# ORGANIC AMENDMENT EFFECTS ON PRODUCTIVITY OF WELLSITES RECLAIMED WITH SUBOPTIMAL TOPSOIL REPLACEMENT DEPTH IN NORTHEASTERN ALBERTA

By

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A Thesis Submitted to the Faculty of Graduate Studies of
The University of Manitoba
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

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## ABSTRACT

Takudzwa Nawu, M.Sc., The University of Manitoba, May 2022. Organic amendment effects on productivity of wellsites reclaimed with suboptimal topsoil replacement depth. Advisor: Dr. Francis Zvomuya

Reclamation and revegetation of boreal sites disturbed by oil exploration depend on the availability of topsoil salvaged on-site during the disturbance. For successful reclamation, current regulations require the salvaging of enough soil to attain at least 80% of the original topsoil depth. However, salvaged topsoil at many sites is often insufficient to achieve the 80% topsoil replacement depth (TRD80) required for successful reclamation in western Canada. We tested organic amendment effects on reclamation success of wellsites reclaimed with insufficient salvaged topsoil to a level similar to the mandatory TRD80. Two main studies were conducted to examine (i) soil responses and (ii) vegetation responses to 50% topsoil replacement depth without organic amendment (TRD50) or amended with either peat (PTRD50) or biochar (BTRD50), relative to the TRD80 treatment, following wellsite reclamation at Cold Lake, Alberta.

Soil properties results showed that both amendments improved topsoil properties. Peat-amended plots showed a 113% increase in total Kjeldahl nitrogen concentration relative to the mean of other treatments, while biochar produced significantly greater potassium concentrations in TRD50 relative to peat-amended plots.

Vegetation responses to insufficient topsoil depth and organic amendment application showed an increase in native species, graminoid and woody species richness while non-native and forb species richness decreased across all treatments 5 yr after reclamation. Tree growth was greater in the peat amendment and the TRD80 treatment than in the TRD50 and the biochar treatments. Peat and biochar improved soil properties of disturbed boreal sites reclaimed with insufficient salvaged

topsoil to a level suitable for successful restoration. However, in the revegetation study peat improved vegetation establishment and plant community development while biochar showed no benefits on vegetation variables. Overall, the peat treatment (PTRD50) produced similar vegetation performance results to the mandatory TRD80 treatment, indicating that peat amendment can improve reclamation success at disturbed boreal sites where salvaged soil is insufficient to achieve the optimal 80% TRD.

ACKNOWLEDGEMENTS

This research project was made a success by contributions from several people along the way. My heartfelt appreciation goes to my Advisor Dr. Francis Zvomuya for his guidance, encouragement, and kind supervision throughout the research thesis's final writing. I would like to express my gratitude to Dr. Sylvie Renault and Dr. Inoka Amarakoon for agreeing to serve on my committee and for their invaluable support, guidance, and input towards my thesis.

My acknowledgment goes to Dr. Michelle Young and Dr. Asfaw Bekele for their vast knowledge on this research work and, for devoting many hours to our monthly update meetings, assisting, and suggesting great ideas in shaping the outcome of this work. Thanks to Paragon Soil and Environmental Consulting Inc. for their generous assistance in data collection and field work during the reclamation research program. I owe my special thanks to Imperial Oil Resource Ltd and Mitacs for funding this research project, as well as to the Faculty of Graduate Studies for the University of Manitoba Graduate Fellowship (UMGF) award, for providing stipend.

A special thanks to my parents and my brothers Takunda and Tavonga Nawu for believing in me, encouraging, and praying for me during my studies. A special shout out to my friends Taurai Matengu, Clemence Muitire, Nikisha Muhandiram, Dr. Theresa Adesanya, and all my colleagues for their motivation, assistance, and their companionship up to the completion of my research.

I want to extend my deepest gratitude to Almighty God for the protection and strength necessary to complete my studies during this pandemic era.

## FORWARD

This thesis was written in manuscript format following the thesis guidelines of Department of Soil Science, University of Manitoba. This thesis is divided into 4 chapters, the first chapter (chapter 1) of which is a general introduction and review of literature while chapters 2 and 3 were prepared as separate research manuscripts. Chapter 2 focused on determining the effects of organic amendments on the soil properties in oil wellsites reclaimed with suboptimal topsoil replacement depth. Chapter 3 examined vegetation responses to peat and biochar in boreal sites reclaimed with insufficient topsoil replacement depth. Chapter 4 is the overall synthesis which summarizes the findings and recommendations reported in Chapters 2 and 3. Chapters 2 and 3 are being prepared for submission to the journal of Land Degradation and Development.

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# 1. GENERAL INTRODUCTION

## 1.1 Background information

Oil and natural gas are essential sources of energy for Canada. However, the exploration and production of these nonrenewable resources can adversely impact the environment. Construction and site preparation for natural gas and oil activities require vegetation and soil removal, which result in the disruption of soil biogeochemical properties and therefore, low productivity and damage to natural ecosystems, especially in boreal regions. Reclamation, which is the process of improving disturbed land to a level similar to the pre-disturbed condition or for a specified use, is therefore, necessary to restore land productivity to a level equivalent to that before disturbance.

The quantity and quality of topsoil returned to the site during reclamation determines the success of wellsite restoration. However, at some wellsites where topsoil was not salvaged, it may not be possible to attain the required 80% topsoil replacement depth to attain the goal of equivalent capability. Therefore, in cases where there is insufficient topsoil for reclamation, organic amendments may improve reclamation and enhance productivity. Currently, there is limited information on the use of organic amendments, such as biochar and peat, to supplement insufficient salvaged topsoil in the reclamation of wellsites in forest ecosystems. Most of the published research on suboptimal topsoil replacement depth (TRD) and organic amendment use in reclamation has focused on cropland, with little focus on forest environments. There is a need to investigate the impact of insufficient topsoil replacement with or without organic amendments on the productivity of reclaimed wellsites. Therefore, this research examined the effects of topsoil replacement depth and organic amendment type on topsoil physiochemical properties and vegetation attributes following reclamation. Results from the study will be used to provide guidelines for the reclamation of borrow infill sites and wellsites where salvaged soil is insufficient to achieve the required TRD.

## 1.2 Oil and natural gas wellsites in Alberta

Oil and natural gas activities can reduce land productivity through alteration of soil biogeochemical properties, vegetation destruction, and environmental degradation. Global demand for oil and natural gas has led to increased extraction of these natural resources, leading to frequent land disturbances and an increased number of abandoned wellsites (Lupardus et al., 2019). Currently, there are about 157,000 active wells and 96,000 inactive wells and 76,000 decommissioned (unreclaimed) wells in Alberta, representing approximately 17% of all wellsites in the province (Government of Alberta, 2021). To safeguard the land resources, the Government of Alberta requires the reclamation of disturbed land up to a standard of equivalent land capability (Alberta Environment, 1995; Powter et al., 2012). Equivalent land capability is defined as the ability of the reclaimed wellsite to support various uses to the level that existed before soil disturbance (Alberta Environment, 1995; ESRD, 2013).

During site preparation for oil and natural gas exploration, companies are required to salvage and stockpile topsoil for later use in reclamation following the end of oil and natural gas production (Larney et al., 2005; Powter et al., 2012; ESRD, 2013). Successful reclamation is required to restore the land to its ecological integrity. The success depends on the quality and quantity of salvaged soil to be used in the reclamation process, most importantly the topsoil replacement depth (Larney et al., 2005; ESRD, 2013). The province of Alberta mandates that reclaimed soils must have a minimum of 80% topsoil replacement depth, that is, no more than 20% variance from undisturbed soils (ESRD, 2013).

