutrient pools between treatments. More positive *T* values indicate greater similarity between groups. An *A* value closer to 1 indicates increasing homogeneity between points within groups. A *p* value of less than 0.05 indicates that the *T* and *A* values differ significantly from what would be expected by chance alone.

Treatments Compared		*T*	*A*	*p*
Soil				
FFM	PM	-2.769	0.150	0.019
FFM	FFM_{Fert}	0.290	-0.015	0.547
FFM	PM_{Fert}	-5.434	0.333	0.001
FFM	Ref	-4.691	0.260	0.001
PM	FFM_{Fert}	-4.552	0.245	0.002
PM	PM_{Fert}	-0.471	0.022	0.277
PM	Ref	-5.619	0.360	0.001
FFM_{Fert}	PM_{Fert}	-5.939	0.375	0.001
FFM_{Fert}	Ref	-4.763	0.261	0.002
PM_{Fert}	Ref	-6.322	0.414	0.001
Bioavailable				
FFM	PM	-5.459	0.335	0.002
FFM	FFM_{Fert}	-0.178	0.010	0.350
FFM	PM_{Fert}	-2.912	0.171	0.021
FFM	Ref	-6.075	0.434	0.001
PM	FFM_{Fert}	-5.731	0.297	0.006
PM	PM_{Fert}	-1.959	0.066	0.047
PM	Ref	-6.175	0.438	0.001
FFM_{Fert}	PM_{Fert}	-2.341	0.113	0.033
FFM_{Fert}	Ref	-4.619	0.341	0.004
PM_{Fert}	Ref	-6.115	0.425	0.001
Foliar				
FFM	PM	-2.130	0.113	0.036
FFM	FFM_{Fert}	-0.424	0.026	0.251
FFM	PM_{Fert}	-2.342	0.120	0.025
FFM	Ref	-5.397	0.309	0.001
PM	FFM_{Fert}	-2.362	0.140	0.026
PM	PM_{Fert}	0.051	-0.003	0.448
PM	Ref	-5.107	0.340	0.001
FFM_{Fert}	PM_{Fert}	-1.758	0.089	0.060
FFM_{Fert}	Ref	-4.343	0.270	0.004
PM_{Fert}	Ref	-5.120	0.302	0.002
Combined				
FFM	PM	-5.282	0.318	0.002
FFM	FFM_{Fert}	0.036	-0.002	0.445
FFM	PM_{Fert}	-1.637	0.078	0.069
FFM	Ref	-5.700	0.434	0.001
PM	FFM_{Fert}	-6.086	0.404	0.001
PM	PM_{Fert}	-0.664	0.028	0.207
PM	Ref	-6.174	0.429	0.001
FFM_{Fert}	PM_{Fert}	-3.078	0.142	0.014
FFM_{Fert}	Ref	-5.845	0.433	0.001
PM_{Fert}	Ref	-6.159	0.417	0.001

Table 2.5 – Comparison of relative decrease in similarity (*MRPP T*) of treatments to the natural reference sites, relative to the most similar treatment, per nutrient pool.

	Soil		Bioavailable		Foliar		Combined	
Most Similar	FFM	--	FFM_{Fert}	--	FFM_{Fert}	--	FFM_{Fert}	--
	FFM_{Fert}	-1.54 %	FFM	-31.53 %	PM	-17.61 %	PM	-2.54 %
	PM	-19.80 %	PM_{Fert}	-32.39 %	PM_{Fert}	-17.89 %	PM_{Fert}	-8.05 %
Least Similar	PM_{Fert}	-34.79 %	PM	-33.68 %	FFM	-24.27 %	FFM	-8.31 %

Table 2.6 – Comparison of effect of addition of fertilization on relative similarity (*MRPP T*) of reclamation treatments to the natural reference sites, per nutrient pool.

	Soil	Bioavailable	Foliar	Combined
FFM_{Fert} vs FFM	-1.54 %	+23.97 %	+19.53 %	-2.54 %
PM_{Fert} vs PM	-12.51 %	+0.96 %	-0.24 %	+0.24 %

Table 2.7 – Mantel Test results (*r*-values) comparing foliar nutrient pool to soil and bioavailable nutrient pools for each treatment type at study site and natural reference sites, with skewing variables removed. Foliar pool had Fe and Mn removed; Bioavailable pool had Ca and Mg removed. Positive *r*-values indicate a positive correlation of changes in one matrix with another; negative values indicate negative correlation. *r*-values closer to zero represent a weaker correlation. Significant results bolded. * = $p < 0.1$; ** = $p < 0.05$

Treatment Type	Foliar-Soil Pools	Foliar-Bioavailable Pools
FFM	**-0.432***	-0.003
FFM_{Fert}	-0.037	0.257
PM	-0.084	-0.192
PM_{Fert}	0.088	-0.074
Reference	-0.183	**0.537****

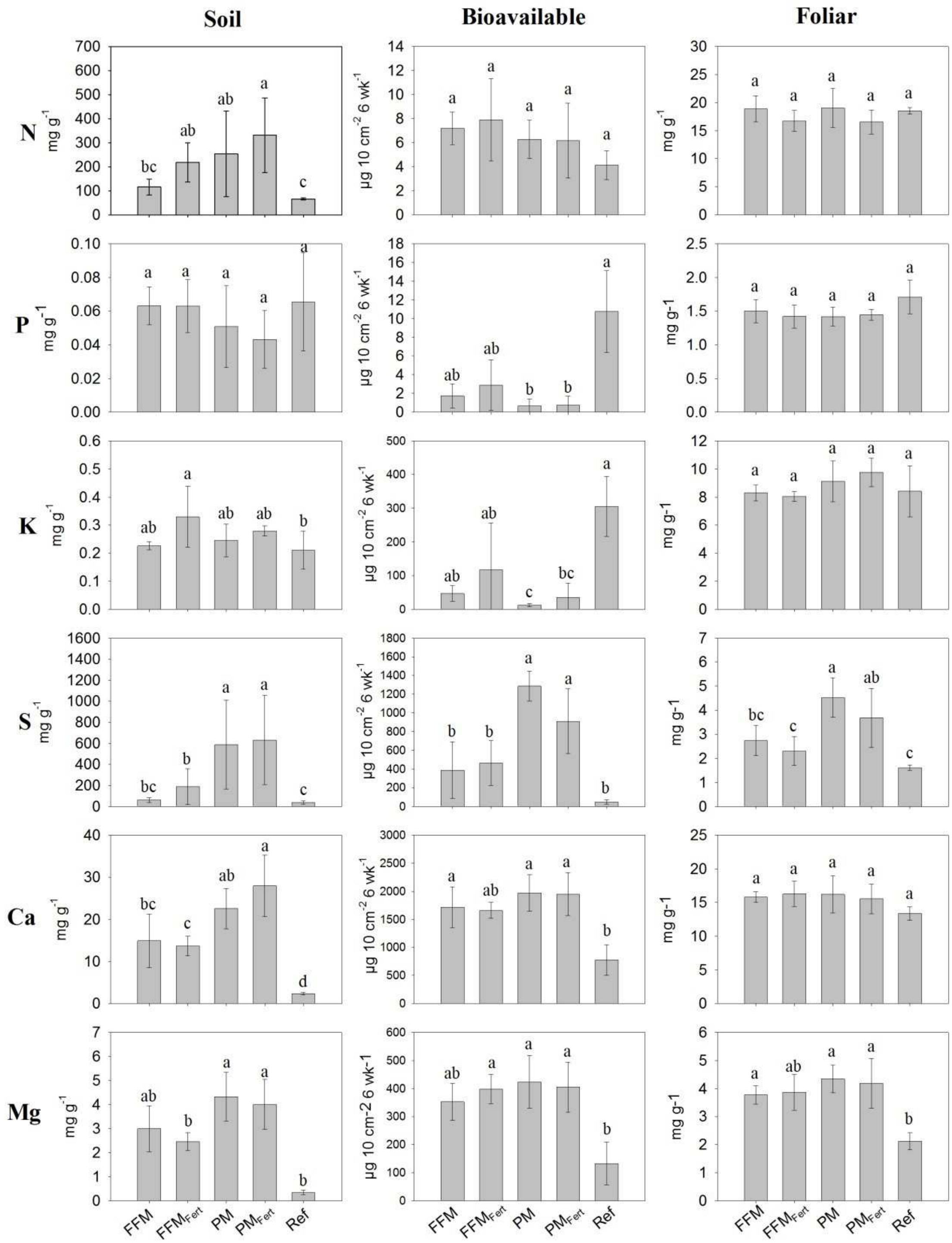
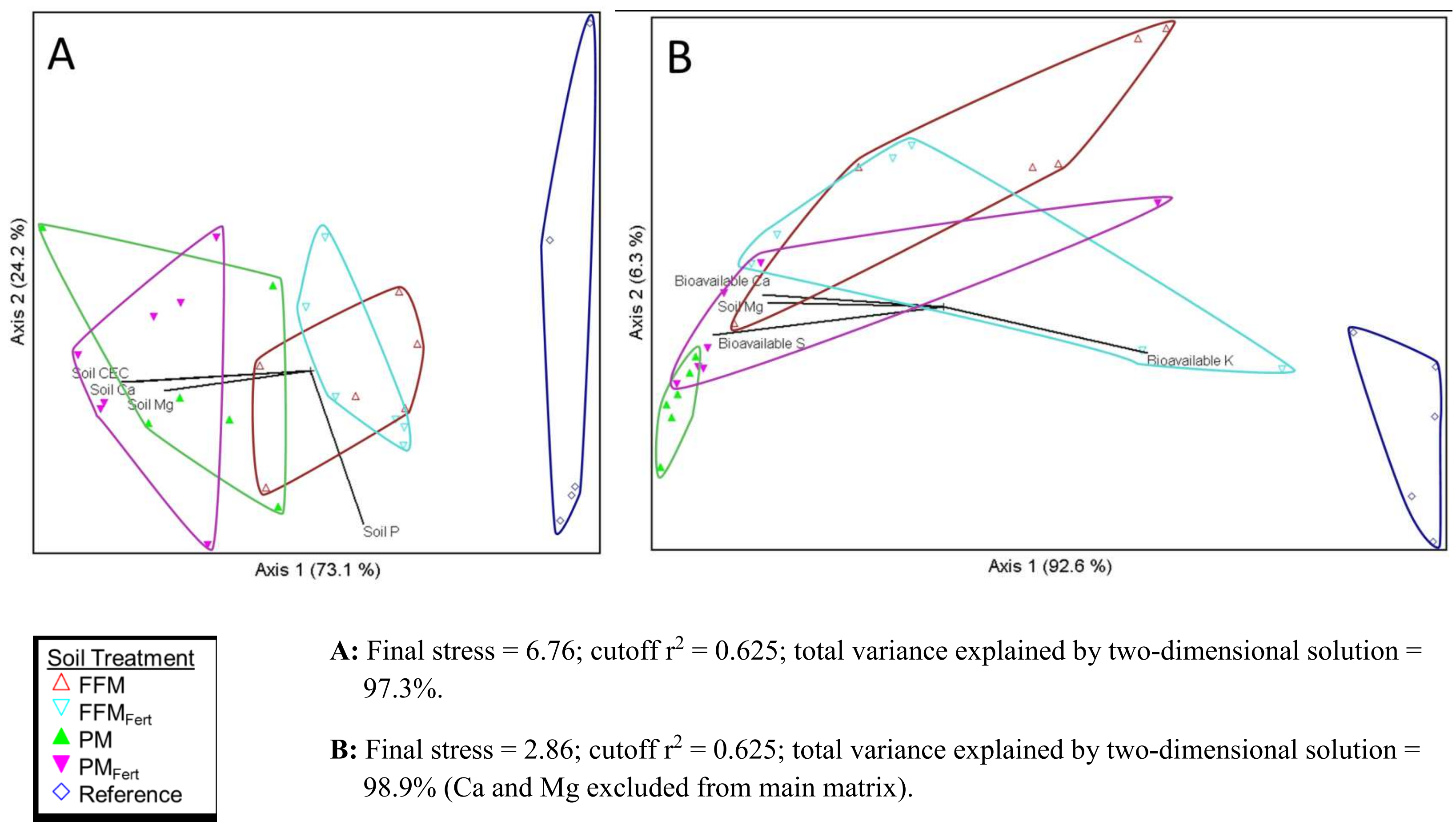


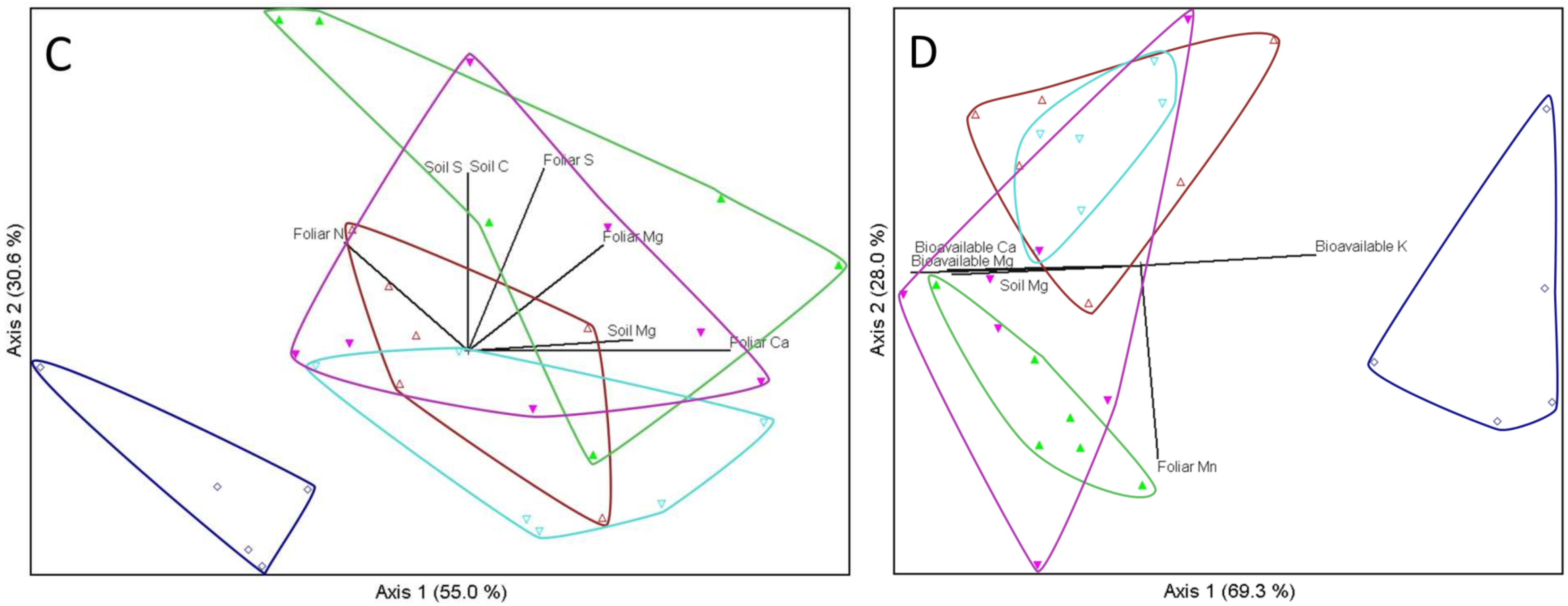
Figure 2.1 – ANOVA of nutrient levels in soil, bioavailable, and foliar nutrient pools from different reclamation treatments at study site and natural reference sites. Treatments with a non-normal distribution were compared using the Kruskal-Wallis test instead of ANOVA. Treatments marked with the same letter do not differ significantly from each other for that nutrient in that pool (significance level of $p < 0.05$).



A: Final stress = 6.76; cutoff r^2 = 0.625; total variance explained by two-dimensional solution = 97.3%.

B: Final stress = 2.86; cutoff r^2 = 0.625; total variance explained by two-dimensional solution = 98.9% (Ca and Mg excluded from main matrix).

Figure 2.2 – Non-metric multidimensional scaling (NMS) ordination bi-plot of nutrient supply rates in soil (A), bioavailable (B), foliar (C), and combined (D) nutrient pools of reclaimed soil treatments at study site compared with natural reference sites. Each axis is labelled with the percentage of variance it explains.



C: Final stress = 7.45; cutoff r^2 = 0.500; total variance explained by three-dimensional solution = 95.1% (Mn and Fe excluded from main matrix) (Dimensions 1 and 2 shown).

D: Final stress = 6.53; cutoff r^2 = 0.625; total variance explained by two-dimensional solution = 97.3%.

Figure 2.2 – Non-metric multidimensional scaling (NMS) ordination bi-plot of nutrient supply rates in soil (A), bioavailable (B), foliar (C), and combined (D) nutrient pools of reclaimed soil treatments at study site compared with natural reference sites. Each axis is labelled with the percentage of variance it explains.

3.0 The Impact of Upper Subsoil Placement Depth on Reclamation Success in the Alberta Oil Sands Region as Examined by a Nutrient Profile Similarity Index

3.1 – Introduction

In the Athabasca oil sands region (AOSR) ecosystem degradation is caused by the complete or partial removal of all soil and vegetation in order to access subsurface bitumen, to facilitate mine infrastructure, or to store waste byproducts from mining operations such as overburden or tailings (Rowland et al. 2009). The Government of Alberta mandates, under the *Environmental Protection and Enhancement Act* (EPEA), that land used for oil sands mining must be reclaimed to an equivalent land capability that existed prior to industrial operations, though not necessarily to an identical land use (Alberta Environment and Sustainable Resource Development [AESRD] 2013). Depending on treatments used in reclamation – soil materials, amendments, and preparations, vegetation planting regimes, follow-up activity, etc., the reclamation process can be accelerated, returning the disturbed site to an equivalent land capability more rapidly.

A key aim of choosing these treatments is shifting the environment from one resembling a primary succession, where there are no biological or substrate legacies from prior ecosystems, to one of a secondary succession, where these legacies can assist ecosystem regeneration. As soil is completely removed during mining operations and must be replaced as part of the reclamation process, two soil materials are commonly used in the AOSR, forest floor-mineral mix (FFM), which is salvaged from upland forest ecosystems, and peat mix (PM), which is salvaged from lowland wetland ecosystems (MacDonald et al. 2012; MacDonald et al. 2015a; Mackenzie and Naeth 2010). PM has a high water holding capacity, and is generally contains a large supply of soil organic C and N, which helps facilitate plant establishment and growth (McMillan et al. 2007; MacDonald et al. 2012; MacDonald et al. 2015a; Pinno and Errington 2015). However, PM tends to lack the biological and substrate legacies that are appropriate to an upland forest ecosystem, which is generally the target of most reclamation operations in the AOSR (Mackenzie and Naeth 2010). This makes sense, as biological and substrate legacies in PM would originate from the lowland ecosystems PM is harvested from. Conversely, FFM has more similar soil properties to the soil of upland forest ecosystems, including substrate legacies from prior disturbance events such as wildfires (Mackenzie and Naeth 2010; MacDonald et al. 2015b). As for biological legacies, FFM also has the potential to recover viable propagules and microorganisms from salvaged upland forest organic layers (McMillan et al. 2007; Mackenzie and Naeth 2010; MacDonald et al. 2015b). Unfortunately, the supply of FFM is very limited in the AOSR, which means that FFM must be used as efficiently as possible during reclamation operations. While PM is more abundant than FFM, efficient use of soil resources remains a generally beneficial operating procedure and research into combinations of techniques and amendments that help improve resource-use efficiency and rate of recovery of equivalent land capability is ongoing in the AOSR.

In addition to the choice and depth of topsoil, the choice and depth of subsoil material is another potential area of concern in reclamation capping. Highly compacted subsoil or non-soil fill materials that are within tree rooting depth may limit availability of water and nutrients for growing trees by reducing the storage capacity of the soil, ultimately slowing recovery of the ecosystem (Naeth et al. 2011; Jung et al. 2014). Much of the fill material in the AOSR is overburden that was removed during mining, in addition to waste products from refining processes, such as tailings sand (Kessler et al. 2010; Naeth et al. 2011). Storage piles of excess overburden are often themselves reclaimed in place. In addition to possessing properties undesirable for a plant-growth medium, such as poor water holding capacity and structure, these fill materials may contain substances that are harmful to plants, such as residual hydrocarbons leftover from extraction and refinement processes or high concentrations of salts (Kessler et al. 2010; Naeth et al. 2011). Even non-contaminated overburden material in the AOSR originated as marine sediments, meaning that it potentially has a high enough salinity/sodicity to interfere with plant growth if inherent salts were to become bioavailable within the rooting zone of plants, which they can through chemical weathering and capillary action, if non-saline/sodic capping material is too shallow (Kessler et al. 2010). Suitable non-saline/sodic subsoil material, like appropriate topsoil material, is not an infinite resource and does have costs associated with collection, transport, storage and placement. Efficient use of subsoil material should provide a suitable growth medium for regenerating plant communities without consuming excess supply. Adequate placement depths and layer thicknesses will reduce impediments to plant growth and accelerate regeneration on sites, allowing land managers to achieve more successful reclamation of different projects throughout the AOSR.

The Criteria and Indicators Framework for Oil Sands Mine Reclamation Certification [CIF] (AESRD 2013) establishes several criteria that are to be met in order to determine whether successful reclamation has been achieved. Each criterion has a series of indicators that can be used as evidence to determine if the criterion has been met. The features of a good indicator are: quantification of important information, with emphasis of that information’s relevance; simplification of complex information for easier communication; and, making the process of measurement and monitoring of the information simpler and more cost-effective (National Research Council 2000). If the success of a reclamation project is to be declared, then it requires that the indicators being used to make that judgment are valid metrics for the criteria that have been outlined. Some indicators from the CIF do not necessarily meet this requirement, such as the Foliar Nutrients indicator as part of the criterion evaluating that nutrient cycling has been re-established on the reclaimed site (Hogberg et al. Manuscript in Review). While this indicator possesses those three key features of a good indicator, perhaps a fourth key feature should be added to that list: The indicator is a valid representation of the larger, more complex process in all situations where it is being used as an indicator.

One of the weaknesses of the current CIF for assessing nutrient profiles of sites undergoing reclamation in Alberta is that only foliar nutrient concentrations (N, P, K, Ca, Mg, Cu, Zn, Fe, Mn, B, and optionally S) are considered indicators of nutrient cycling being re-established (AESRD 2013), despite that, by definition, this process requires multiple pools for nutrients to transfer between. Foliar nutrient concentrations represent the pool of nutrients held in plant tissue and can be used as indicators to monitor the health of plant species of interest in both agricultural and wildland ecosystems (Ballard and Carter 1986; Wang and Klinka 1997; Chen et al. 1998; Gower et al. 2000; Brockley 2001; Gonzalez et al. 2010). The soil nutrient pool, including nutrients in soil organic matter and nutrients bonded to the surface of soil particles, represents nutrients present in the soil itself. Forces holding these ions in place can be released slowly over time (Mengel et al. 1990; Röing et al. 2006). For an ecosystem with long-lived perennial plants an enduring nutrient supply is critical for long-term sustainability. Finally, the bioavailable nutrient pool consists of nutrients that are dissolved in water within the pore space of soils and is the source of soluble nutrients for plants and other soil organisms (Lafleur et al. 2013). Because these nutrients are not adsorbed to the surface of soil particles, they are more readily available for plant uptake (Smethurst 2000; Qian and Schoenau 2002; Johnson et al. 2005).

Individual analysis of foliar macronutrients was ineffective when it came to distinguishing between different reclamation treatments (Hogberg et al. Manuscript in Review). This method also underperformed for recognizing differences between ecosystems recovering naturally from wildfire and sites undergoing reclamation after extreme disturbance from oil sands mining (Hogberg et al. Manuscript in Review). Correlation between foliar and belowground nutrient concentrations was generally not strong enough to consider foliar nutrients as a good indicator for concentrations of nutrients within the soil (Hogberg et al. Manuscript in Review). The set of nutrients that is most important to examine for one ecosystem will not necessarily be as important for others. While not all nutrients significantly drive differences between sites, it is important to verify that nutrients that are monitored as indicators are sufficiently representative of differences between sites. However, if examining all three nutrient pools – soil, bioavailable, and foliar (representative of plant nutrients as a whole), and multiple soil horizons or plant species are examined, the list of variables to measure and compare quickly balloons. While Hogberg et al. (Manuscript in Review) were able to use non-metric multidimensional scaling (NMS) and multiple response permutation procedures (MRPP) to analyze differences between reclamation and natural sites as a multivariate analysis follow-up to univariate examinations of individual nutrients, these methods had some limitations. Primarily, NMS is difficult to quantify in meaningful ways, and even with the use of MRPP, they could only examine overall similarity of treatment groups, rather than individual plots within treatments (McCune et al. 2002). Another way to explore all of these variables from all three nutrient pools and evaluate as to whether they make good indicators for differences between sites is to use principal components analysis (PCA), which, like NMS, makes no assumptions about how much variation will be explained by each component (Brejda et al. 2000). An index can be developed from the most strongly correlated variables with each axis (Masto et al. 2008; Mukhopadhyay et al. 2014). This sort of index is useful for repeat measurements, as it means not necessarily having to continually re-analyze entire sample sets with PCA to determine which nutrients drive differences between sites. Although, depending on the type of data that is collected during each sampling period, a PCA performed on each subsequent sampling could show shifts in what nutrients (in which pools) are primarily responsible for differences between sites over the course of site development. This would also ensure that the most suitable minimum data set is being used to create the index for each comparison. The more similar sites are to each other overall, the closer the sites will appear in the PCA results (McCune et al. 2002).

The specific research questions of this study were: (1) When using a PM topsoil, does placing subsoil material to a greater depth on reclamation sites improve ecological function, as evidenced by water availability and nutrient availability in multiple nutrient pools, as well as the growth rates of trees? (2) Does the overall similarity of the PM-capped reclamation sites to FFM-capped reclamation sites and natural reference sites improve when using a greater depth of subsoil material placement? (3) Can principal components analysis be used to generate an index that can clearly and accurately distinguish between different reclamation treatments and natural reference sites to identify the critical differences in the nutrient profile, consisting of the soil, bioavailable, and plant nutrient pools? (4) Does this index provide a greater insight into the underlying factors that drive the differences between reclamation treatments and natural ecosystems than examining plant macronutrient availability alone?

3.2 – Methods

3.2.1 – Study Area and Experimental Design

The study was conducted on a reclaimed oil sands mine overburden dump site, approximately 75 km north of Fort McMurray, Alberta, Canada (57° 20' 5" N 111° 32' 7" W). The natural ecosystem region is composed primarily of boreal mixedwood forests with major upland tree species of jack pine (*Pinus banksiana*), trembling aspen (*Populus tremuloides*) and white spruce (*Picea glauca*), and major lowland species of black spruce (*Picea mariana*), birch (*Betula* spp.), and tamarack (*Larix laricina*) (Natural Regions Committee 2006). Upland soils are generally dry and coarse-textured, with some finer-textured mineral soils interspersed. Lowland soils are often saturated and composed primarily of organic materials. Upland soils predominantly belong to the Brunisol and Luvisol soil orders; lowland soils are predominantly Organic and Gleysolic soils. (Natural Regions Committee 2006). This region has a continental climate type with mean temperatures of 16.8 °C for July and -18.8 °C for January with mean annual precipitation of 455 mm (Environment Canada 2016).

The primary research site, the Aurora Soil Capping Study (ASCS), which was a reclaimed overburden dump constructed and capped in 2012 with a different topsoil and subsoil materials, replicated at the operational scale for research purposes (Figure 3.1; Figure 3.2; Table 3.1) (Barber et al. 2015; Hankin et al. 2015). Topsoil material was either upland-derived forest floor-mineral mix (FFM) or lowland-derived peat mix (PM). Subsoil material for the studied treatments was blended Brunisolic B and C horizon material salvaged from a 1 m depth and above. The B/C blended subsoil (BCB) contained tarballs mixed into soil material (NorthWind Land Resources Inc. 2013; Barber et al. 2015). Subsoils were placed in horizons of deep (120 cm), medium (70 cm) or shallow (30 cm) thickness (NorthWind Land Resources Inc. 2013;

Barber et al. 2015). Treatments were established in 1 ha areas in triplicate within the study area. Sampling was conducted within 25 m x 25 m vegetation plots established within treatment blocks. Each plot was planted at a rate of 10,000 stems per hectare of aspen, jack pine, or white spruce. A fourth plot in each treatment contained a mixture of all three species (Hankin et al. 2015; Barber et al. 2015). These mixed plots were the plots examined in this study. Natural sites were used as reference for functional comparison and included nearby forest impacted by the 2011 Richardson wildfire, which affected 576,000 ha of area in northeastern Alberta during its 4-month duration and included intensities greater than 10,000 kW m^{-1} and peak spread rates greater than 30 km per day (Pinno and Errington 2016). These sites' regeneration from the disturbed state would begin at the same time as growth of planted and naturally-recovering trees and other plants on the reclamation area, allowing a comparison of recovering sites that are at approximately the same state of maturity. The regenerated mixed aspen and pine stands were classified as "b" ecosites, and had predominantly Orthic or Eluviated Dystric Brunisol soil, with some Eutric Brunisol patches (Natural Regions Committee 2006; Haynes and Naidu 1998).

Samples were collected in July and August 2015. From each plot sampled, soil samples were collected from the topsoil (TS) and upper subsoil (US) layers. Soil core samples (88.7 cm^3) were collected after a soil pit was dug to see clear separation of soil layers. Foliar tissue was collected at each plot from aspen and pine seedlings. For pine seedlings, only new annual growth was sampled.

3.2.2 – Soil Nutrient Analysis

Soil samples (150 g) were weighed into a mason jar and then adjusted to a water content of approximately 60 % of field capacity using deionized water. To determine concentrations of bioavailable nutrients in the soil solution, a pair of ion exchange resin membranes were inserted into each jar, and left in place for two weeks while the sealed jars were incubated at 30 °C (Qian and Schoenau 2002; Johnson et al. 2005). Jars were removed from the incubator, and unsealed for approximately 1 – 5 minutes every 48 hours to allow re-aeration to occur. Rehydration was performed with distilled water during these times as necessary to maintain the approximately 60 % of field capacity soil moisture content. Upon removal from soil, the membranes were washed with distilled water and extracted with 15 mL 0.5 M HCl per membrane pair for analysis of cations and anions. Samples were analyzed with a SmartChem Discrete Wet Chemistry Analyzer, Model 200 (Westco Scientific, Limited, Brookfield, CT, USA) for P, NO_3^- and NH_4^+. Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES) using an iCap 6000 Series ICP (ThermoFisher Scientific Inc., Waltham, MA, USA) was used to determine K, S, Ca, Mg, Mn, Mo, Al, Fe, Cu, Zn, B, Cd, Na, and Pb.

Soil samples were air dried, sieved to 2 mm, then ground. A subsample of 0.05 g of each soil was digested with 5.0 mL of 1 M HNO_3 in a Mars Microwave Digestion System (CEM Corporation, Matthews, NC, USA) (Baker and Suhr 1982). Digested samples were then filtered through Whatman No. 42 filter paper and diluted with 25 mL of deionized water. ICP-OES using an iCap 6000 Series ICP (ThermoFisher Scientific Inc., Waltham, MA, USA) was used to determine the concentrations of Ca, Mg, Mn, Al, Fe, Cu, Mo, Ni, Zn, B, S, P, K, and Na. Soil carbon and nitrogen were determined via dry combustion using an ECS 4010 CHNSO Analyzer (Costech Analytical Technologies, Inc., Valencia, CA, USA). The pH of the soil was measured from saturated pastes using water (Kalra and Maynard 1991).

3.2.3 – Foliar Nutrient Analysis

The foliar tissue was oven dried at 40 °C then ground. A 0.03 g subsample from each foliar tissue sample was digested with 5.0 mL of 1 M HNO_3 in a Mars Microwave Digestion

System (CEM Corporation, Matthews, NC, USA) (Baker and Suhr 1982). Digested samples were then filtered through Whatman No. 42 filter paper and diluted with 25 mL of ultrapure water. As with the soil samples, ICP-OES using an iCap 6000 Series ICP (ThermoFisher Scientific Inc., Waltham, MA, USA) was used to determine the concentrations of Ca, Mg, Mn, Fe, Cu, Mo, Ni, Zn, B, S, P, K, and Na. Total foliar carbon and nitrogen were determined via dry combustion using an ECS 4010 CHNSO Analyzer (Costech Analytical Technologies, Inc., Valencia, CA, USA).

3.2.4 – Statistical Analysis

3.2.4.1 – Univariate Statistics

A one-way Analysis of Variance (ANOVA) was performed on the six plant macronutrients (N, P, K, S, Ca, and Mg) (Marschner 2002), within each pool and each horizon, followed by a Tukey's Honest Significant Difference test (Tukey 1949). All four reclamation treatments and the natural reference sites were compared to each other using R (R Core Development Team 2015-16) and the Agricolae package. A one-way ANOVA was also performed on the heights of the two different tree species (aspen and pine) on each treatment type and the reference sites. This was also followed by a Tukey's Honest Significant Difference test to compare treatment means. The available water holding capacity (AWHC) of each plot was calculated according to LCCS guidelines (CEMA 2006). Mean AWHC for each treatment and the natural reference plots was calculated, and means were compared with a one-way ANOVA and a Tukey's Honest Significant Difference test.

3.2.4.2 – Nutrient Profile Index

The aim of using a principal components analysis (PCA) is to quickly narrow down the large amounts of data into only the most useful indicators of the overall trends – the minimum data set. PCA considers all available data and determines which combination of variables explains the most overall variation in the dataset. There is a correlation score for each principle component for each variable that is a part of the data matrix, and through these correlation scores, it becomes possible to see which variables have the strongest correlations along each PC axis, positive or negative. From there, it becomes possible to eliminate those variables that have little correlation in either direction along the axes. Further data reduction is possible at this step by identifying and eliminating any variables that are strongly correlated with a variable that is more strongly correlated to each axis itself (Masto et al. 2008). The strength of this type of analysis is that it can consider numerous variables and the relationships between those variables simultaneously, and reduce a massive dataset down to the most important aspects (McCune et al. 2002).

A PCA was performed on the entire multivariate dataset in order to examine the influence of measured aspects from the nutrient pools (nutrient concentrations and some environmental factors) on these ecosystems' nutrient cycling function and to determine if certain elements influence this function more strongly than others. For the mixed tree species plots, there were 87 individual quantitative variables included in the analysis (Full list in Table A.3). This would amount to 1.55×10^{26} individual comparisons of variables and comparisons of variables in relation to each other, which would take an impractical amount of time and effort to examine using only univariate methods. The analysis was performed using the correlation cross-products matrix in PC-ORD v 6.19 (MjM Software Design, Gleneden Beach, 163 Oregon, USA). As part of the procedure of a PCA, variables are all relativized by column (i.e. each value is divided by the sum of all values for that variable) (McCune et al. 2002).

Some nutrients were in concentrations so low in their respective pools that there were greater than seven instances of a value of “0”. These nutrients were removed from the analysis, as too many zero-values in an analysis can badly skew determination of eigenvalues (Peck 2010). The nutrients removed from the analysis were: soil Mo, bioavailable Cd and Mo, and foliar Mo and Ni. Tree heights, while measured, were not included in the analysis due to a limitation of the data due to the structure of the ASCS. Planted trees on reclamation sites had been grown for one year in nursery prior to being planted, while trees on reference sites regenerated naturally following the fire (Barber et al. 2015). Consequently, trees at the reclaimed site had a year of growth under optimal conditions, while trees at the natural sites had to grow with limited resource availability and in competition with other plants at the site. This amounts to over 25 % of the total lifespan of the trees that regenerated or were grown in 2012 by 2015. This could have increased their growth rates of planted seedlings relative to naturally regenerated trees, as they would have faced no competition from other plants or physical limitations to their growth. There was no control to determine the impact of the greenhouse relative to natural conditions, meaning that the advantage was unaccounted for.