## 1.3 Borrow pits in Alberta

In Alberta, an increase in human population, infrastructure development (e.g., road network expansion), and natural resource extraction have led to an increased demand for topsoil fill. This excavated topsoil is usually sourced offsite for purposes such as road construction and reclamation of wellsites, leaving behind desurfaced sites (borrow pits) (Alberta Environment and Parks, 2018). As of 2018, there were approximately 900 Class I (5 ha or more in size) and more than 1,500 class II (less than 5 ha in size) borrow pits in Alberta (Alberta Environment and Parks, 2018). Abandoned borrow sites have little or no topsoil, organic matter, or native seed propagules required for the reclamation and ecological restoration (the process of initiating or accelerating the recovery of an ecosystem that has been degraded, damaged, or destroyed (Harper and Kershaw, 1996; Harper and Kershaw, 1997). The remaining subsoils are characterized by low nutrient availability, high bulk densities, poor soil structure, and low moisture retention, limiting the restoration process in these borrow pits (Harper and Kershaw, 1996; Harper and Kershaw, 1997). Previous studies have demonstrated that natural succession on borrow pits requires several decades to recover the ecosystem (Harper and Kershaw, 1996; Řehounková and Prach, 2008). Hugron et al. (2011) reported failure of borrow pit recovery following five decades of natural succession in the Northern Territories, Canada. These studies emphasize the need for reclamation of borrow pits to initiate plant establishment and restoration.

## 1.4 Disturbances in the forest ecosystem

Forest ecosystems are often exposed to natural disturbances such as fires, altering the integrity of the ecosystem. However, a major cause of forest disturbances is anthropogenic activities, including exploration and extraction of oil and natural gas (Macdonald et al., 2015; Angel et al., 2017). A large portion of oil and natural gas resources are in the heart of the Canadian boreal forest (Schneider and Dyer, 2006). Approximately 168 billion barrels of oil deposits (oil sands and conventional) lie under the Canadian boreal forest region in northern Alberta (Government of Alberta, 2014). Alberta also has vast forests characterized by a diverse flora and fauna ecosystem, providing timber to local companies and tourism services (Timoney, 2003; Pickell et al., 2015). Before 1900, forests wholly covered Alberta's land, but currently, it covers approximately 50% of the province's land area due to increased industrialization and mining activities (Woynillowicz et al., 2005). Unfortunately, site construction for oil and natural gas exploration involves vegetation clearing an area of approximately 100 m × 100 m and soil disturbance, which consequently alter the productivity of the forest ecosystem (Lupardus et al., 2019).

Drilling rig setup requires a firm and level subsurface, which in turn requires topsoil stripping, excavation, and compaction of subsoil for successful oil and natural gas exploration (Lupardus et al., 2019). Further, seed bank propagules for native species are removed together with the topsoil during soil stripping, leaving the site prone to low productivity (Lupardus et al., 2019; Mackenzie and Naeth, 2009; Macdonald et al., 2015). Reclamation of these disturbed sites is essential to return the land to productivity levels similar to those before the disturbance.

### 1.5 Reclamation process and challenges

The Environmental Protection and Enhancement Act (EPEA) in Alberta requires oil and natural gas companies to reclaim disturbed lands, soon after production ceases, to a sustainable level that existed before the disturbance (ESRD, 2013). The reclamation process is measured by the thickness of the soil returned, soil organic matter content, drainage, and vegetation recovery on the disturbed site. The reclamation process entails topsoil salvage and stockpiling, topsoil replacement and, finally, revegetation.

### 1.5.1 Topsoil salvaging

Topsoil salvaging is a mandatory critical pre-disturbance step that involves conserving topsoil and subsoil for later use in reclamation and restoration of disturbed sites after production ceases (ESRD, 2013). The process involves stripping topsoil and subsoil separately using excavation equipment (ESRD, 2013; Natural Resources Canada, 2017). The salvaged topsoil can be stored in stockpiles for future use in reclamation after the decommissioning of oil and natural gas facilities (Natural Resources Canada, 2017; Akala and Lal, 2000). Current Alberta regulations mandates that 80% topsoil replacement depth of the original depth be salvaged and replaced during reclamation to meet the goal level of equivalent capability (productivity of the land similar prior the disturbance) (Powter et al., 2012; ESRD, 2013).

However, most of Alberta's older well sites and borrow pits have insufficient or no salvaged topsoil to meet the required 80% topsoil replacement depth for successful reclamation (Powter et al., 2012; ESRD, 2013). Several factors have exacerbated this problem, including topsoil loss during salvaging and placement in early wellsite establishment, and compaction of organic horizons during soil handling (Mitchem et al., 2009; Mason et al., 2011; Day et al., 2015).

Additionally, prior to 1994, there were no regulations requiring salvaging of topsoil for use in reclamation after oil and natural gas exploration (Olsen and Jones, 1989; Powter et al., 2012). Excavated topsoil was exported to other sites to serve various purposes (borrow activities), resulting in suboptimal or no topsoil left on-site (borrow pit) for reclamation of the sites (Larney et al., 2005). While topsoil can be imported from other sites to reclaim desurfaced sites, it can be expensive. Additionally, the donor site so disturbed will need reclamation (Larney et al., 2003).

### 1.5.2 Soil Stockpiling

Soil stockpiling is a process of storing salvaged topsoil and subsoil in piles for future use in the reclamation of disturbed land (Strohmayer, 1999; Natural Resources Canada, 2017). The topsoil can be stockpiled for durations ranging from a few months to several years prior to use in reclamation and restoration (Strohmayer, 1999; Ghose, 2001; Natural Resources Canada, 2017).

Practices such as admixing of topsoil and subsoil during stockpiling and the use of oversized or undersized stockpiles often alter soil physical (higher bulk density, reduced aggregate stability), chemical (low nutrient availability) and biological properties (low microbial population and activity) (Abdul-Kareem and McRae, 1984; Dickie et al., 1988; Akala and Lal, 2000). Previous studies have shown significant reductions in soil organic carbon and total nitrogen concentrations (Visser et al., 1984; Kundu and Ghose, 1997). Long-term stockpiling for more than 8 mo is detrimental to soil and native plant seed quality (Natural Resources Canada, 2017). Stockpiles are more prone to anaerobic conditions, especially at more profound depths in large stockpiles, reducing essential microbial activity and soil nutrient cycling and availability in most disturbed sites reclaimed using such stockpiled soil (Stark and Redente, 1987; Dickie et al., 1988; Golos et al., 2016; Dhar et al., 2019).

### 1.5.3 Topsoil replacement

When oil and natural gas wells are no longer productive, the wellsites are decommissioned and salvaged topsoil is replaced on the site for reclamation and revegetation (ESRD, 2013, Alberta Environment and Water, 2012). Optimal topsoil replacement is required to reclaim the disturbed land to a level that existed prior to the disturbance (Larney et al., 2005, 2012; Powter et al., 2012; ESRD, 2013). While a portion of the topsoil can be recovered from salvaging, the replacement soil usually has altered properties (compaction and deficiencies of required nutrients) and may not perform to a level comparable to that of undisturbed adjacent land (Samborsky, 2016). Previous research has demonstrated considerably decreased soil nutrient availability on disturbed sites reclaime with less topsoil depth compared to sites reclaimed with more topsoil depth (Bowen et al., 2005; Larney et al., 2005, 2012).

Addition of organic amendments to the topsoil is becoming a significant component in augmenting reclamation where topsoil is scarce (Larney et al., 2005; Larney and Angers, 2012). Organic amendments incorporated during soil replacement improve soil physical and biological properties and nutrient availability, which are essential for successful early revegetation.

### 1.6 Organic amendments in reclamation

Topsoil and organic matter are critical for vegetation establishment as they enhance soil physical, chemical, and biological properties (Macdonald et al., 2015; Dietrich and MacKenzie, 2018). Desurfaced soils often lack topsoil and organic matter, which are required for sustainable soil and vegetation productivity (Akala and Lal, 2000; Larney et al., 2012). Compacted subsoils with low organic matter content, high bulk densities, low permeability, and altered soil chemical properties often result in low vegetation productivity (Sheoran et al., 2010; Macdonald et al., 2015 ; Dietrich and MacKenzie, 2018). Other soil properties, such as aggregation, mineralization, and soil organic carbon (SOC) accumulation are indirectly affected by suppressed soil microbial activities in low organic matter soils (Larney et al., 2012; Bekele et al., 2015; Srivastava et al., 2016). Natural replenishment of organic matter in these degraded soils is hampered by unfavorable soil physical and hydrological properties and by low nutrient availability (Larney et al., 2012; Larney and Angers, 2012).