Principal components were considered significant if they had eigenvalues >1.0, and explained ≥5.0 % of the total variation within the examined data (Masto et al. 2008). Because the individual nutrients and environmental properties are relativized as part of the PCA process, an eigenvalue greater than 1 means that a PC axis explained more variation than any individual variable included in the analysis (Brejda et al. 2000). When the eigenvectors of these principal components are scaled to their standard deviations (V-Vectors) and applied to a PCA of a correlation matrix, they are equal to the correlation coefficient between scores for each variable in the initial matrix and each principal component variable (McCune et al. 2002). Using these correlation coefficients, the most strongly correlated nutrients for each principal component can be identified. The correlation can be positive or negative, as the sign indicates that two categories are related along opposing directions of the same axis, which indicates that different groups of data within the set have opposite correlations with that particular variable, which makes the variable a good indicator for identifying whether a third group of data is more similar to one group or the other. As Masto et al. (2008)'s methods suggested, the variable with the strongest correlation was identified for each principal component, and all variables with correlation coefficients within 10 % (absolute) of the strongest correlation coefficient were also identified and retained. This pared down list of significant variables for each principal component was used to create the nutrient profile index equation. After the highly-weighted variables were determined by principal components analysis, ANOVA and Tukey's Honest Significant Difference tests were performed on any variables that had not already been analyzed with these univariate methods, to determine if there were significant, observable patterns for these variables between different treatments. Unlike the method used by Masto et al. (2008), variables that were highly correlated with each other were not removed from the equation, as the different nutrients are all essential for plant growth, and unlike the soil physical and chemical properties examined by Masto et al. (2008), are not necessarily redundant.

Each retained variable (relativized by column) from each principal component is multiplied by the percentage of variation that the principal component explains (Masto et al. 2008). These values are summed together to generate a raw index score. For variables with negative correlation coefficients, a negative value was used in the equation. In order to keep individual variables at similar weights within the nutrient profile equation, all variables were relativized by column (ie: the value for each variable from each individual plot was divided by the sum of all values for that variable across all plots) before being substituted into the equation. Raw index scores were used to determine the mean index score for the reference sites. Each individual plot's raw index score was then compared to this average reference score in order to determine each plot's absolute similarity to the theoretical target reference site (the nutrient profile index value). As the absolute difference from the reference site mean approaches 0, the more similar the plot is to the theoretical reference site.

3.3 – Results

3.3.1 – Univariate Comparisons between Treatments

3.3.1.1 – Tree Heights

The reference sites and FFM treatments had significantly taller aspen than the PM-Medium and PM-Deep treatments, while the PM-Shallow treatment did not have aspen that were significantly different in height from any other treatment (Figure 3.3). For pine trees the only significant difference in height was between FFM and the reference sites (Figure 3.3).

3.3.1.2 – Available Water Holding Capacity

The AWHC of the reference sites was significantly greater than the AWHC of the PM-Shallow treatment (Figure 3.4). No other treatments were significantly different from each other (Figure 3.4). The full AWHC calculations according to the LCCS (CEMA 2006) are detailed in Table A.2.

3.3.1.3 – Soil Total Nutrient Pool Macronutrient Concentrations

The soil total nutrient pool showed significant differences among treatments in the topsoil, with the reference sites and FFM reclamation treatment having lower N, S, Ca, and Mg than the three PM reclamation treatments (Figure 3.5A, column 1). Phosphorus was only significantly different between the FFM treatment and PM-Medium treatments (Figure 3.5A, column 1). Potassium was only significantly lower in the FFM treatment compared to the PM-Shallow and PM-Deep treatments (Figure 3.5A, column 1).

For the soil nutrient pool in the upper subsoil, differences were less pronounced, expect for the reference sites having greater N and P than all reclamation treatments (Figure 3.5B, column 1). The soil N, S, and Ca levels for the upper subsoil were generally lower than the topsoil total soil nutrient levels for all nutrients and treatments, while the P, K, and Mg levels were similar to those found in the topsoil (Figure 3.5).

3.3.1.4 – Soil Bioavailable Nutrient Pool Macronutrient Availability

In the soil bioavailable nutrient pool in the topsoil, a much greater range of variability was seen, and no significant differences were observed for N and P, though a pattern of greater P in the FFM and reference sites, compared to the PM treatments seems to be emerging (Figure 3.5A, column 2). The FFM treatment and reference sites both had significantly higher levels of bioavailable K compared to the PM treatments (Figure 3.5A, column 2). Other macronutrient differences in the topsoil bioavailable pool included: S, for which the PM-Shallow treatment had a greater availability than the FFM treatment and reference sites; Ca, for which the PM-Shallow and PM-Deep treatments had a greater availability than the FFM treatment and reference sites; and Mg, for which the PM-Shallow and PM-Deep treatments had a greater availability than the reference sites (Figure 3.5A, column 2).

In the bioavailable soil nutrient pool in the upper subsoil, there were no significant differences between treatments for N, P, K, and Ca (Figure 3.5B, column 2). The PM-Shallow treatment had significantly greater levels of S and Mg than the reference sites (Figure 3.5B, column 2). In all nutrients except K, availability was greatly reduced in the upper subsoil, compared to the topsoil (Figure 3.5).

3.3.1.5 – Foliar Nutrient Pool Macronutrient Concentrations

Aspen foliar nutrient concentrations differed significantly among treatments for: N, for which the PM-Deep and PM-Medium treatments had a greater concentration that the reference sites; P, for which the FFM treatment and reference site had greater concentrations than all PM treatments; and Mg, for which the FFM treatment had a greater concentration than all PM treatments, while the reference sites had significantly greater concentrations than the PM-Medium and PM-Deep treatments, but not the PM-Shallow treatment. (Figure 3.5A, column 3). No significant differences were found in aspen foliar nutrient concentrations for K, S, or Ca (Figure 3.5A, column 3).

For pine foliar nutrient concentrations, significant differences among treatments were observed for: P, for which the FFM treatment and reference sites had greater concentrations than all PM treatments; S, for which all reclamation treatments had greater concentrations than the natural reference sites; and Ca, for which all PM treatments had greater concentrations than the FFM treatment and reference sites (Figure 3.5A, column 3). No significant differences were found in pine foliar nutrient concentrations for N, K, or Mg (Figure 3.5A, column 3).

3.3.2 – Principal Components Analysis

PCA found that there were six principal components with eigenvalues > 1.0 and explained ≥ 5.0 % of overall variance (Table 3.2; supplemental information in Table A.3). PC1 explained 33.2 % of total variation, and the variable with the strongest correlation to the principal component axis was aspen foliar Mn (-92.7 %) (Table 3.2). Seven other variables also had correlations within 10 % of aspen foliar Mn's correlation: topsoil soil S (92.7 %), pine foliar Mn (-91.3 %), pine foliar Ca (90.0 %), topsoil soil B (89.0 %), topsoil soil C (86.6%), topsoil bioavailable Mg (87.5 %), and topsoil bioavailable Ca (85.3 %) (Table 3.2). PC2 explained 12.6 % of total variation and the most strongly correlated variable was aspen foliar Fe (-82.2 %) (Table 3.2). No other variables had correlation values within 10 % of this value (Table A.3). PC3 explained 8.4 % of total variation and the variable with the strongest correlation was upper subsoil soil B (-81.3 %) (Table 3.2). Upper subsoil soil K (-74.2 %) was also strongly correlated with the axis of PC3 (Table 3.2). PC4 explained 7.9 % of total variation and the variable most strongly correlated to the axis of PC4 was topsoil bioavailable Fe (65.8 %) (Table 3.2). Other variables strongly correlated with the axis of PC4 were topsoil bioavailable Pb (65.2 %), topsoil bioavailable Cu (-56.4 %), and topsoil bioavailable B (+56.2 %) (Table 3.2). PC5 explained 5.6 % of total variation, and only topsoil bioavailable Al (-72.9 %) was strongly correlated with its axis (3). Finally, PC6 explained 5.2 % of total variation, and had pine foliar K (-69.3 %) and pine foliar Zn (-68.3 %) strongly correlated with the axis (Table 3.2). The final NPI equation was:

$$
\begin{aligned}
&NPI\ Raw\ Score \\
&= (33.159 * -[aspen\ foliar\ Mn]) + (33.159 * [topsoil\ soil\ S]) + (33.159 \\
&* -[pine\ foliar\ Mn]) + (33.159 * [pine\ foliar\ Ca]) + (33.159 \\
&* [topsoil\ soil\ B]) + (0.33159 * [topsoil\ bioavailable\ Mg]) + (33.159 \\
&* [topsoil\ soil\ C]) + (33.159 * [topsoil\ bioavailable\ Ca]) + (12.640 \\
&* -[aspen\ foliar\ Fe]) + (8.436 * -[upper\ subsoil\ soil\ B]) + (8.436 \\
&* -[upper\ subsoil\ soil\ K]) + (7.914 * [topsoil\ bioavailable\ Fe]) + (7.914 \\
&* [topsoil\ bioavailable\ Pb]) + (7.914 * -[topsoil\ bioavailable\ Cu]) \\
&+ (7.914 * [topsoil\ bioavailable\ B]) + (6.513 \\
&* -[topsoil\ bioavailable\ Al]) + (5.24 * -[pine\ foliar\ K]) + (5.24 \\
&* -[pine\ foliar\ Zn])
\end{aligned}
$$

3.3.3 – Nutrient Profile Index

Each individual plot was assessed using the NPI equation (Table 3.3). The raw scores of the reference sites were averaged together, and then the absolute difference was calculated between this average value and all plots' raw scores (Table 3.3). Of non-reference plots, the plot that was most similar to the mean reference site nutrient profile value, plot # M-18, was part of the FFM reclamation treatment group (Table 3.3). The least similar plot to the calculated mean nutrient profile was plot # M-25, in the PM-Shallow reclamation treatment group. If the mean NPIV is calculated for each treatment group, FFM is most similar to the reference sites, followed by PM-Medium, then PM-Deep, while PM-Shallow is the least similar overall (Table 3.3). A Tukey's Highly Significant Difference test calculated no significant difference for mean NPIV between the FFM treatment and the reference sites, while the PM reclamation treatments were all significantly different from those two treatment groups, but not from each other (Table 3.3).

ANOVA of the treatment means on the individual variables selected for the NPI equation show that there is a significant difference between treatments in the variables strongly correlated with PC1 and PC2 (Figure 3.6A). For all other variables used in the NPI equation, from PC3-PC6, there is however, no significant difference between any treatments, except for pine foliar Zn in PC6, where the PM-Deep treatment has significantly more foliar Zn than the reference sites (Figure 3.6A; Figure 3.6B).

3.4 – Discussion

3.4.1 – Impact of Topsoil Material and Subsoil Depth on Similarity to Natural Reference Sites

Aspen trees were taller on the FFM treatment and reference sites than on PM treatments. All reclamation treatments had taller pine trees than the reference sites, but were not significantly different from each other (Figure 3.3). Without a control for the impact of the initial nursery incubation provided for trees planted on reclamation treatments, meaningful conclusions about tree height cannot be drawn from comparing reclamation treatments directly to natural reference sites. As for differences among reclamation treatments, FFM facilitated the growth of taller trees. This increased height was significant for aspen, but not for pine. There was no significant difference between different subsoil depths for PM treatments for either tree species (Figure 3.3).

The natural reference sites had the greatest AWHC, but only PM-Shallow had significantly lower AWHC than the reference sites (Figure 3.4). The PM-Shallow's lower AWHC is due to the soil profiles of PM-Shallow treatment plots having depths of only 60 – 70 cm compared to the full 100 cm of other treatments (Table A.2). While the sandy BCB subsoil material may not provide the greatest water holding capacity, it provides more than the overburden material on which soils are placed. This does demonstrate the advantages of having actual soil material occupying the first 100 cm of the constructed soil profile. However, the results of the PCA indicate that this difference in AWHC however was never a significant determinant in overall site similarity (Table 3.2; Table A.3). Additionally, while having deeper soil does provide greater capacity for water and nutrient storage, the most important reservoir appears to be the topsoil layer, rather than the upper subsoil or lower subsoil layers. Differences in nutrient availability in the upper subsoil layers between treatments were much less pronounced and overall levels were much lower in upper subsoil layers than the topsoil layers (Figure 3.5; Figure 3.6). Additionally, the upper subsoil layer was much less likely to provide a significant indicator of differences between sites in the PCA, with only upper subsoil total soil B and K being highly weighted variables (Table 3.2; Table A.3).

The FFM topsoil reclamation treatment was more similar to recently fire-disturbed “b” reference ecosites than any PM topsoil reclamation treatment, among treatments that used a B/C Blended subsoil material. Where differences do exist between the different treatments for individual nutrients in different nutrient pools, the FFM treatment is generally more similar to reference sites than the PM treatments, such as for topsoil soil total N, S, Ca, and Mg; topsoil bioavailable P, K, S, and Ca; aspen foliar P and Mg; and pine foliar P and Ca (Figure 3.5A). Subsoil macronutrients were less useful for elucidating differences between reclamation treatments, as treatments generally showed no clear distinctions between FFM or PM treatments of different subsoil depth in either the soil total or soil bioavailable nutrient pools (Figure 3.5B). Foliar macronutrients did not generally reflect trends seen in belowground nutrient pools (Figure

3.5A), and alone would not necessarily present the full picture of differences in nutrient availability on different sites (Hogberg et al. Manuscript in Review).

3.4.2 – Generating a Similarity Index for Assessing Reclamation Success

The results of the PCA suggest that rather than the crucial differences between sites' nutrient profiles being rooted in soil N and P, or foliar macronutrients, that Mn, as evidenced by aspen and pine foliar Mn concentrations, may be a limiting nutrient for PM reclamation treatments (Figure 3.6A; Table 3.2). Conversely, topsoil soil S appears to be in great excess at PM sites compared to FFM reclamation sites and natural reference sites; the same is true of topsoil soil B, which was strongly correlated with S along the PC1 axis (Figure 3.6A; Table 3.2). While not a plant nutrient, total soil C in the topsoil was also a strong indicator of differences between site types, with PM treatment sites having much greater levels of C than the FFM treatment or reference sites. Topsoil bioavailable Mg and Ca, pine foliar Ca, and aspen foliar Fe were also higher in all reclamation treatments compared to reference sites (Figure 3.6A; Table 3.2). One implication of these results is that more of a nutrient, even a necessary macronutrient such as S, Ca, or Mg, is not always better when it comes to reclamation, as an excess of a nutrient decreases similarity of a site to ecosystems regenerating from natural disturbances. Another implication is that critical micronutrients may play a more important role in overall site similarity than commonly monitored macronutrients.

For variables selected from PC3 through PC6, there appears to be little difference between different reclamation sites and reference post-fire ecosites. There may be some concern in emphasizing the impacts of these specific variables, which are associated with Principal Components that only explain 5.2 – 8.4 % of total variation each; especially when these variables are only themselves correlated along these axes at rates sometimes less than 70 %, while variables strongly correlated along the PC1 axis, which explains 33.2 % of total variation, have been discarded (Table 3.2). This could be a case of the whole being more than the sum of its parts; and that by breaking principal components down to only the strongest individual variables, some explanatory power is lost. However, the role of a minimum data set is not to necessarily explain the entirety of the differences between designated treatments, but to provide the best possible set of the fewest variables possible that can still clearly differentiate between groups (Masto et al. 2008). While the ANOVA results for individual variables may not appear to show significantly differences, the NPI does show a clear trend as to which treatments and individual sites are most similar to the target reference ecosystem, and thus, the most currently "successful" reclamations. The PC-ORD program does include internal randomization tests for PCA to determine which axes are worth exploring to determine overall differences between samples. These stopping rules include Rnd-Lambda, Rnd-F, Avg-Rnd, and Broken Stick (BS), per Peres-Neto et al. (2005). According to the most conservative stopping rules, Rnd-Lambda and Avg-Rnd, the last useful axis is PC2. Rnd-F suggests that PC5 is the last useful axis. The BS stopping rule would include all axes through PC12. Given that there is no assumption of normality or correlation of all variables within this dataset, Rnd-Lambda and Rnd-F would likely be the most appropriate guidelines for stopping rules (Peres-Neto et al. 2005). Rnd-F's stopping point of PC5 largely agrees with the results obtained using Masto et al. (2008)'s stopping rule, which included PC6. Rnd-Lambda appears to be more in alignment with the differences shown between highly weighted variables in PC1-6, with the variables in PC1 and PC2 showing strong difference, but those in PC3-6 showing mostly no difference among treatments (Figure 3.6). There is no commonly accepted significance testing for individual variables within a PCA. The necessity of significance test for PCA variables has long been debated; a common thread among discussion is that significant variables should be selected on the basis of understanding and interpreting their meaning (Kaiser 1960).

One element of PCA to keep in mind is that outliers in the dataset will badly skew correlation coefficients (McCune et al. 2002). Ideally, many samples can be taken from the same type of plot, allowing outliers to be discarded or their impact minimized, as a more complete picture of the trend for that site type is identified. Another complication with using PCA is that it looks only for linear relationships between variables and links those relationships to the different plots, which can be categorized by site type/treatment following the PCA. If there are other kinds of relationships between variables, such as exponential relationships, or relationships involving threshold values, they may not be well-represented by PCA (McCune et al. 2002).