Reclamation success of surface-stripped soils rests on reconstructing surface horizons with enough soil organic matter to sustain productivity (Akala and Lal, 2000; Larney et al., 2003; Larney et al., 2005). The effectiveness of organic amendments in improving soil productivity during the restoration of oil wellsites is well-known (Larney and Janzen, 1996; Larney et al., 2000; Larney et al., 2003; 2005; Zvomuya et al., 2007; 2008). Organic amendment application improves soil organic matter content, which plays an essential role in augmenting vital soil physiochemical properties, including total nutrient supply and reserves (Zvomuya et al., 2007; Hemstock et al., 2010), soil structure, and water holding capacity (Bendfeldt et al., 2001; Larney and Angers, 2012; Page-Dumroese et al., 2018).

Organic amendment use in reclamation has attracted growing attention due to the resulting long-term sustainable productivity (Enders et al., 2012). Organic amendments are a cost-effective strategy in the restoration of wellsites with suboptimal topsoil. At many locations, organic amendments such as fresh or composted animal manure from local livestock operations, biosolids, coarse woody debris, wood shavings, and peat are readily available on-site, hence at minimal transportation costs (Larney and Pan 2006; Rowland et al., 2009; Page-Dumroese et al., 2018).

### 1.6.1 Biochar

Biochar is produced from the slow pyrolysis of biomass under an oxygen-limited environment. It can be made from various sources, including animal manure, municipal wastes, papermill sludges, agricultural wastes, and wood residues from local timber harvest (Joseph et al., 2010; Lehmann and Joseph, 2015). Additionally, biochar can be produced from waste wood (unmerchantable round wood) from timber harvesting, making it a relatively low-cost method of producing a high-carbon amendment for use in forest reclamation. (Page-Dumroese et al., 2018; Rodriguez-Franco and Page-Dumroese, 2021).

Biochar is predominantly used as a soil organic amendment in rangeland, agricultural, and forest environments (Atkinson et al., 2010; Beesley et al., 2010). Biochar was first mentioned in the 1820s, as a soil amendment in the revegetation and restoration of forest ecosystems in Scotland (Thomas and Gale, 2015). The use of biochar as a soil amendment has been shown to enhance the chemical, physical, and biological properties of the soil, resulting in improved soil fertility to support vegetation establishment and survival (Lehmann et al., 2003; Tammeorg et al., 2014; Rawat et al., 2019). Biochar is porous and can increase soil porosity; it is a low-density material that reduces bulk density of most compacted soils and can provide a habitat for microbial life in the soil (Rawat et al., 2019). It has various carboxylic and phenolic functional groups, providing a large surface area for cation exchange capacity (CEC), water retention, and increased retention of the majority of cations (Glaser et al., 2002; Tammeorg et al., 2014; Page-Dumroese et al., 2016). Biochar’s recalcitrant properties give the amendment the ability to resists microbial decomposition, supplying C and other nutrients slowly, hence sustaining soil productivity in the long term (Baldock and Smernik, 2002; Enders et al., 2012). It has been reported to contain high concentration levels of nutrients such as K that are readily available to plants (Dumroese et al., 2018). However, the bioavailability of nutrients such as nitrogen in biochar varies with the source of biochar, and with the temperature and duration of the pyrolysis process (Joseph et al., 2010; Lehmann and Joseph, 2015).

Majority of previous studies have focused on the role of biochar in augmenting soil quality in agricultural soils (Jeffery et al., 2011; Liu et al., 2013). However, recently biochar has gained popularity and use as an amendment in forest reclamation studies (Petelina et al., 2014; Thomas and Gale, 2015; Page- Dumroese et al., 2018). In a study by Schultz et al. (2017), biochar increased total organic carbon (TOC) from 62.2 to 95.5 µg $g^{-1}$ in reclaimed sodic soils in North Dakota, USA. Dietrich and MacKenzie (2018) reported a significant increase in soil K and seedling growth of trembling aspen (*Populus tremuloides*) on reclaimed peat mineral mix cover soil with biochar in the Athabasca oil sands region of Alberta, Canada. Ramlow et al. (2018) reported an improvement in N availability and a 26% increase in soil moisture retention in reclaimed soils amended with woody biochar.

In addition to enhancing soil properties in reclaimed sites, biochar has proved to be a useful tool in carbon sequestration and in reducing greenhouse gas emissions in forest ecosystems (Page-Dumroese et al., 2016; Majumder et al., 2019).

### 1.6.2 Peat

Peat is a naturally occurring surface organic layer formed by the partial biochemical decomposition of accumulated dead plant material under waterlogging, oxygen-limiting, nutrient deficient, and acidic conditions (Cao, 2019). Peat is usually black or brown, with characteristics making it an excellent soil organic amendment for improving the properties of compacted subsoils (Cao, 2019; Li et al., 2004). Peat is abundantly available in Alberta's oil sands region, which is home to approximately 64% of peat wetlands in the province's boreal forest (Rooney et al., 2012). It has found use in reclamation since it is readily accessible within the boreal forest, which reduces transportation costs (Rowland et al., 2009). Peat has a large pore space volume and a high specific surface area, which impart a high CEC and improved water storage and nutrient sorption capacity (Li et al., 2004; Rezanezhad et al., 2016; Cao, 2019).

Furthermore, peat's high permeability improves infiltration, saturated hydraulic conductivity, and plant root penetration in compacted desurfaced soils (Brown and Naeth, 2014; Ojekanmi and Chang, 2014). Moreover, peat contains a large proportion of humic acids, which may stimulate microbial activity and promote aggregation (Cao, 2019). Peat can supply nutrients such as N, P, and K, which are essential for sustainable reclamation success (Bragazza et al., 2006; 2013). However, some studies have shown low available forms of N in peat due to reduced nitrification and mineralization rates compared to forest floor mix (Li et al., 2004). Pinno et al. (2012) reported a significant increase in tree (trembling aspen) height, biomass, and foliar N in oil sands reclaimed soils amended with peat in a greenhouse experiment. Despite its general use as an organic amendment, peat is a potential option in reconstructing topsoil at wellsites with no or insufficient salvaged topsoil for reclamation.

## 1.7 Role of organic amendments in soil reclamation

### 1.7.1 Soil chemical properties

Soil disturbance alters soil physical and chemical properties. Organic amendments influence soil organic carbon (SOC), soil pH, electrical conductivity, cation exchange capacity, base saturation, and nutrient availability (Glaser et al., 2002; Larney and Angers, 2012; Mukherjee and Zimmerman, 2013). These soil chemical properties are essential for soil productivity and vegetation growth in reclaimed sites (Larney and Angers, 2012; Petelina et al., 2014; Quideau et al., 2017; Page-Dumroese et al., 2018). Larney et al. (2005; 2012) examined SOC changes in soils amended with wheat straw, manure, compost, and alfalfa hay at three wellsites in central and southern Alberta. They found that compost and manure enhanced SOC to a greater extent than wheat straw and alfalfa hay across all wellsites. However, wheat straw and alfalfa hay had no significant effect on SOC levels in one of the study years relative to unamended plots (control). Bekele et al. (2015) reported a slight increase in SOC in clay sand subsoils amended with biochar relative to subsoils amended with humalite.

Nitrogen is one of the essential nutrients in short supply in most disturbed forest wellsites (Lanning and Williams, 1981). Zvomuya et al. (2007) studied the effects of differential rates of compost, manure, wheat straw, and alfalfa hay on the reclamation of abandoned natural gas wellsites. While they found no significant effect of compost rate on nitrate-N content in the 0–60 cm soil layer, nitrate-N concentration increased by 7.78 kg $ha^{-1}$ for each Mg $ha^{-1}$ increase in alfalfa hay application (Zvomuya et al., 2007). A study by Page-Dumroese et al. (2018) showed that biosolids in the 0-3 cm soil layer gave significantly higher inorganic N content than biochar and woodchips after 19 mo of a mine site reclamation in northeastern Oregon national forests, USA. Another essential nutrient that can be influenced by organic amendments in wellsite recovery is phosphorus. Zvomuya et al. (2007) reported an increase of 3.24 kg $ha^{-1}$ in available P for every Mg $ha^{-1}$ increase in compost rate. Larney et al. (2011) examined the residual effects of organic amendments on the productivity of desurfaced soils and found that total P concentration remained high in soils amended with fresh, old, and composted cattle manure, poultry, or hog manure after 11.5 yr.