3.4.3 – Considerations for Future Index Iterations

Masto et al. (2008)'s study was conducted on agricultural sites, where soil parameters could adhere to specific prescribed guidelines. This allowed Masto et al. to assign scores to the data, based on whether there was enough of a needed nutrient or parameter (For example: N, P, K), not too much of a detrimental parameter (P-fixing capacity and K-fixing capacity), or the parameter values fell into an optimum range that was neither too much nor too little (porosity). In a wildland ecosystem, it can be argued that there are no appropriate thresholds that can be set in this manner, as the usage and cycling of nutrients will not be as controlled and predictable, nor will nutrients be removed from the ecosystem as harvested biomass. Thus, unlike Masto et al. (2008) and the soil quality index systems based on their work (Mukhopadhyay et al. 2014) or earlier soil quality index systems used as the basis for their system, we could not standardize each variable to a score out of 1, nor can a value of 1 be a target value for each individual variable. Instead, the mean raw index score for the reference sites was used as the target value for the overall index. This means that when using this system, the target value for the index will be dependent on the number and type of reference sites used. This does allow for the index to be adapted towards multiple types of natural reference sites. For example, if both “a” and “b” ecosite types exist in a region, it would be simple to calculate raw index score means for both types of sites and compare reclaimed sites to both means and determine if there were different similarity rankings. Whichever reference is selected as the “target” ecosystem, the more similar another site is to it, the closer that site’s NPIV will be to 0. If a reclaimed site had perfect similarity to the reference site for a nutrient’s concentration, then each would have the same fraction of the total value for that nutrient. If a reclamation operation had been perfectly successful in replicating a natural site’s nutrient profile, then the difference in the index score between that site and the reference site would be 0. Theoretically, it is possible that using the generated equation, a total result of 0 could be generated with extreme values for individual variables that cancel each other out in a way that brings overall similarity to the values of the reference sites, but the odds of such a statistical coincidence are low.

Plants regulate their internal nutrient concentrations, meaning that foliar nutrient concentrations are not guaranteed to change proportionately with changes in the belowground nutrient pools (Munson and Timmer 1995). This means that foliar nutrient concentrations alone are not necessarily the best indicators for nutrient cycling. Total plant nutrient content could be approximated through the multiplication of nutrient concentrations by estimated biomass of the plants at each site. Though the heights of trees at the study sites were measured, and could represent a proxy measurement for biomass, we believe that this approximation would not be appropriate in this case. Heights of selected individual trees do not necessarily represent the entirety of plant biomass that is present at each site. Estimates of cover density for not only aspen and pine, but for other species at the sites, which would also be competing for nutrients and other resources (Staples et al. 1999; Pinno and Bélanger 2009), would help provide a more comprehensive plant biomass estimate. A group of densely-packed, shorter trees may contain more biomass than a few sparsely-spaced individual trees. Another concern for the use of height as a proxy for biomass is that trees on reclamation sites were planted after one year of growth in nursery, while trees on reference sites are the products of natural regeneration from the year of the fire (Barber et al. 2015). This means that the trees at the reclaimed site have received optimal nutrition and growing conditions for their first year of growth prior to transplanting, compared to trees at the natural site. This represents an extra 25 – 33 % of the total lifespan of the trees that regenerated in 2011, depending on when exactly sampling and planting took place. This could have influenced their growth rates and nutrient content. Until a better method for controlling for these factors can be determined, it was decided to not incorporate plant height or biomass into the NPI.

3.5 – Conclusion

Using FFM as the topsoil for capping the reclamation of an overburden dump appears to provide a better soil environment for nutrient availability and movement between soil total, soil bioavailable, and plant tissue nutrient pools than using PM topsoil. Not only does FFM support the growth of taller trees, both aspen and pine, but the overall similarity of sites capped with FFM topsoil to natural reference sites is higher than the similarity of PM treatment sites to the natural reference sites. When using a blend of salvaged B and C horizons as an upper subsoil to underlay PM topsoil, different subsoil depths had a minimal impact on overall site similarity to natural reference sites. There was no significant difference in AWHC, and limited differences in soil chemical and physical properties and nutrient availability from different nutrient pools.

There was no significant difference in growth rates of aspen on pine on the different subsoil depth treatments. While the medium-depth subsoil treatment sites were most similar to reference post-fire sites and the shallow-depth subsoil treatment sites were least similar, differences between the similarity scores of all three subsoil depth treatments were not significant.

While the Nutrient Profile Index is not necessarily a perfect system of indication of differences between reclamation sites in the Athabasca oil sands region, it represents a step in the right direction for establishing the usefulness of multivariate data analysis for determining the most important factors for determining reclamation success, and how the principles of minimum data set analysis can improve representation of differences between sites. The results of this analysis provide more specific insight into how the nutrient profiles of the different reclamation treatments are dissimilar from the nutrient profiles of the natural ecosystems that they seek to replicate, compared to the recommended Criteria and Indicator Framework system that is currently in place. It is possible that refining this index, possibly through use of a standardized scoring function, may further simplify the data and make it easier to interpret (Masto et al. 2008; Mukhopadhyay et al. 2014). However, it is uncertain that such a scoring function could be developed for a wildland ecosystem as it has been for agricultural ecosystems, given the inherent variability and competition for resources in those ecosystems (Ballard and Carter 1986; Chang et al. 1996; Hu et al. 2014). Further testing of these techniques on other reclamation sites and natural reference sites, with different species of vegetation and potential methods for properly incorporating differing biomass, will help refine and standardize the index and how it is implemented. The same principles behind this Nutrient Profile could be applied to other aspects of ecosystem function. By analyzing a multivariate data set that includes other ecosystem metrics, such as biodiversity, microbial biomass, net primary productivity, or any number of other properties, a true Ecosystem Functional Similarity Index could be established. This Ecosystem Functional Similarity Index could potentially provide an even more accurate and in-depth tool for determining reclamation success from an ecological perspective.

Tables and Figures

Table 3.1 – Mean basic soil characteristics for reclaimed mixed tree sites and benchmark plots at study site and natural reference sites. Reporting means with standard error in brackets. Treatments marked with the same letter are not significantly from each other for that property in that horizon. TC = Total Carbon; TN = Total Nitrogen; EC = Electrical Conductivity

		TC (mg g^{-1})	TN (mg g^{-1})	pH	EC (dS cm^{-1})
	Topsoil / A Horizon				
PM-Shallow		241.00 (20.29) b	8.62 (1.71) b	7.09 (0.38) a	1.23×10^{-2} (2.04×10^{-3}) a
PM-Medium		321.77 (20.64) a	14.57 (2.89) a	7.54 (0.14) a	4.98×10^{-3} (4.40×10^{-3}) bc
PM-Deep		228.77 (23.54) b	10.47 (2.29) ab	7.49 (0.02) a	7.86×10^{-3} (7.80×10^{-4}) ab
FFM		24.35 (4.85) c	0.76 (0.17) c	5.88 (0.32) a	9.99×10^{-4} (3.50×10^{-4}) c
Reference		21.15 (4.31) c	0.97 (0.32) c	6.45 (1.42) a	2.81×10^{-4} (6.23×10^{-5}) c
	Upper Subsoil / B Horizon				
PM-Shallow		4.56 (1.59) ab	0.07 (0.10) b	7.27 (0.59) a	1.77×10^{-3} (1.02×10^{-3}) ab
PM-Medium		3.38 (0.58) b	0.01 (0.01) b	7.79 (0.22) a	3.22×10^{-3} (2.06×10^{-4}) a
PM-Deep		5.23 (1.72) ab	0.06 (0.07) b	7.56 (0.27) a	1.41×10^{-3} (6.93×10^{-4}) ab
FFM		3.96 (0.56) ab	0.01 (0.02) b	6.91 (0.54) a	5.22×10^{-4} (8.98×10^{-5}) ab
Reference		7.06 (1.77) a	0.40 (0.11) a	5.79 (0.04) b	2.60×10^{-4} (5.66×10^{-5}) b

Table 3.2 – Principal components analysis of the correlation matrix of nutrients from soil, bioavailable, and foliar nutrient pools from mixed tree plots at Aurora Soil Capping Study. For each principal component, the bolded values represent the variable with the highest absolute eigenvector (V-vector) and every variable with an absolute value within 10% of that value.

Principal Components	PC-1	PC-2	PC-3	PC-4	PC-5	PC-6
Eigenvalue	28.8	11.0	7.4	6.9	5.7	4.6
Variation (%)	33.2	12.6	8.4	7.9	6.5	5.2
Cumulative Variation (%)	33.2	45.8	54.2	62.2	68.7	73.9
Eigenvectors (V-Vectors)						
Environmental Physical or Chemical Properties						
Soil Total C – Topsoil	**0.866**	0.258	-0.100	-0.333	-0.048	-0.002
Foliar Nutrients						
Ca – Pine	**0.900**	0.171	-0.026	-0.243	-0.106	-0.198
Fe – Aspen	0.286	**-0.822**	0.095	0.140	-0.043	-0.039
K – Pine	-0.428	-0.094	0.053	0.282	0.087	**-0.693**
Mn – Aspen	**-0.927**	-0.110	-0.075	0.107	-0.073	-0.006
Mn – Pine	**-0.913**	-0.227	-0.040	0.164	-0.033	-0.030
Zn – Pine	0.539	-0.378	-0.166	-0.149	-0.083	**-0.683**
Bioavailable Nutrients						
Al – Topsoil	0.231	-0.240	0.004	0.427	**-0.729**	0.160
Ca – Topsoil	**0.853**	0.181	0.005	0.266	0.050	-0.206
Cu – Topsoil	0.037	-0.329	-0.192	**-0.564**	0.173	0.368
Fe – Topsoil	0.440	-0.027	-0.323	**0.658**	0.126	0.068
Mg – Topsoil	**0.875**	-0.066	-0.010	0.330	-0.082	-0.087
Pb – Topsoil	0.220	-0.310	-0.196	**0.652**	-0.279	0.218
Soil Nutrients						
B – Topsoil	**0.890**	0.356	-0.026	-0.188	-0.08	0.038
B – Upper Subsoil	0.214	0.217	**-0.813**	0.296	0.115	-0.018
K – Upper Subsoil	-0.036	0.149	**-0.742**	0.260	0.103	-0.114
S – Topsoil	**0.927**	0.275	-0.043	-0.013	0.114	0.119

Table 3.3 – Summary of application of nutrient profile index equation to individual plots at study site and natural reference sites. Treatment means were compared to the reference mean to determine relative similarity to the natural reference sites used as a benchmark for recovery from disturbance. Treatments marked with the same letter are not significantly from each other for Mean NPIV.

Treatment	Plot #	Raw Index Score	Treatment Mean Raw Index Score	Absolute Difference from Reference Mean (Nutrient Profile Index Value)	Mean NPIV by Treatment
PM-Shallow	[a]M-3	19.90		28.21	
	M-15	12.65	17.74	20.96	26.05 **a**
	M-25	20.66		28.97	
PM-Medium	M-7	14.03		22.34	
	M-22	15.29	13.76	23.59	22.07 **a**
	M-26	11.97		20.28	
PM-Deep	M-23	19.31		27.62	
	M-30	18.22	16.57	26.52	24.88 **a**
	M-34	12.12		20.50	
FFM	M-10	-4.03		4.28	
	M-18	-6.32	-5.78	1.98	2.52 **b**
	M-24	-7.00		1.38	
Reference	Ref-1	-7.15		1.15	
	Ref-2	-9.84	-8.31	1.53	1.02 **b**
	Ref-3	-7.93		0.38	

[a]The ASCS contains subplots within each treatment plot that have tree stands of different species compositions, including pure stands of aspen, pine, and spruce, as well as the mixed plots that contain a mixture of all three species (Barber et al. 2015). For this study, only the mixed plots were examined.

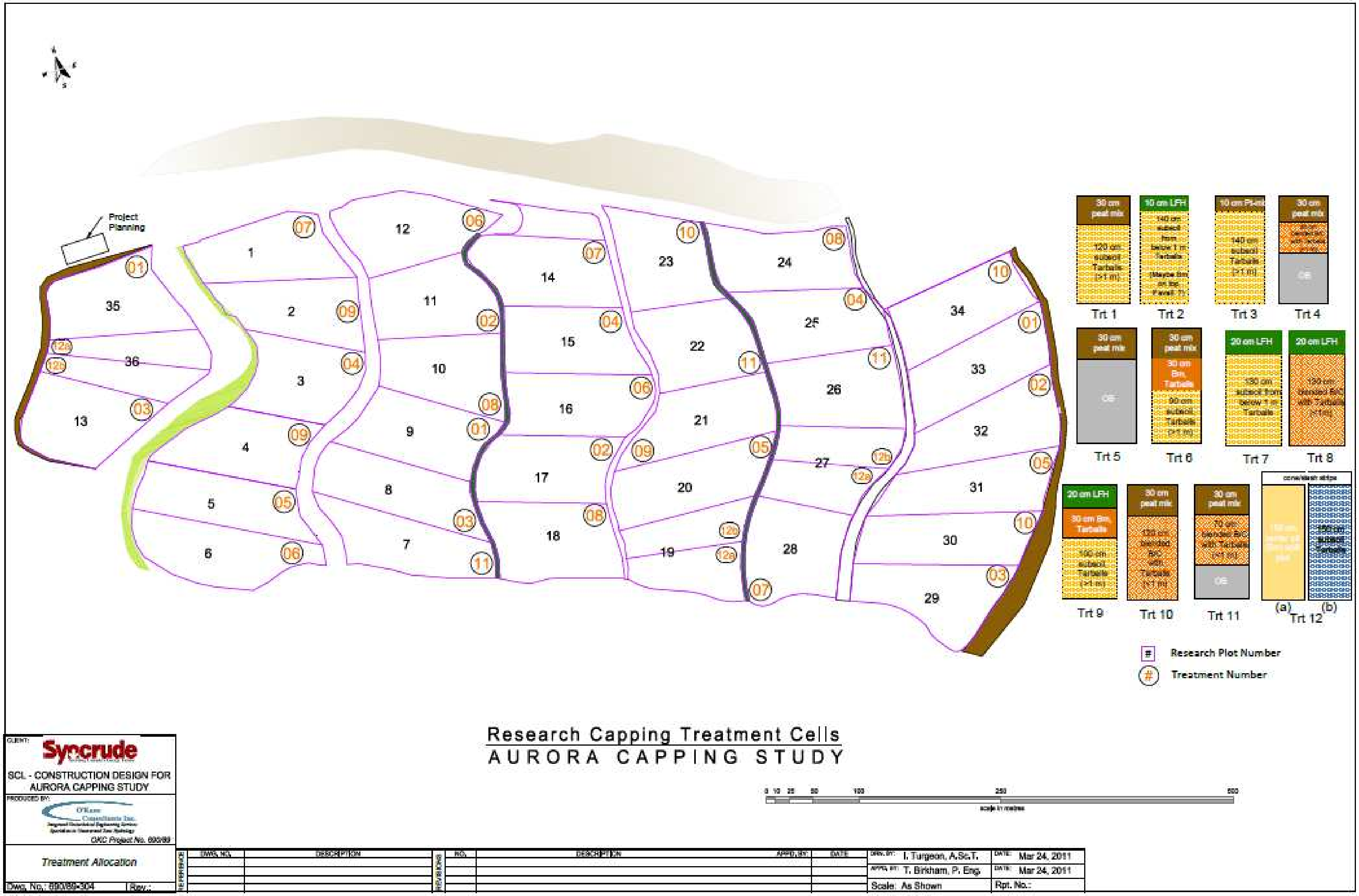


Figure 3.1 – Map of the Aurora Soil Capping Study area at Syncrude, with treatment design legend. From NorthWind Land Resources Inc. (2013).

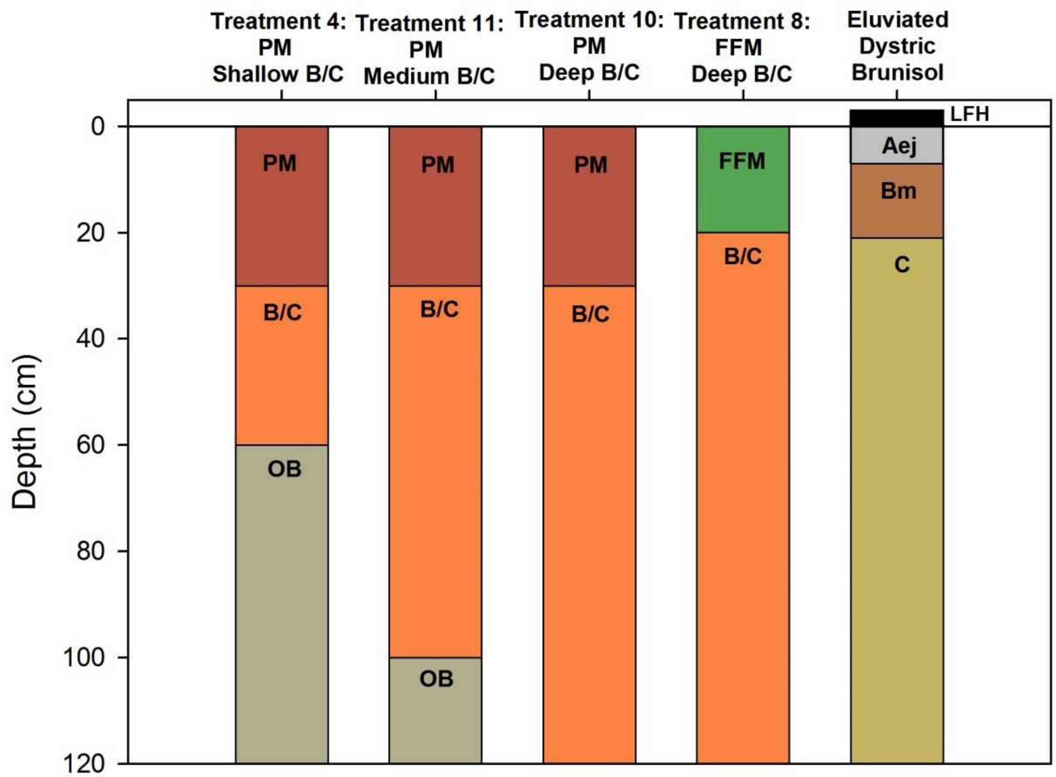


Figure 3.2 – Diagram of the soil treatments from the Aurora Soil Capping Study selected for analysis and used to develop and test the Nutrient Profile Index. PM = Peat-mineral mix topsoil; FFM = Forest floor-mineral mix topsoil; B/C = Salvaged B and C horizon blended material upper subsoil; OB = Overburden lower subsoil material; Brunisol horizons named in accordance with Canadian System of Soil Classification