Soil CEC affects nutrient uptake by plants and ion movement in the soil (Gao and Chang, 1996). The high specific surface area and various functional groups of most organic amendments enhance cation exchange reactions, thereby increasing soil fertility. Fellet et al. (2011), reported an increase in CEC in mine tailings amended with 5% and 10% rates of biochar compared to the control and 1% biochar rate. In a study on the reclamation of copper mine tailings, biosolids increased CEC in the 0-15 cm layer (Gardner et al., 2010).

### 1.7.2 Soil physical properties

Organic amendments positively influence various soil physical properties, including aggregate stability, soil structure, drainage, water retention, and bulk density (Larney et al., 2003; Bekele et al., 2015; Rezanezhad et al., 2016). Compacted soils are often high in bulk density, and therefore low in porosity, infiltration, and root penetrability while they are prone to high runoff (Lupardus et al., 2019). Organic amendments can reduce soil bulk density and increase total porosity, thus enhancing infiltration (Larney and Angers, 2012; Bekele et al., 2015; Glaser et al., 2002). Mukherjee et al. (2014) found that biochar reduced bulk density by 13% and increased sub-nanopore surface area by 15% compared to the control (no amendment) in a greenhouse study using artificially degraded soil.

Aggregate stability is a valuable physical property and measures the resistance of soil aggregates to breaking down from disruptive forces such as water or tillage (Tisdall and Oades, 1982; Tisdall, 1994; Aksakal et al., 2020). The interaction between biological and physicochemical properties of the soil affects aggregate stability (Tisdall and Oades, 1982). Quality organic amendments can improve aggregate stability in disturbed soils (Sun et al., 1995). Organic amendments indirectly promote soil aggregation by stimulating microbial activity in the soil. Microbial decomposition of organic amendments secretes binding agents such as fungal mycelia growth, polysaccharides and fungal hyphae, which bind soil particles into stable aggregates (Mikha and Rice, 2004). Bekele et al. (2015) reported an increase in dry aggregate sizes up to > 8 mm in clay soils amended with biochar, oxidized lignite, labile organic mix, and 2-way combinations of these organic amendments after two years. However, the proportion of > 8mm aggregates decreased at the end of the fourth year, while aggregate sizes of 4 to 8 mm, 4 to 2 mm, and 1 to 2 mm increased. In wellsite reclamation, Larney et al. (2005) examined the effects of organic amendments on soil responses and reported an increase in aggregate stability in soils amended with plant-derived organic amendments compared to manure- and compost-amended soils at two of the three sites in the first year of their study. The average aggregate stability values from straw and alfalfa applications were 59.3 and 57.4%, respectively, with alfalfa substantially higher than compost (52.6%) and the control (54.1%). After 40 mo, they sampled the same plots and reported significantly greater aggregate stability in straw, alfalfa and manure amended plots (28.7 - 54.6%) relative to compost and control plots (22.8 - 47.2%) at all the three sites.

Water retention (or water-holding capacity) is an essential measure of plant-available moisture, which is essential for revegetation during wellsite restoration (Larney and Angers, 2012; Macdonald et al., 2015; Rezanezhad et al., 2016). Naturally, the majority of organic amendments have high water holding capacity compared to most mineral soils (Camberato et al., 2006). Disturbed soils amended with organic amendments have improved water retention capacity and plant available water capacity (Fierro et al., 1999). Organic amendments may influence disturbed sand to a greater extent than fine-textured soils since the latter have high water retention properties (Zibilske et al., 2000; Rawls et al., 2003; Gardner et al., 2010). Page-Dumroese et al. (2018) found a considerably higher plant-available moisture content within the 0-3 cm soil layer in soils amended with biosolids (16.9 %) than those amended with biochar + woodchips (13.7 %) and the control (14.0%) throughout the year. Mukherjee et al. (2014) conducted a field study on the impacts of soils amended with biochar, humic acid, and water treatment residual, and reported a 63% increase in plant available water capacity in biochar-amended soil after one year.

Increased porosity in soils receiving amendments such as peat and biochar (Rezanezhad et al., 2016) can encourage microbial population growth due to improved aeration in the soil (Dietrich et al., 2017).

### 1.7.3 Revegetation

Revegetation involves the replanting of vegetation in disturbed wellsites during reclamation. It is a measure of soil productivity and successful wellsite recovery. Revegetation is undertaken to reduce soil erosion by stabilizing soil particles with plant roots, restoring the ecological integrity of forest ecosystems (Sheoran et al., 2010). Anthropogenic activities involve vegetation clearance, and topsoil loss results in compacted desurfaced soils, often limiting vegetation recovery (Elseroad et al., 2003; Macdonald et al., 2015). For successful vegetation recovery, climatic, topographic, vegetation, and soil characteristics must be considered (Smreciu, 2003). Topsoil salvage is one of the crucial processes that can help support successful revegetation during restoration. Topsoil contains seed propagules, organic matter, plant root residues, and high microbial activity, which are essential for the revegetation process (Smreciu, 2003). Disturbed wellsites with limited salvaged topsoil require extensive preparation and more plant species seed propagules for revegetation relative to those with intact topsoil and subsoil (Smreciu, 2003). However, the use of organic amendments can enhance revegetation during reclamation.

The addition of organic amendments enhances the interaction of soil physical, chemical, and biological processes to support vegetation recovery in disturbed sites (Sheoran et al., 2010; (Ramlow et al., 2018; Macdonald et al., 2015). Organic amendments contain nutrients that plants can utilize for their establishment and nourishment (Rowland et al., 2009; Brown and Naeth, 2014; Page-Dumroese et al., 2018). Additionally, they reduce bulk density by increasing total porosity, improving root growth and penetration (Sheoran et al., 2011; Ramlow et al., 2018). Further, increased water retention promotes vegetation establishment and cover during reclamation (Rawls et al., 2003; Benigno et al., 2013). Page-Dumroese et al. (2018) reported improved total percent vegetation cover when biochar and biosolids were applied singly, with a smaller bare ground of

23 and 17%, respectively. Macdonald et al. (2015) concluded that forest floor material could promote vegetation establishment in boreal forests during reclamation. Ramlow et al. (2018) reported an increase in the total percent vegetation cover and root biomass in wood strand mulch amended soils.

## 1.8 Use of organic amendments in reclamation

A 5-year study on abandoned natural gas wellsites in Alberta, Canada, examined the impact on reclamation performance and spring wheat crop biomass yields of factorial combinations of organic amendment type (compost, wheat straw, manure, and alfalfa) and topsoil replacement depth (0, 50, 100 and 150% of recommended TRD) (Larney et al., 2003; 2012; Zvomuya et al., 2007). Larney et al. (2003) reported the highest crop biomass yield and the best reclamation outcome from the combination of 100% TRD and compost. In contrast, straw application in the absence of topsoil replacement (0% TRD) produced the lowest yields and worst reclamation result. Zvomuya et al. (2008) reported an increase in grain N concentration and uptake in spring wheat with increasing TRD in conjunction with the low C/N ratio amendments, compost, and alfalfa, compared to the high C/N ratio wheat straw. Additionally, improvements in spring wheat yield were observed in plots amended with alfalfa and compost at rates of up to 6 and 10 Mg $ha^{-1}$, respectively, on reclaimed wellsites.

Larney et al. (2012) examined residual effects of topsoil replacement depths and organic amendments on abandoned oil and natural gas wellsites after 10 yr in southeastern Alberta. The authors re-sampled plots studied by Larney et al. (2003) and observed that increasing TRD from 50 to 100% or 150% did not significantly improve SOC or total N and available P concentration in the 0 - 15 cm layer of the reclaimed soil. However, all three TRD treatments had an average of

18% higher SOC than the 0% TRD control treatment. Furthermore, compost and manure treatments increased SOC by approximately 8% relative to straw, alfalfa hay, and control plots after ten years. Zvomuya et al. (2007), recommended applying alfalfa at 12 Mg $ha^{-1}$ or compost at > 20 Mg $ha^{-1}$ to enhance soil C storage and nutrient availability while reducing the risk of nutrient loss from surface runoff.