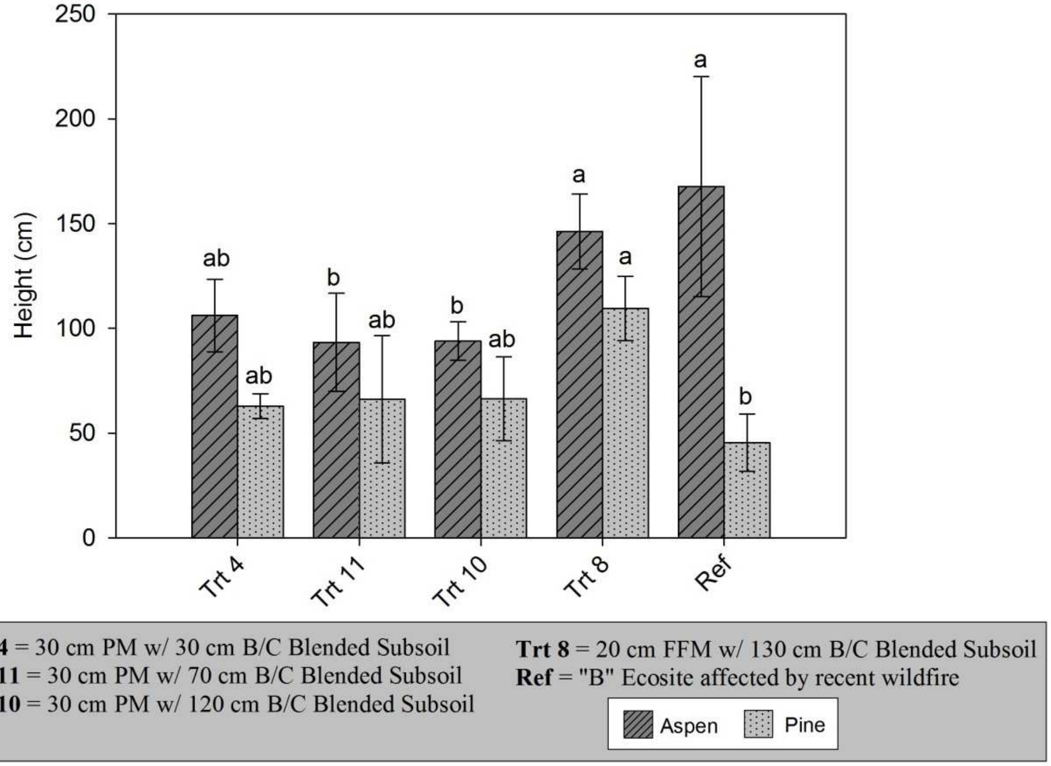


Figure 3.3 – ANOVA of tree heights at study site and natural reference sites. Treatments marked with the same letter do not differ significantly from each other for that species (significance level of $p < 0.05$).

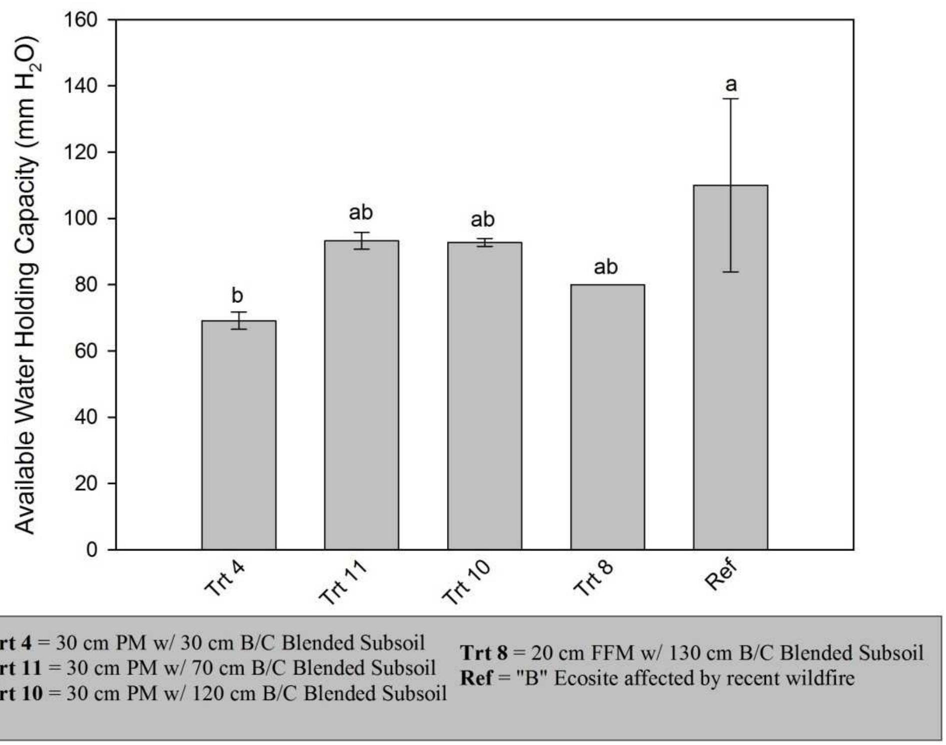


Figure 3.4 – ANOVA of available water holding capacity in top 100 cm of soil at study site and natural reference sites. Treatments marked with the same letter do not differ significantly from each other (significance level of $p < 0.05$).

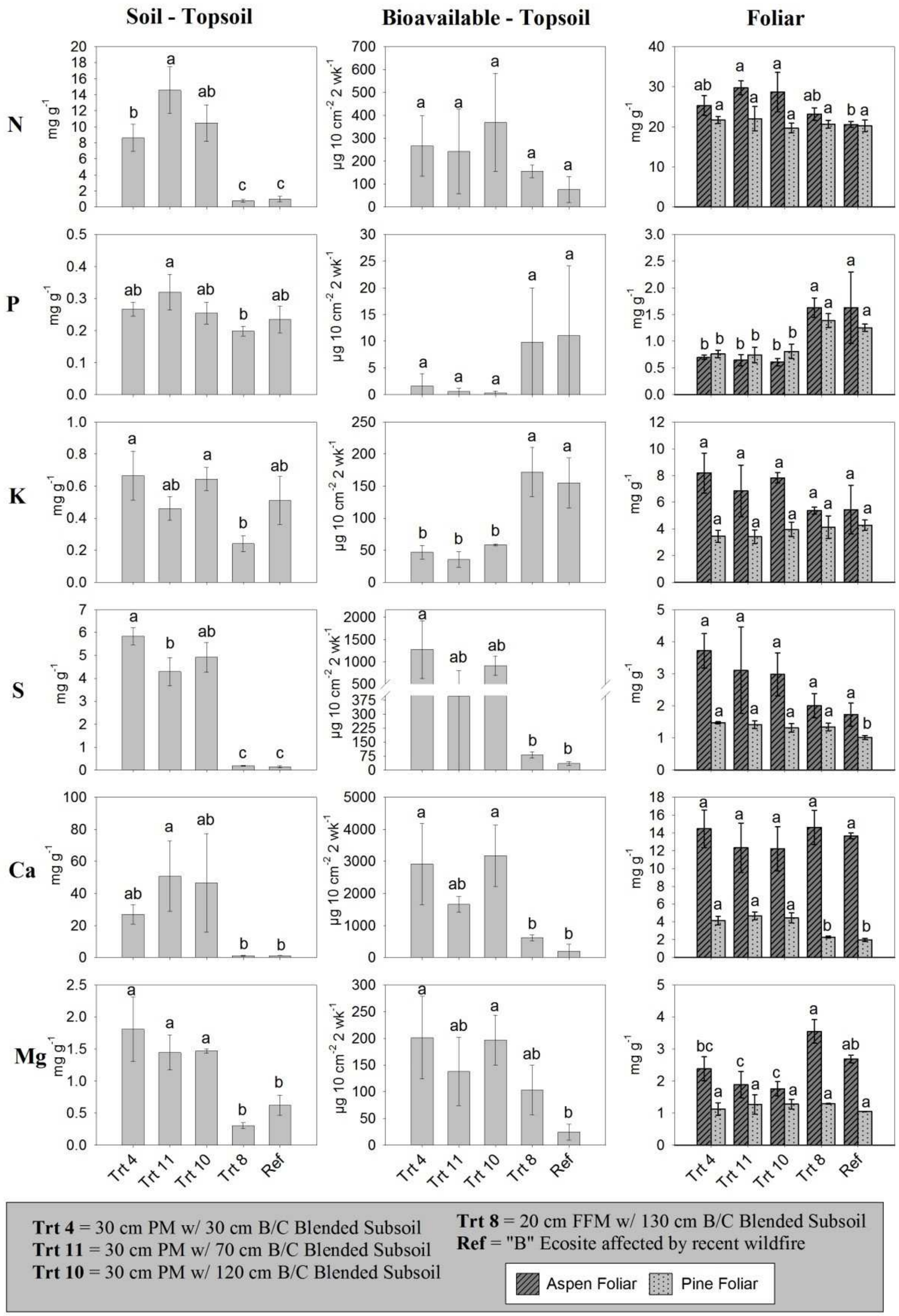


Figure 3.5A – ANOVA of nutrient levels in topsoil soil and bioavailable nutrient pools and foliar nutrient pool from different mixed species reclamation treatments at study site and natural reference sites. Treatments with a non-normal distribution were compared using the Kruskal-Wallis test instead of ANOVA. Treatments marked with the same letter do not differ significantly from each other for that nutrient in that pool (significance level of $p < 0.05$). Foliar significance levels apply only within each species.

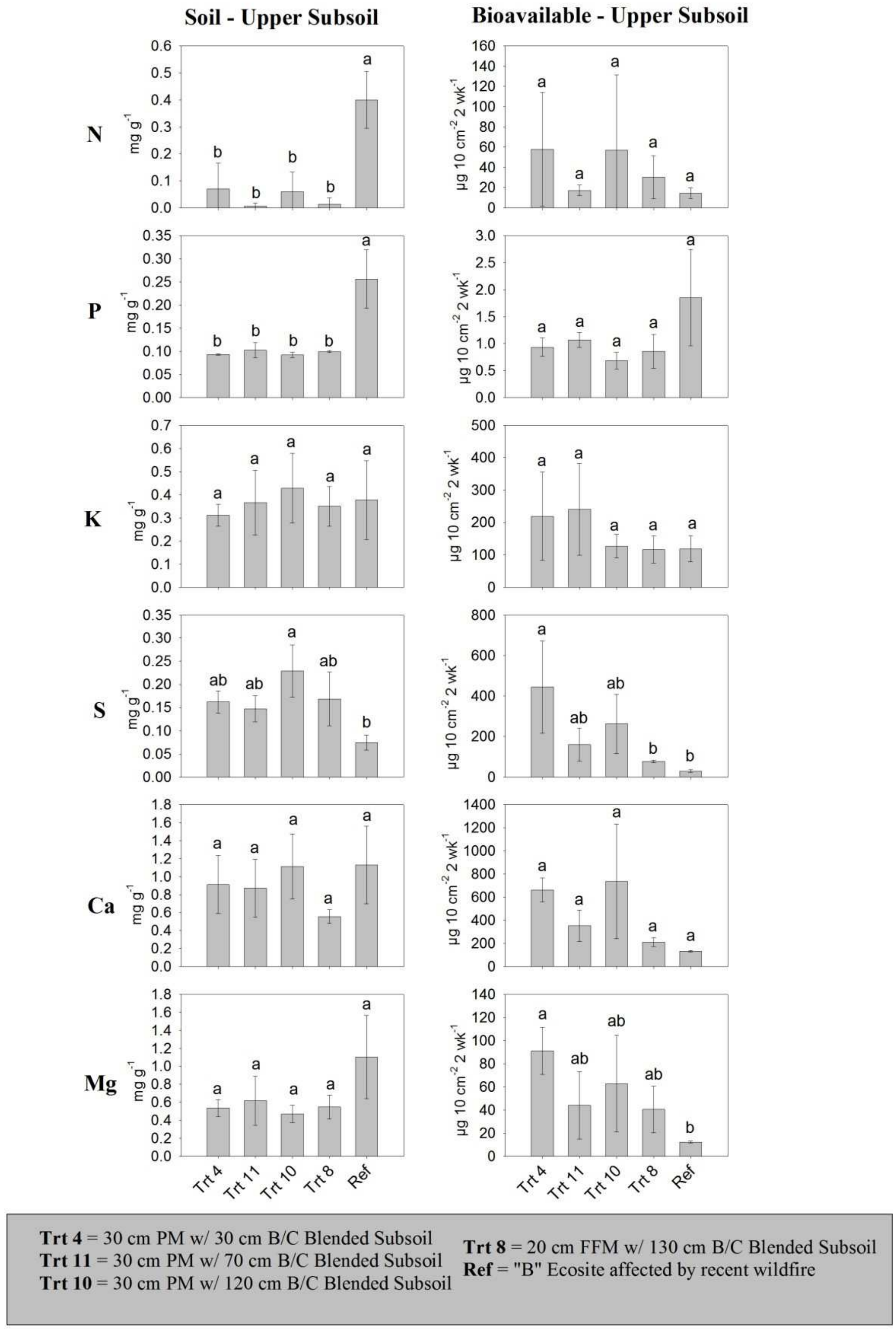


Figure 3.5B – ANOVA of nutrient levels in upper subsoil soil and bioavailable nutrient pools from different mixed species reclamation treatments at study site and natural reference sites. Treatments with a non-normal distribution were compared using the Kruskal-Wallis test instead of ANOVA. Treatments marked with the same letter do not differ significantly from each other for that nutrient in that pool (significance level of $p < 0.05$).

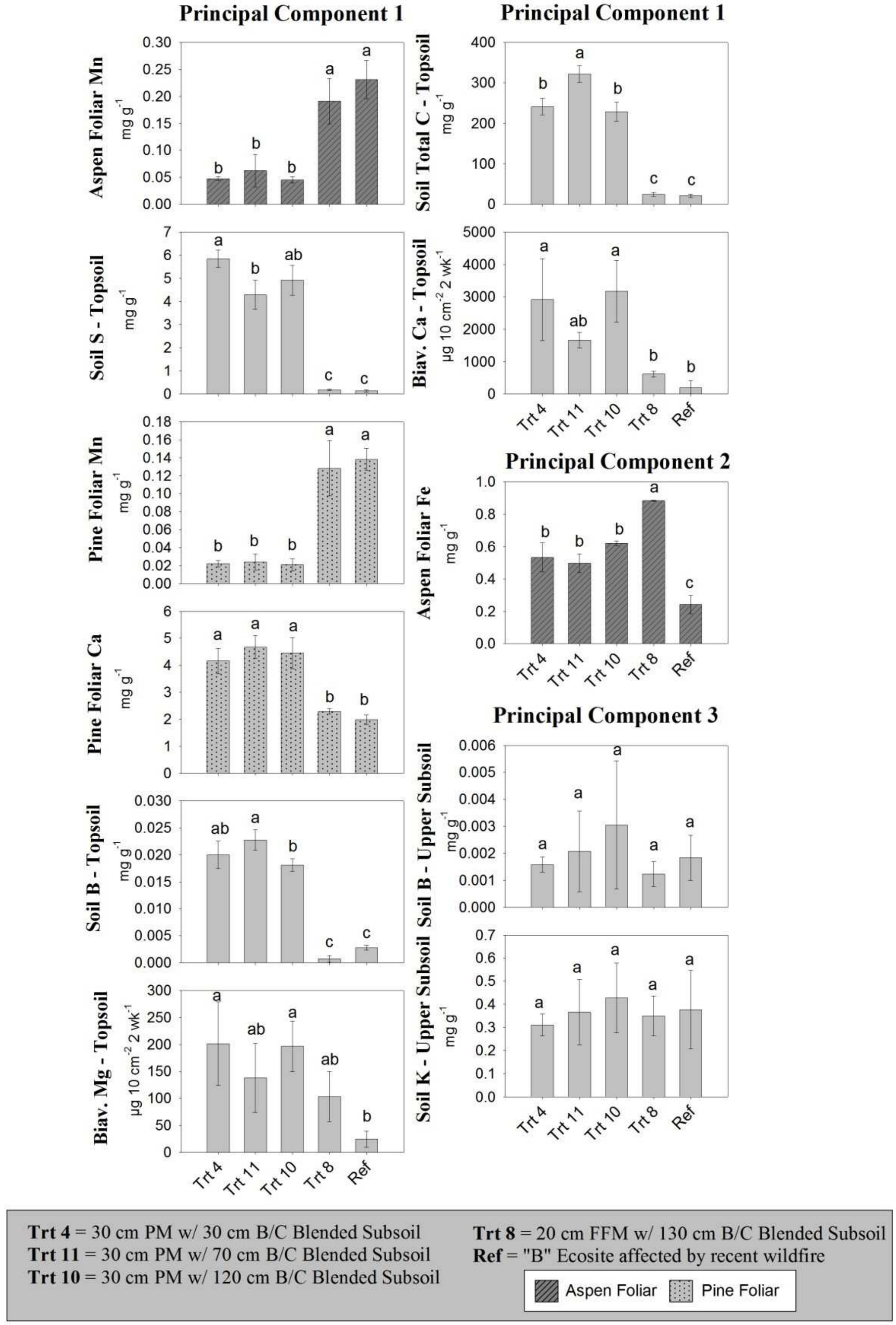


Figure 3.6A – ANOVA of nutrient levels of major components of Nutrient Profile Index Equation, as determined by PCA, from different mixed species reclamation treatments at study site and natural reference sites. Treatments marked with the same letter do not differ significantly from each other for that nutrient in that pool (significance level of $p < 0.05$).

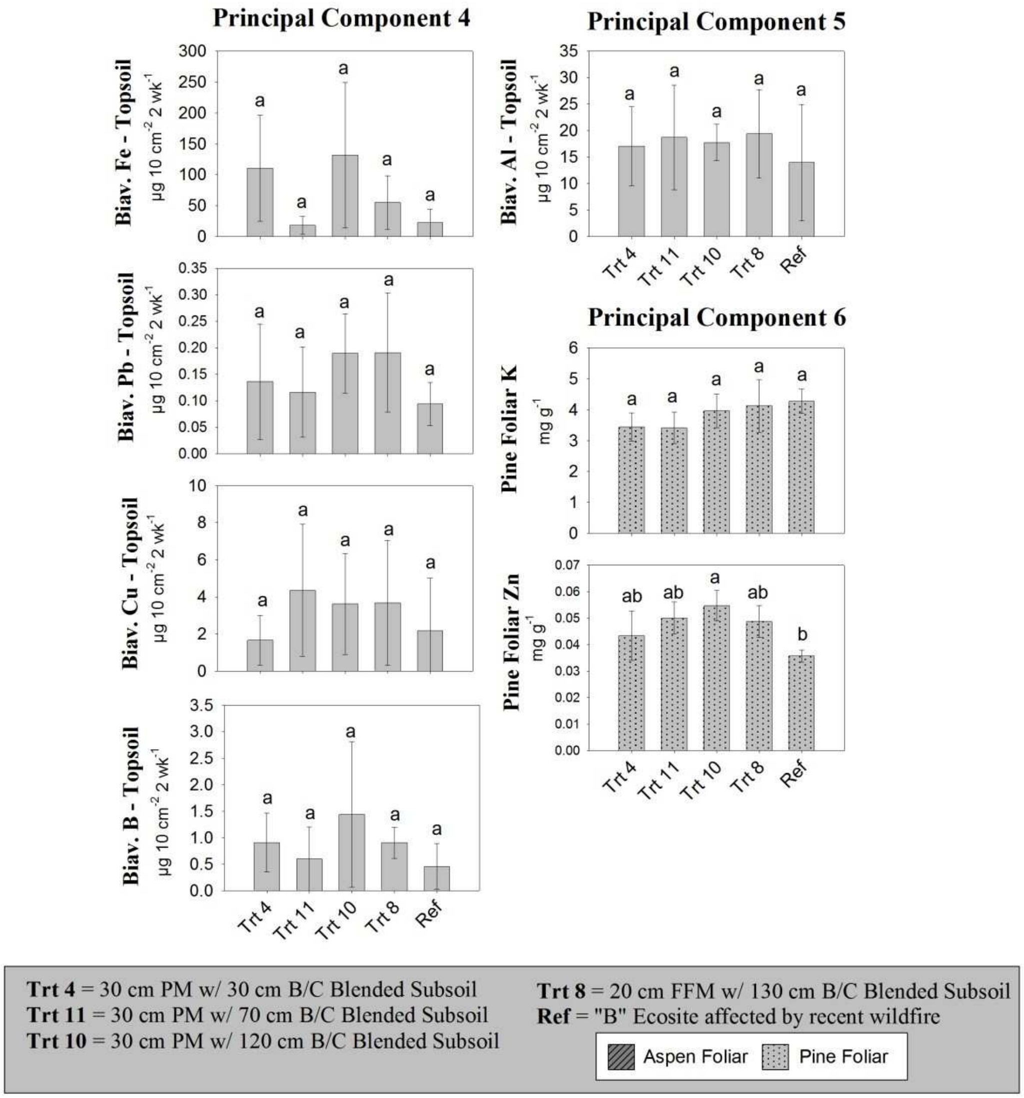


Figure 3.6B – ANOVA of nutrient levels of major components of Nutrient Profile Index Equation, as determined by PCA, from different mixed species reclamation treatments at study site and natural reference sites. Treatments marked with the same letter do not differ significantly from each other for that nutrient in that pool (significance level of $p < 0.05$).

4.0 Technology Transfer and Study Implications

4.1 Introduction

In order to summarize and communicate research findings with industrial and regulatory partners, this chapter will compile the major points from these studies to contribute to the development of future research and best management practices in the AOSR. The central objective of land reclamation is to return disturbed land from a degraded state into one that is more capable of supporting end land-use goals (Jacobs et al. 2015). These end land-use goals are generally a mixture of multiple purposes, primarily ecological and economic (Jacobs et al. 2015). While it is generally agreed that disturbed landscapes should not be left in a degraded state with minimal ecological function, different stakeholders will have different values that they place on the objectives of land reclamation. In the AOSR, with the requirement of equivalent land capability, the specific goal of land reclamation is to re-establish ecosystems that are similar to the ecosystems that existed prior to industrial disturbance while capable of supporting any economic interests in that area. Ultimately, the goal is to reduce the amount of time it takes for an ecosystem to be capable of sustaining itself. Eventually, these sites will ideally begin to truly resemble natural forests, not just on an aesthetic level, but also in terms of underlying ecological function.

It is likely that reclaimed ecosystems, rather than completely returning to the ecosystem configuration that existed prior to disturbance, will form novel forest ecosystems on the reclaimed sites. This is not necessarily in violation of the objectives of land reclamation, as long as the reclaimed site has a similar capability to original ecosystem and integrates sensibly with the surrounding ecosystems. In face of changing climate circumstances, it may be unreasonable to expect that the reclaimed ecosystem will be identical to an ecosystem that was removed from a site decades prior. It is here that current undisturbed ecosystems, analogous to the removed ecosystem, are useful comparison points, as they represent what that prior ecosystem might resemble had it been allowed to continue developing under the changing environmental conditions. The effectiveness of reclamation efforts by practitioners can depend upon the reliability of information that is being used to determine whether the reclamation is proceeding along an acceptable trajectory, regardless of what specific trajectory the reclamation follows.

4.2 – Reclamation Materials

In both the CNRL Horizon study and the ASCS study, sites capped with FFM materials were found to be more similar to natural reference sites. This is not unexpected, as the natural reference sites are themselves upland forest ecosystems, so the upland-derived FFM topsoil caps provide a soil environment that is more similar to these sites than a PM topsoil cap would, along with more ecologically-appropriate biological and substrate legacies (McMillan et al. 2007; Mackenzie and Naeth 2010; MacDonald et al. 2015b). PM is undoubtedly still capable of supporting desired tree and understory species on the sites, as evidenced by the fact that aspen, pine, spruce, and other species still do grow on those soils and that foliar nutrient concentrations are similar to or exceed those found in the natural reference sites. Peat is often used in the horticultural industry as a medium for plant growth, and it has many beneficial aspects, including low bulk density, high water-holding capacity and high nutrient concentrations of N, S, Ca, and Mg, thanks to its high soil organic matter content. While PM can support the growth of plant communities, it is uncertain how resilient ecosystems on PM sites will be when encountering natural disasters such as wildfire, or anthropogenic disturbances such as timber harvesting. PM's high organic matter content means that fire could damage the structure and stability of the soil, which could lead to losses of the material itself and a reduced recovery of plant propagules. For that matter, while upland forest soils have been demonstrated to have resilience following wildfires in upland forests, it is uncertain if FFM will behave in exactly the same manner, or if it will be incapable of facilitating a natural regeneration. My studies supported the findings in literature that FFM is more similar to native upland forest soils with respect to nutrient availability than PM. One possible way to address the lack of biological and substrate legacies in PM is to add analogues of these legacies as soil amendments. Planting different tree and understory species acts as a replacement for the propagules that would otherwise be found in an upland forest soil. For substrate legacies, incorporating biochar into PM topsoil may act as a replacement for the charcoal that would be naturally added to a forest soil following a wildfire (Howell 2015).

Fertilization with N-P-K-S fertilizer seems to have had little impact on overall similarity of all three nutrient pools combined between reclamation sites and natural reference sites. However, my study focused on only one year's worth of data from a limited selection of sites from within a larger study, and it would be negligent to draw conclusions from this work alone. Additionally, the purpose of fertilization is not to increase similarity to natural references, but rather to create an initial surge of nutrients that can be taken up by plants, providing the resources to create a strong initial permanent plant presence for the regenerating ecosystem. In a year-to-year analysis from the initial plot establishment, Howell et al. (2017) suggest that there are more benefits to be derived from the fertilization of PM than FFM, and that fertilization regimes should be tailored specifically to each site in order to achieve the maximum increase in overall similarity. As with Howell (2015)'s study, P may be the most limiting nutrient in PM, and so fertilization efforts should be focused on increasing P levels in soil. Fertilization did not consistently improve the correlation between foliar and belowground nutrient pools for all nutrients. This includes both univariate correlation of individual nutrients and the collective multivariate correlation, as assess with the Mantel Test. More meaningful properties to examine when determining the positive or negative impact of fertilization may include the growth of desired species compared to an unfertilized control, both in terms of biomass and cover. Differences in the species composition and distribution on fertilized and unfertilized sites would also be important to examine, to determine if the surge of nutrients was taken up by desired native species, or whether opportunistic invaders out-competed those desired species, which may make it difficult to shift the plant community back towards that of the natural ecosystem (Davis et al. 2000).

From the ASCS, it appears that a B/C Blended subsoil horizon that is 70 cm thick yields a more similar nutrient profile to the natural reference sites than either the 120 cm thick horizon or the 30 cm horizon, though not significantly more so. It is possible that too thin of a subsoil horizon does not provide optimum water and nutrient availability for the growing plants (Naeth et al. 2011; Jung et al. 2014). Conversely, too thick of a subsoil horizon may decrease similarity to the natural reference sites because the lack of an impermeable barrier immediately beneath the plants' rooting zone may allow water to infiltrate deeper than the plants can readily access. This could also lead to soil nutrients leaching deeper into the profile and becoming inaccessible as well. This reinforces the need for efficient use of reclamation materials, both in ecological and economic terms. Ultimately, more research would be required to investigate the specific mechanisms for the underlying differences between the sites. Examining development of the plant community itself over time might provide more insight into whether or not there are any differences in reclamation success due to subsoil placement depth.

4.3 – Assessing Reclamation Success

A forest is more than just how much biomass its trees can produce. The process of restoring a forest ecosystem is not as simple as only increasing the supply of all nutrients and trusting that will be sufficient to restart the complex interactions and cycles found in a native, undisturbed forest (Oliet and Jacobs 2012; Jacobs et al. 2015). For most nutrients, the natural reference sites had the lowest amounts between the different treatments. Ideally, rather than merely attempting to maximize nutrient concentrations in the soil, some attempt can be made to put the nutrient concentrations into some sort of context. One method for this is to use the biological benchmarking approach, comparing the reclaimed site to an equivalent natural site (MacDonald et al. 2012). This principle ties into the use of equivalent land capability as the driving guideline for land reclamation. It can be difficult, however, to find a suitable comparison site. Spatial and temporal differences between sites can have significant implications for the sort of soil that is present on a site or the type of ecosystem that a site can support. For instance, judging a juvenile site against a fully mature site would be misguided, as the nutrient dynamics of an ecosystem change as different seral communities colonize the site. This means that early reclamation site nutrient dynamics will likely differ greatly from mature site nutrient dynamics (Wang and Klinka 1997; Mukhopadhyay et al. 2014). Given the high variability of soil and environmental conditions, even across short distances, this is also a problem when using nearby sites for comparison.

In order to construct a reclamation framework that accurately represents the similarity of a reclaimed site to a natural environment, more data should be considered than just water, total soil C, total soil N, and total soil P. Even if N and P are most likely to be the two most limiting nutrients in a boreal forest environment (Gower et al. 2000), there are still important implications if other nutrients become limiting. Assuming that those nutrients will never become limiting and thus not monitoring for them in the soil means that if they were to become limiting, there would be no way of knowing until the plants started showing signs of deficiency. When using topsoil materials with a large quantity of organic matter, such as PM, it is more likely that the lithologically-derived nutrients, such as K, will become limiting. As demonstrated by the results of the PCA on the ASCS sites, plant macronutrient concentrations may appear relatively similar while micronutrients such as Mn play a major role in forming the clearest axis along with sites can be differentiated. Additionally, the use of only a single axis for changes in total nutrition, like the system found in the LCCS, may not reveal underlying dynamics of what makes a reclaimed site more similar to a natural site. While the idea of a "limiting" nutrient in a natural ecosystem is not as straightforward as in a controlled environment, such as agriculture, lower levels of a nutrient could reduce the ability of an ecosystem to support certain species in a plant community, or a certain percentage of composition of that community. This may shift the community on a reclaimed site, and therefore the ecosystem, away from one that would be found naturally within the same region. By examining the plant nutrient pool itself, and output metrics such as growth or biomass production rates, it can be determined if these nutrients are only different between sites, or whether they are actually limiting ecosystem development.

The inclusion of multiple nutrient pools in assessment of differences between sites will also lead to far more meaningful information being provided for decision-making. While foliar nutrient concentrations provide useful information about the current status of plant health on a site, they are less useful in forecasting the short-term or long-term supply of nutrients on a site, so long as the minimum nutrient demands of the plants are being met. By assessing the bioavailable and soil nutrient pools directly, reclamation practitioners will be able to make predictions about the development of the ecosystem along the reclamation trajectory that would otherwise be impossible based on foliar nutrient concentrations alone. Furthermore, as plants regulate their internal nutrient concentrations, foliar nutrient concentrations alone may not provide enough discretionary power between different reclamation practices to make meaningful commentary on the relative strengths of one in relation to another. Foliar nutrient concentrations are good indicators in specific situations, such as agriculture, where a stronger knowledge base has been built and maintained; and as the knowledge base is added to for wildland ecosystems, they may become more appropriate to use in reclamation monitoring. However, at this current point in time, it is more practical to base assessment tools on methods that can quickly adapt to the variations in natural environments. This is where the use of a similarity index, such as that found in the mine soil quality index (Mukhopadhyay et al. 2014), becomes more helpful for creating those frames of reference.

4.4 – Ecosystem Functional Similarity Index

In the context of being useful as a system by which land managers and regulatory agencies could employ to determine reclamation success, this represents an early step towards the creation of an Ecosystem Functional Similarity Index (EFSI). In terms of effectiveness, clear boundaries between acceptable and unacceptable would have to be established – How similar is similar enough? Statistical analysis could help determine if there are meaningful differences between groups, but it would be preferential to have straightforward cutoff values. Paired with this would be the necessity of definitively distinguishing between what would be acceptable condition for a reclaimed site and what is not. While differing reclamation trajectories could lead to differences in similarity, there would likely be certain outcomes that are too dissimilar from the target ecosystem to have equivalent land capability, but functional ecosystems of some sort would still exist on those sites. Does it become a matter of linking the site to another reference ecosystem type, or does it become necessary to restart the long process of reclamation? If reclamation trajectory can be determined earlier in the reclamation cycle, it would mean less effort being wasted on failed attempts while enough information is being collected to demonstrate the failure of the attempt.

In terms of soundness, all data entered into the index would be based on sound existing science, but could the combination of all of that information be said to have the same validity? All data collected would be measurable and should be replicable. Data would likely be gathered using the same individual methods that are currently in use. The key question for the soundness of the EFSI would be if all of the indicators combined into the ordination would be responsive and whether or not there would be a high amount of noise in the data. Data reduction methods, such as PCA can help filter out this noise, but this requires a strong statistical backing. Optimally, more individual plots could be sampled for each site type to reduce the impact of outliers. However, this does create more work in both the field and lab for reclamation practitioners and increases the cost of sample collection, storage, and analysis.

The practicality of the system may be the hardest aspect to convince land managers and regulators of. While the technology and methods exist for collecting individual measurements, and the end result of equivalent land capability would absolutely be in line with EPEA regulations, there would likely be a lot of skepticism surrounding whether it was more technically feasible and reasonable to have to feed data through specialized software rather than being tallied by a field technician using a guidebook. There would be questions both in terms of the extra time that it would take and extra costs it would invoke, for the software and training time to operate it and interpret output.

As it stands, another restriction on moving forward with the EFSI system is that foliar nutrient concentration would ideally be replaced by some form measurement that can account for the entire nutrient content within the plant community on a site. Foliar concentration without the context of how that concentration is distributed across the site in terms of plant biomass is not necessarily reflective of the entire plant nutrient pool; given that plant nutrient concentration is variable depending on the species and growing conditions (Ingestad and Åagren 1988; Munson and Timmer 1995). Foliar nutrient concentration cannot be as simply scaled up to content as the soil or bioavailable nutrient concentrations. Both trees and understory species, which would also ideally be incorporated into a full EFSI, contain nutrients that were extracted from the soil, so to ignore one category in favour of another would be to discount the fate of a potentially large portion of belowground nutrient content. The density of the plant community at the site would need to be accounted for just as the size of the individual plants would have to be factored into any allometric equations used to estimate total plant biomass.

The soil mine quality index method that was the basis for the NPI has an aim of obtaining the minimum dataset necessary to reliably determine differences between sites (Masto et al. 2008). As their factors were linked to agricultural sites, they were able to simplify their data for individual variables through the use of a scoring function that rated suitability of a property from 0 – 1. This would be useful in determining practical differences between sites, as opposed to larger differences that have little functional impact on plant growth. This could help reduce statistical noise and make sure that the most important differences are highlighted by the PCA. However, as previously mentioned, wildland ecosystems do not lend themselves as easily to having clear critical values where plant growth is necessarily optimized, due to their highly variable nature with high levels of competition between plants. Another question that can be raised with this is whether or not it is appropriate to treat all variables as equal. Applying a weighting system to the variables based on ecological principles may help to limit the impact of otherwise minor ecosystem components, such as micronutrients with inconsistent concentrations between sites. However, this leads to a similar problem as with the current CIF recommended indicators: potentially ignoring of useful information because it is not expected to be useful. Furthermore, while reducing data to the minimum dataset is useful for managing the amount of information that is necessary to interpret, the question arises as to whether or not it is appropriate to discard variables that are definitely strongly correlated to principal component axes because they are not the most strongly correlated variables. Perhaps the best approach that could be recommended for these variables is to work on improving their levels similarity to the natural reference sites as well in order to ensure as many potential sources of dissimilarity are being address during corrective actions. Addressing these issues and creating the best system of indicating reclamation success for the AOSR will absolutely require further research.