Bekele et al. (2015), conducted a greenhouse study to evaluate the effects of amending subsoil (reconstructed topsoil) with biochar or humalite only or a combination of these amendments with a labile organic mix (LOM) (a mixture of sawdust, alfalfa hay, and wheat straw) on soil physicochemical properties and microbial biomass. They concluded that application of either humalite or biochar mixed with LOM could be beneficially used in the reconstruction and restoration of disturbed agricultural lands and sustaining long-term soil productivity.

Page-Dumroese et al. (2018), studied the effects of using biochar, woodchips, biosolids, and two or three-way combinations of these organic amendments on soil properties and revegetation in mine site restoration in northeastern Oregon, USA natural forest for two years. They found that mixing the three organic amendments (biochar + biosolids + woodchips) improved nutrient availability and soil moisture content. They also reported a doubling in percent plant cover in organic amended treatments relative to the control treatments after two growing seasons.

While previous studies have focused on wellsite reclamation on cropland, there are currently no published studies on the effects of organic amendments on forest ecosystem restoration. Therefore, this study examined the effects of biochar and peat application on reclamation success when using 50% TRD in a forest ecosystem.

## 1.9 Research objectives

To our knowledge there is limited information on the use of organic amendments (biochar and peat) in the reclamation of wellsites with suboptimal salvaged topsoil in the forest environment. There is a need to investigate the impact of organic amendments on the productivity of wellsites reclaimed using insufficient topsoil. This thesis research examined the effects of organic amendments (peat and biochar) on the progression and success of reclamation and revegetation on borrow infill sites and wellsites with suboptimal topsoil depth. The specific objectives of the study were to (i) evaluate changes in topsoil chemical properties as functions of topsoil replacement depth and amendment type (Chapter 2); (ii) explore the relationship between vegetation attributes (that is, establishment, composition, growth/survival, and performance) vs. topsoil replacement depth and amendment type (Chapter 3). Results from this study will provide guidelines for the reclamation of borrow infill sites and wellsites where salvaged soil is insufficient to achieve the required topsoil replacement depth.

## 1.10 Thesis Outline

This thesis layout is written in accordance with the thesis guidelines of the Department of Soil Science, University of Manitoba. The thesis has four chapters including (Chapter 1) overall synthesis (Chapter 4) and two individual research chapters (Chapters 2 and 3) which were written in manuscript format as follows:

Chapter 2: Soil properties following oil wellsite reclamation with insufficient topsoil amended with peat and biochar

Chapter 3: Revegetation of wellsites reclaimed with suboptimal topsoil replacement depth and organic amendments

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# 2. SOIL PROPERTIES FOLLOWING OIL WELLSITE RECLAMATION WITH INSUFFICIENT TOPSOIL AMENDED WITH PEAT AND BIOCHAR

## 2.1 Abstract

An inevitable consequence of oil and gas exploration and development are disturbed sites, such as wellsites and borrow sites, which will require reclamation to restore and sustain levels of productivity equivalent to those that existed before the disturbance. However, salvaged topsoil at many sites is often insufficient to meet the 80% topsoil replacement depth (TRD80) required for successful reclamation in western Canada. This study examined the effectiveness of peat and biochar to augment reclamation success of abandoned oil wellsites using suboptimal (i.e., 50%) topsoil replacement depth (TRD) relative to 80% TRD (TRD80) without amendments. Amendments were applied once at rates calculated to raise the total organic carbon (TOC) mass ($m^{-2}$) in the TRD50 equivalent to that in the TRD80 treatment. Soil properties were measured annually for 5 yr. Results showed a 143%, 87% and 116% increase in total Kjeldahl nitrogen concentration in the peat-amended (PTRD50) plots relative to TRD80, TRD50, and biochar-amended (BTRD50) treatments, respectively, while biochar produced significantly greater soil potassium (K) concentration than peat. In addition, peat, and biochar significantly increased TOC concentrations by 83% and 88%, respectively, relative to the mean of the TRD80 and TRD50 treatments. Our results show that peat and biochar can improve soil properties of disturbed boreal sites reclaimed with insufficient salvaged topsoil to a level suitable for successful restoration.

## 2.2 Introduction

A large portion of Canada's oil and natural gas production is concentrated in Alberta's boreal forest region, which is home to the fourth-largest oil reserves in the world (EIA, 2021). Oil and natural gas exploration and development and borrow activities in this region often result in land disturbances that necessitate reclamation. Site preparation activities such as vegetation removal, stripping of soil for salvage, and stockpiling alter soil physical and chemical properties (Rowland et al., 2009 Bekele et al., 2015; Lupardus et al., 2019). Reclamation of these wellsites depends primarily on the amount and quality of topsoil returned to the site following decommissioning of the wellsite (Larney et al., 2003, 2005; Powter et al., 2012).

Topsoil replacement is a critical reclamation step that involves spreading salvaged or borrowed topsoil on disturbed land to provide a growing medium for plants during the restoration process (Strohmayer, 1999; Rowland et al., 2009). Recommended practices for topsoil salvaging and stockpiling provide optimal topsoil with good physical and chemical properties necessary to ensure reclamation success (Merino-Martín et al., 2017). Current Alberta regulations require an optimal topsoil replacement depth (TRD) of at least 80% of the original topsoil depth or a variance of no less than 20% from the adjacent undisturbed land (ESRD, 2013). In many cases, lack of available topsoil often negatively impacts soil chemical, physical and biological properties and early vegetation establishment. Therefore, borrow activities, which involve the importation of topsoil from donor sites for reclamation, are necessary at wellsites where optimal topsoil is not readily available  (Larney et al., 2005; MacKenzie et al., 2012). However, borrow activities create land disturbances which in turn require reclamation.

Suboptimal TRD has been shown to result in low nutrient availability and productivity, slowing the reclamation process. Bowen et al. (2005) reported significantly lower total nitrogen (N) and soil organic carbon (SOC) concentrations in control plots (no topsoil replacement) than plots reclaimed with 40 cm and 60 cm topsoil depth in Wyoming, USA. Larney et al. (2005) studied the effect of varying TRD and observed an increase in SOC and nitrate-N concentrations for the 150% TRD relative to the 0% TRD treatment. Topsoil properties required for sustainable reclamation include high water retention, nutrient availability, resistance to surface erosion, and the ability to support plant root development (Rowland et al., 2009).

Due to the scarcity of topsoil for reclamation and since disturbed soils are typically low in organic matter, approaches such as the use of organic amendments have been implemented to reconstruct and improve the quality of disturbed topsoil, hence promote revegetation (Larney and Angers, 2012). Organic amendments play a significant role in reclaiming degraded soils and fostering the development of healthy ecosystems (Bradshaw, 2000). Various organic amendments, such as alfalfa hay, compost, biosolids, and cattle manure have been used to augment soil physical and chemical properties of disturbed soils, including SOC, available nutrient supply, and moisture retention during the reclamation of disturbed wellsites (Zvomuya et al., 2007; Curtis and Claassen, 2009; Hemstock et al., 2010). Peat is commonly used in the reclamation of disturbed upland forests due to its abundance in such ecosystems (MacKenzie and Naeth, 2011; MacKenzie et al., 2012). For example, peatlands account for approximately 64% of Canada's Athabasca Oil Sands Region in the boreal forest region of Alberta (MacKenzie et al., 2012; Rooney et al., 2012). Peat is, therefore, easily accessible for use in the reclamation of disturbed sites in the area, which reduces transportation costs. Peat improves organic carbon concentration, nutrient availability (Hemstock et al., 2010; Quideau et al., 2017), and water retention properties (Rowland et al., 2009; Ojekanmi and Chang, 2014) of soil, all of which are essential for early vegetation growth. Ojekanmi and Chang (2014) reported a 5 to 35 g $kg^{-1}$ increase in SOC and an increase in water holding capacity after mixing mineral soil with 10 to 50% peat by weight. Recently, biochar has gained popularity as an organic amendment due to its sustainable supply of nutrients and water retention properties during the reclamation process (Mukherjee and Lal, 2014; Page-Dumroese et al., 2016; Dietrich and MacKenzie, 2018). Previous studies on the application of biochar have focused primarily on agricultural land (Laird et al., 2010; Enders et al., 2012; Yang et al., 2017), with a few studies conducted on forest sites (Thomas and Gale, 2015; Page-Dumroese et al., 2018). Biochar can be prepared close to or on-site using tree wood chip residues from site preparation as raw materials, thereby reducing transportation costs (Rowland et al., 2009).