4.5 – Conclusion

Ecosystems are complex, and it may require complex data in order to properly understand the dynamics that drive them. That being said, complex does not necessarily mean complicated. Current soil quality assessment criteria and indicators are sold on simplicity, but oversimplify or ignore information that would be necessary to make informed land management decisions for reclaimed wildland ecosystems. In addition to being rooted in agricultural thinking, the use of only foliar nutrient concentrations as a metric for reclamation success ignores many basic elements of plant physiology and the complex nature of the plant-soil-microbial continuum that drives nutrient flow through the different nutrient pools in an ecosystem. It would be far better to have a system that can accurately compare the similarity of reclaimed ecosystems to natural ecosystems, incorporating other aspects. At the very least, expanding criteria for the success of nutrient cycling through different pools to include multiple nutrient pools would be a good first step. Reclamation research continues to progress greatly since its earliest incarnations. Each new piece of information helps to mitigate negative impacts imposed on the environment by resource extraction activities. While the meaning of new results are often debated extensively before incorporation into practices if there is not a clear economic incentive to change methodology, the partnership between industry, academia, and government remains fruitful and continues to work towards the goal of improving the systems that will improve the environment. This research is important not only for local disturbances, such as oil sands mining, but many ultimately help improve reclamation practices throughout the world. With a system such as the NPI and eventually the EFSI, rather than a need to continually research and define new acceptable standards for metrics in different ecosystems, a target reference site type can be designated and used to anchor target values for different variables included in the index. This makes this sort of system flexible and readily adaptable. It is my hope that my studies here will contribute to improved land management in the AOSR and wherever disturbed land requires assessment for reclamation success.

Literature Cited

Alberta Environment. 2010. Guidelines for reclamation to forest vegetation in the Athabasca Oil Sands Region. Prepared by the Terrestrial Subgroup of the Reclamation Working Group of the Cumulative Environmental Management Association, Fort McMurray, Alberta.

Alberta Environment and Sustainable Resource Development [AESRD]. 2013. Criteria and indicators framework for oil sands mine reclamation certification. Prepared by Mike Poscente and Theo Charette of Charette Pell Poscente Environmental Corporation for Reclamation Working Group of the Cumulative Environmental Management Association, Fort McMurray, AB. September 18, 2012.

Asher, C. J. 1978. Natural and synthetic culture media for Spermatophytes. CRC Handb. Ser. Nutr. Food, Sect. G 3, 575-609.

Baker, D. E., Suhr, N. H. 1982. Atomic absorption and flame emission spectrometry. In A. L. Page (Ed), Methods of Soil Analysis. Part 2. Chemical and Microbiological Properties, 2nd Edition, (pp 13-27, 229-231, 238-239, 257-258, 294-295, 302-304, 314-315, 326-327). American Society of Agronomy, Inc., Soil Science Society of America, Inc., Madison, Wisconsin, USA.

Ballard, T. M. and Carter, R. E. 1986. Evaluating Forest Stand Nutrient Status. Land Management Report No. 20. British Columbia Ministry of Forests, Victoria, B.C.

Barber, L., J. Bockstette, D. Christensen, L. Tallon, and S. Landhäusser. 2015. Effect of soil cover system design on cover system performance and early tree establishment. In A. B. Fourie, M. Tibbett, L. Sawatsky, and D. van Zyl, editors. Mine Closure 2015. InfoMine Inc., Vancouver, BC, Canada.

Bayfield, N. G. 1996. Long-term changes in colonization of bulldozed ski pistes at Cairn Gorm, Scotland. J. Appl. Ecol. 33 (6), 1359-1365.

Beckingham, J. D., Archibald, J. H., 1996. Field Guide to Ecosites of Northern Alberta. Natural Resources Canada, Canadian Forest Service, Northern Forestry Centre. Special Report 5.

Boerner, R. E. J. 1984 Foliar nutrient dynamics and nutrient use efficiency of four deciduous tree species in relation to site fertility. J. Appl. Ecol. 21, 1029-1040.

Bradshaw, A. 1997. Restoration of mined lands – using natural processes. Ecol. Eng. 8(4), 255-269.

Brejda, J. J., Moorman, T. B., Karlen, D. L., Dao, T. H. 2000. Identification of regional soil quality factors and indicators: I. Central and southern high plains. Soil Sci. Soc. Am. J. 64(6), 2115-2124.

Brockley, R. P. 2001. Foliar sampling guidelines and nutrient interpretative criteria for lodgepole pine. Extension Note 52, British Columbia Ministry of Forests, Victoria, Canada.

Brosofke, K. D., Chen, J., Crow, T. R. 2001. Understory vegetation and site factors: implications for a managed Wisconsin landscape. For. Ecol. Manag. 146 (1-3), 75-87.

Brown, R. L. and Naeth, M. A. 2014. Woody debris amendment enhances reclamation after oil sands mining in Alberta, Canada. Restor. Ecol. 22 (1), 40-48.

Canadian Association of Petroleum Producers. 2016. Land reclamation. Retrieved from http://www.canadasoilsands.ca/en/explore-topics/land-reclamation. [Verified December 12, 2016].

Chen, H. Y. H., Klinka, K., Kabzems, R. D. 1998. Site index, site quality, and foliar nutrients of trembling aspen: relationships and predictions. Can. J. For. Res. 28(12), 1743-1755.

Chang, S. X., Preston, C. M., McCullogh, K., Weetman, G. F., Barker, J. 1996. Effect of understory competition on distribution and recovery of ^{15}N applied to a western red cedar-western hemlock clear-cut site. Can. J. For. Res. 26 (2), 313-321.
Chapin, F. S. III. 1980. The mineral nutrition of wild plants. Annu. Rev. Ecol. Syst. 11, 233-260.
Ciccarese, L., Mattsson, A., Pettenella, D. 2012. Ecosystem services from forest restoration: thinking ahead. New Forests 43, 543-560.
Cumulative Environmental Management Association. 2006. Land capability classification system for forest ecosystems in the oil sands. Edmonton, Alberta.
Davis, M. A., Grime, J. P., Thompson, K. 2000. Fluctuating resources in plant communities: a general theory of invasibility. J. Ecol. 88, 528-534.
De Mello Prado, R., Caione, G. 2012. Plant Analysis. In Issaka, R. (Ed.) *Soil Fertility* (pp 115-132). InTech.
Environment Canada. 2016. Canadian Climate Normals 1981-2010, Fort McMurray Station Data. Accessed from: http://climate.weather.gc.ca/, [Verified June 16, 2016].
Failing, L. and Gregory, R. 2003. Ten common mistakes in designing biodiversity indicators for forest policy. J. Environ. Manag. 68 (2), 121-132.
Freeden, A. L., Rao, I. M., Terry, N. 1989. Influence of phosphorus nutrition on growth and carbon partitioning in *Glycine max*. Plant Physiol. 89 (1), 225-230.
Gonzalez, E., Muller, E., Comin, F. A., Gonzalez-Sanchis, M. 2010. Leaf nutrient concentration as an indicator of *Populus* and *Tamarix* response to flooding. Perspect. Plant Ecol. Evol. Syst. 12 (4), 257-266.
Gower, S.T., Hunter, A., Campbell, J., Vogel, J., Veldhuis, H., Harden, J., Trumbore, S., Norman, J. M., Kucharik, C. J. 2000. Nutrient dynamics of the southern and northern BOREAS boreal forests. Ecoscience 7 (4), 481-490.
Hankin, S. L., Karst, J., Landhäusser, S. M. 2015. Influence of tree species and salvaged soils on the recovery of ectomycorrhizal fungi in upland boreal forest restoration after surface mining. Botany 93 (5), 267-277.
Hansen, E.A., 1994. A guide for determining when to fertilize hybrid poplar plantations. USDA For. Serv., North Central For. Exp. Sta., Res. Pap. NC-319, 7 pp.
Haynes, R. J. and Naidu, R. 1998. Influence of lime, fertilizer and manure applications on soil organic matter content and soil physical conditions: a review. Nutr. Cycl. Agroecosyst. 51, 123 – 137.
Hendershot, W. H., Duquette, M. 1986. A simple barium chloride method for determining cation exchange capacity and exchangeable cations. Soil Sci. Soc. Am. J. 50 (3), 605-608.
Hogberg, J. I., Pinno, B. D., MacKenzie, M. D. Manuscript in Review. Evaluating foliar nutrient concentration as an indicator of soil nutrient concentrations in natural and reclaimed forests in Alberta, Canada.
Howell, D. M. 2015. Influence of amendments and soil depth on available nutrients and microbial dynamics in contrasting topsoil materials used for oil sands reclamation. MSc thesis. University of Alberta, Edmonton, Alberta.
Howell, D. M., Das Gupta, S., Pinno, B. D., MacKenzie, M. D. 2017. Reclaimed soils, fertilizer, and bioavailable nutrients: Determining similarity with natural benchmarks over time. Can. J. Soil Sci. In Press.
Hu, Y.-L., Hu, Y., Zeng, D.-H., Tan, X., Chang, S. X. 2014. Exponential fertilization and plant competition effects on the growth and N nutrition of trembling aspen and white spruce seedlings. Can. J. For. Res. 45 (1), 78-86.

Ingestad T, Åagren G. I. 1988. Nutrient uptake and allocation at steady-state nutrition. Physiologia Plantarum 72(3), 450-459.

Jacobs, D. F., Oliet, J. A., Aronson, J., Bolte, A., Bullock, J. M., Donoso, P. J., Landhäusser, S. M., Madsen, P., Peng, S., Rey-Benayas, J. M., Weber, J. C. 2015. Restoring forests: What constitutes success in the twenty-first century?. New Forests 46, 601-614.

Johnson, D.W., Verburg, P., Arnone, J., 2005. Soil extraction, ion exchange resin, and ion exchange membrane measures of soil mineral nitrogen during incubation of a tallgrass prairie soil. Soil Sci. Soc. Am. J. 69, 260-265.

Johnson, E. A., Miyanishi, K., Weir, J. M. H. 1995. Old-growth, disturbance, and ecosystem management. Can. J. Bot. 73 (6), 918-926.

Jung, K., Duan, M., House, J., Chang, S. X. 2014. Textural interfaces affected the distribution of roots, water, and nutrients in some reconstructed forest soils in the Athabasca oil sands region. Ecol. Eng. 64, 240-249.

Kaiser, H. F. 1960. The application of electronic computers to factor analysis. Educ. Psychol. Meas. 20(1), 141-151.

Kalra, Y. P., Maynard, D. G. 1991. Methods manual for forest soil and plant analysis. Northern Forestry Centre. Edmonton, AB.

Kessler, S. Barbour, S. L., van Rees, K. C. J., Dobchuk, B. S. 2010. Salinization of soil over saline-sodic overburden from the oil sands in Alberta. Can. J. Soil Sci. 90(4), 637-647.

Kruskal, W.H. 1952. A nonparametric test for the several sample problem. Ann. Math. Stat. 23 (4), 525-540.

Lafleur, B., Paré, D., Claveau, Y., Thiffault, É., Bélanger, N. 2013. Influence of afforestation on soil: The case of mineral weathering. Geoderma 202-203, 18 – 29.

MacDonald, S. E., Quideau, S. A., Landhäusser, S. M. 2012. Rebuilding forest ecosystems after industrial disturbance. In D. H. Vitt & J. S. Bhatti (Eds.), *Restoration and reclamation of boreal ecosystems: Attaining sustainable development* (pp 123-160). Cambridge University Press.

MacDonald, S. E., Landhäusser, S. M., Skousen, J., Franklin, J., Frouz, J., Hall, S., Jacobs, D. F., Quideau, S. A. 2015a. Forest restoration following surface mining disturbance: challenges and solutions. New Forests 46, 703-732.

MacDonald, S. E., Snively, A. E. K., Fair, J. M., Landhäusser, S. M. 2015b. Early trajectories of forest understory development on reclamation sites: influence of forest floor placement and a cover crop. Restor. Ecol. 23 (5), 698-706.

Mackenzie, D. D., Naeth, M. A. 2010. The role of the forest soil propagule bank in assisted natural recovery after oil sands mining. Restor. Ecol. 18 (4), 418-427.

Mackenzie, D. D. 2013. Oil sands mine reclamation using boreal forest surface soil (LFH) in northern Alberta. PhD Dissertation. University of Alberta, Edmonton, Alberta.

Marschner, H. 2002. Mineral Nutrition of Higher Plants. 2nd Ed. Academic Press Inc. San Diego, CA.

Masto, R. E., Chhonkar, P. K., Singh, D., Patra, A. K. 2008. Alternative soil quality indices for evaluating the effect of intensive cropping, fertilization and manuring for 31 years in the semi-arid soils of India. Environ. Monit. Assess. 136, 419-435.

McCune, B., Grace, J.B., Urban, D.L., 2002. Analysis of Ecological Communities. MjM software design. Gleneden Beach, OR.

McMillan, R. Quideau, S. A., MacKenzie, M. D., Biryukova, O. 2007. Nitrogen mineralization and microbial activity in oil sands reclaimed boreal forest soils. J. Environ. Qual. 36 (5), 1470-1478.
Mengel, K., Horn, D., Tributh, H. 1990. Availability of interlayer ammonium as related to root vicinity and mineral type. Soil Sci. 149 (3), 131-137.
Morrow, L. D., Timmer, V. R. 1981. Intraseasonal growth and nutrient composition of jack pine needles following fertilization. Can. J. For. Res. 11 (3), 696-702.
Mukhopadhyay, S., Maiti, S. K., Masto, R. E. 2014. Development of mine soil quality index (MSQI) for evaluation of reclamation success: A chronosequence study. Ecol. Eng. 71, 10-20.
Munson, A. D., Timmer, V. R. 1995. Soil nitrogen dynamics and nutrition of pine following silvicultural treatments in boreal and Great Lakes-St. Lawrence plantations. For. Ecol. And Manag. 76 (1-3), 169-179.
Naeth, M. A., Chanasyk, D. S., Burgers, T. D. 2011. Vegetation and soil water interactions on a tailings sand storage facility in the Athabasca oil sands region of Alberta, Canada. Phys. And Chem. Of the Earth 36(1-4), 19-30.
National Research Council. 2000. Ecological Indicators for the Nation. Washington, DC: The National Academies Press. Doi: 10.17226/9720. [Accessed from http://www.nap.edu/catalog/9720/ecological-indicators-for-the-nation June 9, 2016].
Natural Regions Committee. 2006. Natural regions and subregions of Alberta. Compiled by D.J. Downing and W.W. Pettapiece. Government of Alberta. Pub. No. T/852.
NorthWind Land Resources Inc. 2013. Summary of work completed by NorthWind Land Resources Inc. for Syncrude Canada Ltd.'s Aurora North Capping Study. 11-381, Calgary, Alberta.
Oil Sands Research and Information Network. 2011. Equivalent Land Capability Workshop Summary Notes. Oil Sands Research and Information Network, University of Alberta, School of Energy and the Environment, Edmonton, Alberta. OSRIN Report No. TR-13. 83 pp.
Oliet, J. A., Jacobs, D. F. 2012. Restoring forests: advances in techniques and theory. New Forests 43, 535-541.
Ordoñez, J. C., van Bodegom, P. M., Witte, J.-P. M., Wright, I. J., Reich, P. B., Aerts, R. 2009. A global study of relationships between leaf traits, climate and soil measures of nutrient fertility. Global Ecol. Biogeogr. 18, 137–149.
Paré, D., Bernier, P., Lafleur, B., Titus, B. D., Thiffault, E., Maynard, D. G., Guo, X. 2013. Estimating stand-scale biomass, nutrient contents, and associated uncertainties for tree species of Canadian forests. Can. J. For. Res. 43, 599–608.
Peck, J. E. 2010. Multivariate analysis for community ecologists: Step-by-step using PC-ORD. MjM software design. Gleneden Beach, OR.
Peres-Neto, P. R., Jackson, D. A., Somers, K. M. 2005. How many principal components? Stopping rules for determining the number of non-trivial axes revisited. Comput. Stat. & Data Anal. 49(4), 974-997.
Pinno, B. D., Belangér, N. 2009. Competition control in juvenile hybrid poplar plantations across a range of site productivities in central Saskatchewan, Canada. New Forests 37, 213-225.
Pinno, B. D., Landhäusser, S. M., MacKenzie, M. D., Quideau, S. A., Chow, P. S. 2012. Trembling aspen seedling establishment, growth and response to fertilization on contrasting soils used in oil sands reclamation. Can. J. Soil Sci. 92 (1), 143-151.