There is a dearth of information on the reclamation of wellsites with suboptimal TRD and organic amendments. Most of the few published studies focused on cropland or were conducted under controlled environments, with little focus on forest ecosystems. Therefore, there is a need to investigate the impact of TRD and organic amendments (peat and biochar) on the productivity of wellsites reclaimed using insufficient topsoil in the boreal zone. The objective of this experiment was, therefore, to evaluate the changes in topsoil chemical properties as a function of TRD and amendment type.

## 2.3 Materials and Methods

### 2.3.1 Study site

`The study was initiated in November 2014 at Maskwa field near Cold Lake (54° 36′ 22.26″ N, 110° 29′ 28.24″ W), Alberta, within the Central Mixedwood Region of the Canadian boreal forest. The Central Mixedwood Region is dominated by tree species such as trembling aspen (*Populus* *tremuloides*), white spruce (*Picea glauca*), and basalm poplar (*Populus balsamifera*), with understory shrub species including green alder (*Alnus viridis*), low bush cranberry (*Viburnum edule*), and red-osier dogwood (*Cornus sericea*) (Natural Regions Committee, 2006). The site was a disturbed borrow pit with suboptimal salvaged topsoil used in the reclamation program. The area is dominated by Luvisolic (Orthic Gray Luvisol) soils, underlain by glacial till parent material with sandy loam and sandy clay loam surface textures and a subsurface texture composed primarily of medium-sized gravel with a few fine-sized gravels. The soil structure at the site ranged from weak fine-grained blocky to medium massive. The drainage classes ranged from poor to well-drained, and the slope classes ranged from nearly level to very gentle.

### 2.3.2 Experimental layout

The experiment was laid out in a randomized complete block design with three blocks representing three replicates. Each block was divided into four 20 m × 20 m plots with 2 m buffers (Fig. S1). The topsoil used for reclamation had been stockpiled for 4 yr (that is, since 2010) prior to placement at the site in November 2014. The original (pre-disturbance) topsoil depth at the site was 28 cm. Thus, the recommended 80% of the original topsoil depth (TRD80) corresponded to a topsoil layer of 22 cm thick while the 50% TRD (TRD50) corresponded to a 14 cm thick topsoil layer. The treatments were the optimal topsoil depth (TRD80) as the control, the suboptimal topsoil depth (TRD50), the 50% TRD amended with peat (PTRD50), and the 50% TRD amended with biochar (BTRD50). One-year stockpiled peat was sourced from a well pad near the study site. Biochar was prepared from dry raw pine tree logs using a slow pyrolysis technique at temperatures between 550 °C and 650 °C for 6 to 12 h. The peat and biochar were mixed with topsoil and spread evenly on each plot using a tractor and rototiller. Peat was applied at 20 kg $m^{-2}$ and biochar at 4.75 kg $m^{-2}$ to bring the TOC in each plot to the level comparable to that in the TRD80 plots.

Amendments were incorporated in all plots in May 2015. Baseline chemical characteristics of the amendments were determined at the start of the experiment prior to application in the plots (Table 2.1).

### 2.3.3 Soil sampling

Soil samples were collected from the 0 to 22 cm layer in the TRD80 plots and 0 to 14 cm in the TRD50 plots prior to amendment application in 2014 for baseline characterization (Table 2.2). Thereafter, soil samples were collected from the 0 to 15 cm layer in August each year. At each sampling, 12 cores were randomly sampled from each plot and composited. Soil cores were collected with a 313 $cm^3$ corer for bulk density determination in the topsoil (TS) horizon in each plot. Subsoil samples were taken using a shovel and a hand auger from the bottom of the topsoil layer to the 1 m depth in the northwestern corner of each plot. The subsoil was divided into upper subsoil (USS, bottom of topsoil horizon to 50 cm) and lower subsoil (LSS, 50 cm to 100 cm).

Table 2. 1 Initial properties of peat and biochar.

| Amendment | TOC[a] | TN | TKN | $NO_3$-N | P | K | S | Ca | Mg | ESP | C/N ratio | pH | SAR | EC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | g $kg^{-1}$ | | | mg $kg^{-1}$ | | | | | | % | | | | dS $m^{-1}$ |
| Biochar | 814 | 10 | 2.3 | 3.0 | 5.0 | 684 | 9.1 | 617 | 60 | 0.3 | 945 | 8.0 | 1.0 | 0.38 |
| Peat | 96 | 51 | 18 | 7.0 | 18 | 135 | 7.5 | 5360 | 691 | 0.6 | 17.2 | 4.3 | 0.67 | 0.23 |

[a] TOC, total organic carbon; TN, total nitrogen; TKN, total Kjeldahl nitrogen; ESP, exchangeable sodium percentage EC, electrical conductivity; SAR, sodium adsorption ratio.

Table 2. 2 Initial (pre-treatment) soil properties.

| | TOC[a] | TKN | Phosphorus | Potassium | Sulfur | Nitrate | EC | Bulk density | SAR | pH | Moisture at saturation |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | g $kg^{-1}$ | | mg $kg^{-1}$ | | | | dS $m^{-1}$ | g $cm^{-3}$ | | | % |
| TRD80 | 26.0 | 1500 | 21.7 | 76.0 | 21.7 | 0.53 | 0.29 | 1.38 | 0.51 | 5.8 | 0.4 |
| TRD50 | 30.5 | 1755 | 22.3 | 76.6 | 22.3 | 0.58 | 0.39 | 1.4 | 0.5 | 5.9 | 0.42 |

[a] TOC, total organic carbon; TKN, total Kjeldahl nitrogen; EC, electrical conductivity; SAR, sodium adsorption ratio.

### 2.3.4 Laboratory analysis

Biochar proximate and ultimate analysis was completed by Loring Laboratories Ltd. (Calgary, Alberta). Fixed carbon, volatile matter, ash, and moisture content were determined using the standard ASTM methods D5357 (ASTM, 2014), D3175, D3174, and D3173 (ASTM, 2011a, 2011b, 2011c).

Soil samples were analyzed by Bureau Veritas Laboratories (Calgary, Alberta). Bulk density was determined using the core method (Blake and Hartge, 1986). Total organic C concentration was determined by the dry combustion method with a Leco TruMac analyzer. Total Kjeldahl nitrogen (TKN) concentration was determined according to EPA method 351.2 (USEPA, 1993). A 25-mL aliquot of the sample was heated and digested with concentrated (18 M) $H_2SO_4$ for 2.5 h in a block digester. For available N (sum of extractable $NH_4$-, $NO_2$-, and $NO_3$-N) determination, the soil was extracted with 2 M KCl (1:10 soil to solution ratio) (Maynard et al., 2006). Available $NO_2$- and $NO_3$-N concentrations were determined colorimetrically (cadmium reduction) using a Technicon TrAAcs 800 autoanalyzer (Technicon Industrial Systems Corp., Tarrytown, NY, USA). Available P and K concentrations in the soil were determined using the modified Kelowna extraction method (Qian et al., 1994) in which the two nutrients were extracted using 0.25 M ammonium acetate, 0.015 M ammonium fluoride, and 0.025 M glacial acetic acid (Kelowna extraction), followed by analysis using a Varian Vista Pro ICP-OES spectrometer (Varian Inc., Palo Alto, California, USA). Available S was extracted using a 2:1 ratio (vol./mass) of 0.01 M $CaCl_2$ and air-dry ground soil (Houba et al., 2000) and measured using a Varian Vista Pro ICP-OES spectrometer. Soil pH and electrical conductivity (EC) were measured with a pH/EC meter in a 1:2 (vol/vol) soil:water suspension (Janzen, 1993; Rhoades, 1996). Water-soluble calcium (Ca), magnesium (Mg), and sodium (Na) were extracted using the EPA 200.7 method (USEPA, 1994) followed by ICP-OES analysis. Subsoil samples were analyzed for EC, pH, and water-extractable cations in 2015, 2017 and 2019. The moisture content at saturation (that is, the weight of water required to saturate the pore space divided by the weight of the dry soil, multiplied by 100) was determined using the saturated paste method (Rhoades, 1996).