Pinno, B. D., Errington, R. C. 2015. Maximizing natural trembling aspen seedling establishment on a reclaimed boreal oil sands site. Ecol. Restor. 33 (1), 43-50.
Pinno, B. D., Errington, R. C. 2016. Burn severity dominates understory plant community response to fire in xeric jack pine forests. Forests 7(4), article #83.
Powter, C. Chymko, N. Dinwoodie, G., Howat, D., Janz, A., Puhlmann, R., Richens, T., Watson, D., Sinton, H., Ball, K., Etmanski, A., Patterson, B., Brocke, L., Dyer, R. 2012. Regulatory history of Alberta's industrial land conservation and reclamation program. Can. J. Soil Sci. 92(1), 39-51.
Preston, C. M., Trofymow, J. A., Canadian Intersite Decomposition Experiment Working Group. 2000. Variability in litter quality and its relationship to litter decay in Canadian forests. Can. J. Bot. 78(10), 1269-1287.
Purdy, B. G., MacDonald, S. E., Lieffers, V. J. 2005. Naturally saline boreal communities as models for reclamation of saline oil sand tailings. Restor. Ecol. 13 (4), 667-677.
Qian, P., Schoenau, J., 2002. Practical applications of ion exchange resins in agricultural and environmental soil research. Can. J. Soil Sci. 82, 9-21.
R Core Development Team. 2015-2016. R: A Language and Environment for Statistical Computing 3.2.2 ("Fire Safety") – 3.3.0 ("Supposedly Educational") (Eds.) R Foundation for Statistical Computing, Vienna, Austria.
Röing, K., Andrén, O., Mattsson, L. 2006. 'Non-exchangeable' ammonium in soils from Swedish long-term agricultural experiments: Mobilization and effects of fertilizer application. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 56 (3), 197-205.
Rorison, I. H. 1987. Mineral nutrition in time and space. In I. H. Rorison, J. P. Grime, R. Hunt, G. A. F. Hendry, & D. H. Lewis (Eds.) *Frontiers of comparative plant ecology*. New Phytol. {Suppl.} 106, 79-92.
Rowland, S. M., Prescott, C. E., Grayston, S. J., Quideau, S. A., Bradfield, G. E. 2009. Recreating a functioning forest soil in reclaimed oil sands in Northern Alberta: An approach for measuring success in ecological restoration. J. Environ. Qual. 38, 1580-1590.
Sardans, J., Alonso, R., Janssens, I. A., Carnicer, J., Vereseglou, S., Rillig, M. C., Fernández-Martínez, M., Sanders, T. G. M., Peñuelas, J. 2016. Foliar and soil concentrations and stoichiometry of nitrogen and phosphorous across European *Pinus sylvestris* forests: relationships with climate, N deposition and tree growth. Funct. Ecol. 30, 676-689.
Séré, G., Schwartz, C., Ouvrard, S., Sauvage, C., Renat, J.-C., Morel, J. L. 2008. Soil construction: A step for ecological reclamation of derelict lands. J. Soils Sediments 8(2), 130-136.
Shapiro, S. S., Wilk, M.B. 1965. An analysis of variance test for normality (complete samples). Biometrika 52 (3-4), 591-611.
Smethurst, P. J. 2000. Soil solution and other soil analyses as indicators of nutrient supply: a review. For. Ecol. Manag. 138 (1-3), 397-411.
Sorenson, P. T., Quideau, S. A., MacKenzie, M. D., Landhäusser, S. M., Oh, S.-W. 2011. Forest floor development and biochemical properties in reconstructed boreal forest soils. Appl. Soil Ecol. 49, 139-147.
Spearman, C. 1904. The proof and measurement of association between two things. Am. J. Psychol. 15 (1), 72-101.
Staples, T. E., Van Rees K. C. G., and van Kassel C. 1999. Nitrogen competition using ^{15}N between early successional plants and planted white spruce seedlings. Can. J. For. Res. 29 (8), 1282-1289.

Tukey, J. W. 1949. Comparing individual means in the analysis of variance. Biometrics 5 (2), 99-114.

Vanguelova, E., Pitman, R., 2009. Impacts of short rotation forestry on soil sustainability. In: McKay, H. Ed.), Short Rotation Forestry: Review of Growth and Environmental Impacts. : Forest Research Monograph, 2. Forest Research, pp. 37–78.

Van Lierop, W. 1988. Determination of available phosphorus in acid and calcareous soils with the Kelowna multiple-element extractant. Soil Sci. 146 (4), 284-291.

Vitousek, P. M., Porder, S., Houlton, B. Z., Chadwick, O. A. 2010. Terrestrial phosphorus limitation: mechanisms, implications, and nitrogen–phosphorus interactions. Ecol. Appl. 20 (1), 5-15.

Walworth, J. L., Sumner, M. E. 1988. Foliar diagnosis: a review. In B. Tinker & A. Läuchli, (Eds.) *Advances in Plant Nutrition Vol. 3* (pp. 193-241). Praeger, New York.

Wang, G. G., Klinka, K. 1997. White spruce foliar nutrient concentrations in relation to tree growth and soil nutrient amounts. For. Ecol. Manag. 98, 89-99.

Western Ag Innovations. 2015. PRS technology: analysis. Accessed from https://www.westernag.ca/innovations/technology/analysis_units [Verified June 16th, 2016].

Zackrisson, O., Nilsson, M. C., Wardle, D. A. 1996. Key ecological function of charcoal and wildfire in the Boreal forest. Oikos 77 (1), 10-19.

Appendix

Table A.1 – Comparison of aspen foliar nutrient concentrations from reclaimed plots to natural reference sites and average values determined by Paré et al. (2013)'s literature review. Standard deviation of average values reported in brackets. Values outside of 1 standard deviation from the average concentration bolded.

Nutrient	Average Concentration (mg g^{-1}) (Paré)	Reference		FFM			FFMF			PM			PMF		
		mg g^{-1}	% Avg	mg g^{-1}	% Avg	% Ref	mg g^{-1}	% Avg	% Ref	mg g^{-1}	% Avg	% Ref	mg g^{-1}	% Avg	% Ref
N	21.79 (5.58)	18.52	84.99	18.92	86.83	102.16	16.76	76.92	90.50	19.03	87.33	102.75	16.56	76.00	89.42
P	2.13 (0.70)	1.71	80.28	1.50	70.42	87.71	**1.42**	66.67	83.04	**1.42**	66.67	83.04	1.45	68.08	84.80
K	9.62 (4.23)	8.41	87.42	8.30	98.69	98.69	8.04	83.58	95.60	9.14	95.01	108.68	9.76	101.46	116.05
S	N/A	1.61	-	2.74	-	170.19	2.31	-	143.48	4.53	-	281.37	3.68	-	228.57
Ca	11.52 (3.80)	13.36	115.97	**15.80**	137.15	118.83	**16.29**	141.41	121.93	**16.21**	140.71	121.33	**15.55**	134.98	116.39
Mg	2.24 (0.56)	2.12	94.64	**3.76**	167.86	177.36	**3.87**	172.77	182.55	**4.34**	193.75	204.72	**4.18**	186.61	197.17

Table A.2 – Calculation of available water holding capacity (AWHC) in the top 100 cm of soil for plots at the Aurora Soil Capping Study and natural reference plots based on the Land Capability Classification System for Forest Ecosystems in the Oil Sands (Cumulative Environmental Management Association 2006). Treatment means for total AWHC marked with the same letter are not significantly different from each other (significance level of $p < 0.05$).

Treatment	Plot #	TS Material	TS Multiplier (mm H_2O cm^{-1})	TS Depth (cm)	TS AWHC (mm H_2O)	US Material	US Multiplier (mm H_2O cm^{-1})	US Depth (cm)	US AWHC (mm H_2O)	LS Material	LS Multiplier (mm H_2O cm^{-1})	LS Depth (cm)	LS AWHC (mm H_2O)	Total AWHC > 100 cm (mm H_2O)
PM-Shallow BCB	[a]M-3	[b]PM – S	1.2	[d]37.25	44.70	S	0.8	28.25	22.60	[f]OB	0	34.50	0	67.30
	M-15	PM – S	1.2	33.5	40.20	S	0.8	34.75	27.80	OB	0	31.75	0	68.00
	M-25	PM – S	1.2	39.25	47.10	S	0.8	31.25	25.00	OB	0	29.50	0	72.10
Treatment Mean				**36.67**	**44.00**			**31.42**	**25.13**			**31.92**	**0**	**69.13 b**
PM-Medium BCB	M-7	PM – S	1.2	34.75	41.70	S	0.8	65.25	52.20	OB	0	0.00	0	93.90
	M-22	PM – S	1.2	38.25	45.90	S	0.8	61.75	49.40	OB	0	0.00	0	95.30
	M-26	PM – S	1.2	27.00	32.40	S	0.8	72.50	58.00	OB	0	0.50	0	90.40
Treatment Mean				**33.33**	**40.00**			**66.50**	**53.20**			**0.17**	**0**	**93.20 ab**
PM-Deep BCB	M-23	PM – S	1.2	35.25	42.30	S	0.8	64.75	51.80	OB	0	0.00	0	94.10
	M-30	PM – S	1.2	29.75	35.70	S	0.8	70.25	56.20	OB	0	0.00	0	91.90
	M-34	PM – S	1.2	30.50	36.60	S	0.8	69.50	55.60	OB	0	0.00	0	92.20
Treatment Mean				**31.83**	**38.20**			**68.17**	**54.53**				**0**	**92.73 ab**
FFM-Deep BCB	M-10	FFM - S	0.8	14.75	11.80	S	0.8	85.28	68.20	OB	0	0.00	0	80.00
	M-18	FFM - S	0.8	16.25	13.00	S	0.8	83.75	67.00	OB	0	0.00	0	80.00
	M-24	FFM - S	0.8	17.00	13.60	S	0.8	83.00	66.40	OB	0	0.00	0	80.00
Treatment Mean				**16.00**	**12.80**			**84.00**	**67.20**				**0**	**80.00 ab**
Reference	Ref-1	[c]S	0.8	[e]20.00	16.00	SCL	1.5	60.00	90.00	S	0.8	20.00	16.00	122.00
	Ref-2	LS	1.1	20.00	22.00	SCL	1.5	60.00	90.00	S	0.8	20.00	16.00	128.00
	Ref-3	S	0.8	20.00	16.00	S	0.8	60.00	48.0	S	0.8	20.00	16.00	80.00
Treatment Mean				**20.00**	**18.00**			**60.00**	**76.00**				**16.00**	**110.00 a**

[a]The ASCS contains subplots within each treatment plot that have tree stands of different species compositions, including pure stands of aspen, pine, and spruce, as well as the mixed plots that contain a mixture of all three species (Barber et al. 2015). For this study, only the mixed plots were examined.

[b]Textural class of topsoil materials as classified by NorthWind Land Resources Inc. (2013).

[c]Soil textural classes: S = sand; LS = Loamy sand; SCL = Sandy clay loam

[d]Depth data from NorthWind Land Resources Inc. (2013).

[e]Average depth from multiple sampling sites.

[f]Overburden material is dense, impermeable heavy clay, and has no water holding capacity (NorthWind Land Resources 2013).

Table A.3 – Results of Principal Components Analysis of the correlation matrix of nutrients from soil, bioavailable, and foliar nutrient pools from mixed tree plots at Aurora Soil Capping Study. For each principal component, the bolded values represent the variable with the highest absolute eigenvector (V-vector) and every variable with an absolute value within 10 % of that value.

Principal Components	PC-1	PC-2	PC-3	PC-4	PC-5	PC-6
Eigenvalue	28.8	11.0	7.4	6.9	5.7	4.6
Variation (%)	33.2	12.6	8.4	7.9	6.5	5.2
Cumulative Variation (%)	33.2	45.8	54.2	62.2	68.7	73.9
Eigenvectors (V-Vectors)						
Environmental Physical or Chemical Properties						
pH – Topsoil	0.594	0.369	-0.127	-0.112	-0.452	-0.014
pH – Upper Subsoil	0.812	-0.234	-0.062	-0.446	-0.046	-0.164
Electrical Conductivity – Topsoil	0.817	0.208	0.221	0.181	0.098	0.223
Electrical Conductivity – U. Subsoil	0.779	0.229	0.451	0.184	-0.074	0.065
Available Water Holding Capacity	-0.489	0.509	-0.405	-0.103	-0.467	-0.201
Soil Total C – Topsoil	**0.866**	0.258	-0.100	-0.333	-0.048	-0.002
Soil Total C – Upper Subsoil	-0.419	0.466	0.353	0.233	0.158	-0.096
Foliar Total C – Aspen	0.368	0.021	-0.670	0.090	-0.223	0.438
Foliar Total C – Pine	0.091	-0.031	0.025	0.145	0.505	0.298
Foliar Nutrients						
B – Aspen	0.671	0.069	0.163	0.059	-0.010	-0.201
B – Pine	0.567	-0.506	-0.190	0.318	0.238	-0.116
Ca – Aspen	-0.201	-0.218	-0.161	0.168	0.397	0.032
Ca – Pine	**0.900**	0.171	-0.026	-0.243	-0.106	-0.198
Cu – Aspen	-0.618	0.450	0.033	0.171	-0.207	0.033
Cu – Pine	0.191	-0.233	-0.386	-0.247	-0.375	0.000
Fe – Aspen	0.286	**-0.822**	0.095	0.140	-0.043	-0.039
Fe – Pine	0.514	-0.543	0.241	0.154	-0.095	-0.328
K – Aspen	0.720	0.187	0.156	0.312	-0.364	0.155
K – Pine	-0.428	-0.094	0.053	0.282	0.087	**-0.693**
Mg – Aspen	-0.608	-0.575	0.020	0.186	0.236	0.011
Mg – Pine	0.316	-0.446	-0.618	0.258	-0.127	-0.249
Mn – Aspen	**-0.927**	-0.110	-0.075	0.107	-0.073	-0.006
Mn – Pine	**-0.913**	-0.227	-0.040	0.164	-0.033	-0.030
N – Aspen	0.731	-0.017	-0.436	-0.276	-0.155	0.212
N – Pine	0.235	-0.058	-0.255	-0.090	-0.369	0.201
Na – Aspen	0.380	-0.178	-0.254	0.215	0.336	-0.172
Na – Pine	0.384	-0.466	-0.078	-0.309	-0.414	0.112
P – Aspen	-0.793	-0.295	0.045	0.231	-0.318	0.080
P – Pine	-0.816	-0.416	-0.071	0.323	0.086	-0.037
S – Aspen	0.792	0.118	-0.126	0.323	-0.139	0.094
S – Pine	0.754	-0.413	-0.088	0.118	0.056	0.129
Zn – Aspen	0.697	-0.411	-0.187	0.063	-0.285	0.153
Zn – Pine	0.539	-0.378	-0.166	-0.149	-0.083	**-0.683**

Bioavailable Nutrients						
Al – Topsoil	0.231	-0.240	0.004	0.427	**-0.729**	0.160
Al – Upper Subsoil	-0.075	-0.006	0.428	0.451	-0.327	-0.121
B – Topsoil	0.356	-0.079	-0.536	0.562	-0.061	-0.002
B – Upper Subsoil	0.358	-0.222	0.123	0.174	0.113	0.519
Ca – Topsoil	**0.853**	0.181	0.005	0.266	0.050	-0.206
Ca – Upper Subsoil	0.708	0.245	0.408	0.214	0.008	-0.416
Cu – Topsoil	0.037	-0.329	-0.192	**-0.564**	0.173	0.368
Cu – Upper Subsoil	-0.365	0.386	-0.023	-0.049	-0.464	0.254
Fe – Topsoil	0.440	-0.027	-0.323	**0.658**	0.126	0.068
Fe – Upper Subsoil	0.081	-0.378	0.224	0.491	-0.255	0.153
K – Topsoil	-0.821	-0.334	0.043	0.310	-0.205	0.070
K – Upper Subsoil	0.388	0.168	0.349	-0.096	-0.136	0.253
Mg – Topsoil	**0.875**	-0.066	-0.010	0.330	-0.082	-0.087
Mg – Upper Subsoil	0.673	0.080	0.562	0.289	0.108	-0.171
Mn – Topsoil	0.463	-0.330	0.130	0.476	-0.354	0.036
Mn – Upper Subsoil	-0.325	-0.413	0.137	0.346	0.123	-0.123
N – Topsoil	0.700	0.043	-0.019	0.372	-0.359	-0.321
N – Upper Subsoil	0.317	0.071	0.625	0.076	-0.025	-0.389
Na – Topsoil	-0.588	-0.256	-0.182	-0.350	0.337	0.206
Na – Upper Subsoil	0.047	-0.185	-0.578	-0.062	-0.417	0.077
P – Topsoil	-0.579	-0.167	0.146	0.161	-0.445	0.267
P – Upper Subsoil	-0.560	0.302	0.040	0.120	-0.376	0.157
Pb – Topsoil	0.220	-0.310	-0.196	**0.652**	-0.279	0.218
Pb – Upper Subsoil	0.179	-0.539	0.145	0.276	0.316	-0.050
S – Topsoil	0.723	0.163	-0.113	0.379	0.250	-0.034
S – Upper Subsoil	0.704	0.223	0.442	0.313	0.124	0.157
Zn – Topsoil	0.453	0.008	-0.190	0.429	0.159	-0.166
Zn – Upper Subsoil	0.463	-0.394	-0.236	-0.021	0.150	0.481
Soil Nutrients						
B – Topsoil	**0.890**	0.356	-0.026	-0.188	-0.08	0.038
B – Upper Subsoil	0.214	0.217	**-0.813**	0.296	0.115	-0.018
Ca – Topsoil	0.705	0.244	0.092	-0.395	-0.216	-0.280
Ca – Upper Subsoil	0.033	0.697	0.021	0.119	-0.253	-0.325
Cu – Topsoil	-0.051	0.112	0.550	0.279	0.050	0.531
Cu – Upper Subsoil	-0.565	-0.129	-0.251	0.128	0.368	-0.109
Fe – Topsoil	0.313	0.615	-0.007	0.137	0.503	0.350
Fe – Upper Subsoil	-0.714	0.587	-0.213	0.188	-0.048	-0.024
K – Topsoil	0.446	0.713	-0.037	0.203	0.205	0.092
K – Upper Subsoil	-0.036	0.149	**-0.742**	0.260	0.103	-0.114
Mg – Topsoil	0.814	0.428	0.037	-0.016	0.129	0.132
Mg – Upper Subsoil	-0.618	0.481	-0.234	0.159	-0.247	0.003
Mn – Topsoil	-0.664	0.512	-0.070	0.151	-0.305	-0.130
Mn – Upper Subsoil	-0.731	0.525	-0.079	0.016	0.278	-0.044
N – Topsoil	0.807	0.279	-0.131	-0.361	-0.097	0.002
N – Upper Subsoil	-0.670	0.636	0.117	0.206	0.039	-0.024

Na – Topsoil	0.236	0.492	0.062	0.498	-0.050	0.436
Na – Upper Subsoil	0.698	-0.018	-0.327	0.046	0.231	-0.092
Ni – Topsoil	0.400	0.253	0.286	0.159	0.181	0.428
Ni – Upper Subsoil	-0.066	0.242	-0.420	0.297	0.552	-0.110
P – Topsoil	0.465	0.496	-0.206	-0.278	-0.092	-0.063
P – Upper Subsoil	-0.783	0.547	-0.167	0.077	-0.011	0.003
S – Topsoil	**0.927**	0.275	-0.043	-0.013	0.114	0.119
S – Upper Subsoil	0.626	-0.295	-0.228	0.140	0.083	0.072
Zn – Topsoil	0.134	0.531	-0.323	0.504	-0.082	-0.202
Zn – Upper Subsoil	-0.346	0.532	-0.604	0.127	0.265	-0.038