### 2.3.5 Statistical analysis

Analysis of variance (ANOVA) was carried out using the generalized linear mixed model procedure (PROC GLIMMIX) of SAS 9.4 (SAS Institute, 2013). Data for all soil properties followed a normal distribution, with the exception of available S, which followed a lognormal distribution, and moisture content at saturation, which followed a beta distribution. Treatment, year, and horizon were modeled as fixed effects, with year as a repeated measures factor, while block was a random effect. Based on the Akaike Information Criterion (Littell et al., 1996), the compound symmetry (CS) covariance structure was selected for repeated measures ANOVA of TKN, EC, SAR, pH and soluble cations data while the heterogeneous compound symmetry [CSH] covariance structure was the most suitable for TOC, available P, and S. Treatment means were compared using the Tukey multiple comparison procedure at $\alpha = 0.05$.

## 2.4 Results and Discussion

### 2.4.1 Weather conditions

Cumulative precipitation during the growing season (May to October) ranged from 304 to 366 mm annually (Fig. 2.1). The 30-yr (1981-2010) mean cumulative precipitation for the growing season (May to October) at Cold Lake is 308 mm. The highest mean growing season temperature of 14.5 °C was recorded in 2015, while the lowest mean air temperature was 11.5 °C in 2019, which was slightly below the long term 30-yr (1981-2010) growing season mean temperature of 12.2 °C (Fig. 2.1) (Environment Canada, 2020).

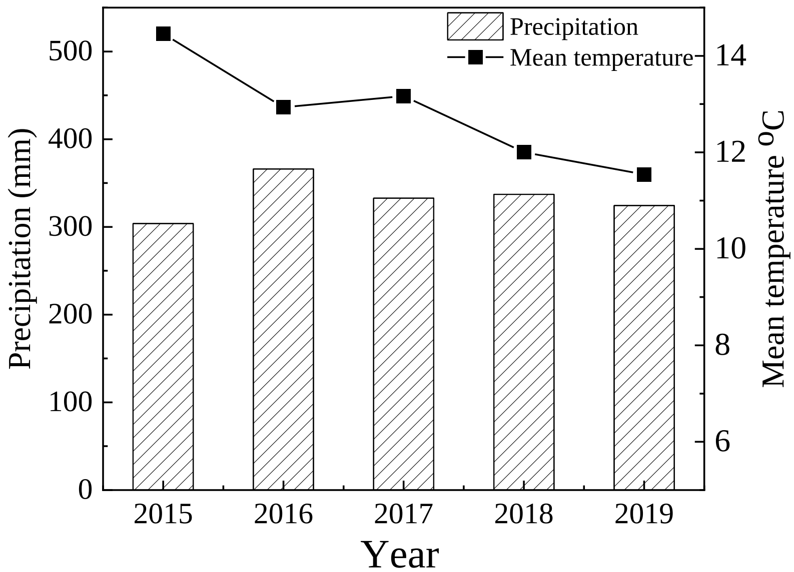


Figure 2. 1 Growing season (May to October) precipitation and mean growing season temperature during 2015-2019 at the borrow pit at Cold Lake, Alberta, Canada.

### 2.4.2 Bulk density

Bulk density was significantly ($P = 0.003$) higher for the TRD80 (control) than the peat (PTRD50) and biochar (BTRD50) treatments but did not differ significantly between the TRD80 and the TRD50 treatments (Table 2.3). However, the addition of peat or biochar to the 50% TRD significantly reduced bulk density by 27 and 15%, respectively than the TRD80. The decrease in bulk density was attributed to the peat and biochar's light weight and high pore volume properties (Lehman et al., 2003; Cao, 2019). Mean bulk density ranged from 0.95 g $cm^{-3}$ in the PTRD50 to 1.3 g $cm^{-3}$ in the TRD80 treatment (Table 2.3). However, these bulk densities were lower than the pre-disturbance bulk density of 1.5 g $cm^{-3}$ in the Ae horizon of the Orthic Gray Luvisolic soil at the site (Solstice, 2010). Although the TRD80 had the highest bulk density, it was still within acceptable limits to support plant establishment and growth.

Our results corroborate those of Forsch (2014), who reported significantly lower bulk density (0.89 g $cm^{-3}$) in plots reclaimed with peat mineral mix compared to the control plots. Similarly, Ojekanmi and Chang (2014) reported a decrease in bulk density from 1.4 Mg $m^{-3}$ to 0.7 Mg $m^{-3}$, and a concomitant increase in SOC in peat-amended soils. Githinji (2014) observed a reduction in bulk density from 1.33 g $cm^{-3}$ to 0.89 g $cm^{-3}$ when biochar rate of application was increased from 25 to 50% (v/v). Mukherjee et al. (2014) reported a 24% decrease in bulk density with biochar application to degraded soils. Several other studies have also demonstrated a reduction in bulk density in soils amended with biochar (Chen et al., 2011; Mukherjee and Zimmerman, 2013; Hardie et al., 2014; Burrell et al., 2016).

### 2.4.3 Total organic carbon and total Kjeldhal nitrogen

Unsurprisingly, organic amendment (peat and biochar) application significantly ($P < 0.0001$) increased soil TOC concentration relative to the TRD50 and TRD80 treatments, but there was no significant difference in soil TOC concentration between the two amendments (PTRD50 and BTRD50) (Table 2.3). Additionally, TOC concentration did not differ significantly between the TRD80 and the TRD50 treatment. Total organic C concentration varied significantly ($P < 0.0001$) during the 5-yr study, but there were no significant temporal trends. Total organic C concentration increased significantly by 43 and 49% in 2016 and 2018, relative to 2019. Changes in TOC concentration in the soil are likely dynamic over time, varying in response to plant biomass and litter accumulation (Reeder et al. 1998). Reeder et al. (2001) concluded that increase in organic C concentrations in soil are most likely due to the accumulation of vegetation biomass over time.

The higher TOC concentrations in plots amended with peat and biochar reflect the C additions from these amendments (Table 2.1). Ojekanmi and Chang (2014) observed an increase of 5.0 to 35 g $kg^{-1}$ in SOC when peat and mineral soil were mixed in proportions ranging from 10 to 50% peat by weight. Similarly, Hemstock et al. (2010) reported a higher concentration of TOC in fibric peat-amended reclaimed soils than at undisturbed sites in oil sands region, Alberta. Their findings indicated that a higher C/N ratio observed in fibric peat resulted in higher soil C levels than other peat amendments. Schultz et al. (2017) reported a 62.2 to 95.5 μg $g^{-1}$ increase in TOC concentration following biochar application to sodic soils during reclamation. In a greenhouse study on reconstructing topsoil functionality following disturbance, Bekele et al. (2015) observed a slight increase in SOC in subsoils amended with biochar compared with those amended with humalite. Dumroese et al. (2018) reported significantly higher C concentrations in plots amended with biochar powder (74%) and pyrolyzed softwood pellet biochar (91%) relative to peat-amended plots (53%). These and our results indicate that amending suboptimal disturbed topsoil with peat or biochar can increase TOC concentration in reclaimed soils relative to the mandated TRD80 and may be used in the restoration and rebuilding of a healthy ecosystem. Both peat and biochar are recalcitrant and resist rapid microbial decomposition, supplying C and other nutrients slowly and thus sustaining soil productivity over time (Baldock and Smernik, 2002; Hemstock et al., 2010; Enders et al., 2012).

Topsoil replacement depth had no significant effect on TKN concentration (Table 2.3). However, peat amendment significantly influenced the TKN concentration relative to the other three treatments as indicated by a significant treatment effect ($P = 0.03$) (Table 2.3). Mean TKN concentration was 143, 87 and 116% higher for the PTRD50 treatment relative to the TRD80, TRD50 and BTRD50 treatments, respectively. Although the TRD80 treatment had the lowest TKN concentration, it was not significantly different from the TRD50 and the BTRD50 treatments (Table 2.3). Higher TKN concentrations observed in the peat-amended plots likely reflect the higher initial TKN concentrations in these plots compared to those that received biochar (Table 2.1). Ojekanmi and Chang (2014) reported an increase of up to 0.56% in total N concentration as the rate of peat application increased during reclamation in the oil sands region of Alberta, Canada. Similarly, McMillan et al. (2007) reported higher total N concentrations for reclamation treatments amended with a peat-mineral mix. Quideau et al. (2017) observed a sevenfold increase in total N concentration in soils amended with peat relative to forest-floor mix during mine reclamation in the oil sands region of Alberta.

The concentration of TKN changed significantly over time and was highest in 2016 and lowest in 2019 (Table 2.3). Higher TKN concentrations in 2016 were likely due to increased nutrient transformations including mineralization and accumulation of N occurring in the early years following topsoil and amendment placement. Additionally, litter accumulation from the vegetation in the plots may have contributed to N accumulation in the topsoil over time. Our results are consistent with those of Moss et al. (1989), who reported a significant increase in TKN concentration in plots amended with sludge and sawdust relative to unamended plots after 5 yr. Bendefelt et al. (2001) sampled the same plots after 16 yr and found a three-fold increase in TKN concentration in unamended soils relative to organically amended plots. The authors attributed the increase in TKN in the unamended plots to the accumulation of N over time from decomposing litter.

Table 2. 3 Topsoil replacement depth and organic amendment effects on total organic carbon (TOC), total Kjeldhal nitrogen (TKN), available phosphorus, potassium, sulphur, and bulk density after amendment application.

| Effect | Bulk density | TOC[a] | TKN | Available P | Available K | Available S |
|---|---|---|---|---|---|---|
| | g $cm^{-3}$ | g $kg^{-1}$ | | mg $kg^{-1}$ | | |
| Treatment | | | | | | |
| TRD80 | 1.3a[b] | 20.2b | 1356b | 13.3 | 78.1ab | 5.9 |
| TRD50 | 1.2ab | 28.3b | 1760b | 15.7 | 79.5ab | 8.8 |
| PTRD50 | 0.95c | 44.4a | 3290a | 15.0 | 74.9b | 5.7 |
| BTRD50 | 1.1bc | 45.8a | 1520b | 14.3 | 92.9a | 7.5 |
| Year | | | | | | |
| 2015 | 1.14 | 28.8ab | 1717bc | 12.6b | 80.4ab | 6.2b |
| 2016 | 1.19 | 44.0a | 2182a | 19.1a | 83.1ab | 12.8a |
| 2017 | 1.14 | 32.7ab | 1803abc | 13.5b | 69.6b | 6.1b |
| 2018 | 1.07 | 45.0a | 2077ab | 12.0b | 76.3ab | 6.3b |
| 2019 | 1.16 | 22.9b | 1558c | 15.7ab | 93.3a | 4.8b |
| | | | P value | | | |
| Treatment | **0.003** | **<0.0001** | **0.03** | 0.71 | **0.04** | 0.42 |
| Year | 0.07 | **<0.0001** | **0.0004** | **0.002** | **0.03** | **0.003** |
| Treatment × year | 0.09 | 0.79 | 0.41 | 0.08 | 0.13 | 0.20 |

[a] TOC, total organic carbon; TKN, total Kjeldahl nitrogen; TRD80, 80% topsoil replacement depth, TRD50, 50% topsoil replacement depth, PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar.

[b] Means within a column followed by the same letter are not significantly different at α = 0.05 according to the Tukey multiple comparison procedure.

Although the change in TKN occurred over a short duration (5 yr) in our study, it was likely enhanced by the accelerated vegetation litter decomposition following reclamation.

As N is the most limiting nutrient in disturbed boreal forest soils where native organic matter is lacking (Bradshaw, 1987), N availability is correlated with the amount of organic matter available for the mineralization process. Organic amendments containing recalcitrant organic C, such as peat, may gradually increase the supply of mineralizable N, enhancing the restoration of disturbed sites (Jamro et al., 2014).

### 2.4.4 Available phosphorus, potassium, and sulfur

Available P (modified Kelowna-extractable P, MKP) concentration did not vary significantly with treatment ($P = 0.07$), regardless of year ($P = 0.08$ for the treatment × year interaction) (Table 2.3). However, MKP concentration was significantly higher in 2016 (19.1 mg $kg^{-1}$) than in all the other years except 2019, for which the MKP concentration did not differ significantly from that in 2016. The higher MKP concentrations in 2016 could be attributed partly to rapid mineralization occurring in the early years following topsoil replacement. Additionally, adequate moisture and warm temperatures conducive to microbial activity likely increased P mineralization in the soil. Reducing TRD from the mandatory 80% (TRD80) to 50% (TRD50) with or without organic amendments had no significant effect on available MKP concentration. In contrast Larney et al. (2005) reported a significantly higher available P concentration with 150% TRD (48 mg $kg^{-1}$) than with 0% TRD (control) and 50% TRD (32-35 mg $kg^{-1}$) 2 yr following reclamation at a wellsite in southcentral Alberta, Canada. They also observed significantly greater available P concentrations in soils amended with compost (62 to 82 mg $kg^{-1}$) than those amended with alfalfa hay (26 mg $kg^{-1}$ to 49 mg $kg^{-1}$). Similarly, Zvomuya et al. (2007) reported a 3.24 kg $ha^{-1}$ increase in MKP accumulation for each Mg $ha^{-1}$ increase in compost rate in the 0- to 30-cm soil layer. Their findings may differ from ours since the amendments used were not identical to ours. As a result, their amendments may have the potential to add more phosphorus to the soil than peat and biochar used in our study.

Available K concentration was significantly ($P = 0.04$) greater for BTRD50 than for the PTRD50 treatment (Table 2.3). However, available K concentration did not differ significantly between TRD80 and TRD50. Available K concentration was significantly greater in 2019 than in 2017 but did not differ significantly between 2019 or 2017 and the other three years. The higher available K concentration in the BTRD50 treatment may be due to the higher initial K concentration in biochar compared to peat (Table 2.1). Our findings corroborate those of Dumroese et al. (2018), who found that soils amended with various biochars had four to ten times higher soluble K concentrations compared with peat-amended soils. Similarly, other studies have also shown increased concentrations of available K with biochar application, indicating that biochar can act as a source of K and thus improve K availability when applied to the soil (Altland and Locke, 2013; Zhang et al., 2014; Wang et al., 2018). Additionally, Dietrich and MacKenzie, (2018) reported a significant increase in soil K availability in a peat-mineral mix cover soil amended with biochar than those amended with forest-floor mix without biochar in the Athabasca oil sands region of Alberta.

Topsoil replacement depth and organic amendment treatments had no significant effect ($P = 0.42$) on available S concentration but varied with year ($P = 0.003$) (Table 2.3). Available S concentration was significantly greater in 2016 than in all the other years (Table 2.3). The lack of organic amendment effect was expected since the two amendments had similar initial S concentrations (Table 2.1). In comparison, Dietrich and MacKenzie (2018) observed an increase in available S concentration in soils amended with a peat-mineral mix with or without biochar amendment relative to a forest-floor mineral mix without biochar.

### 2.4.5 Electrical conductivity, sodium adsorption ratio, soil pH, and water-extractable cations

Electrical conductivity did not vary significantly (P = 0.08) with TRD and organic amendment treatment (Table 2.4). However, EC was significantly greater in the LSS horizon than the TS and USS horizons in 2019, whereas the horizon effect was not significant in 2015 and 2017 (Fig. 2.2a). The increase in EC in the LSS horizon in 2019 reflects the leaching of soluble salts from the overlying soil layers to this horizon over time (Rhoades, 1996). Wang et al. (2014) observed a gradual decrease in salt content in the 0 - 15 cm soil layer with time elapsed following reclamation. Similarly, Merrill et al. (2021) reported a decline in EC 28 yr after the reclamation of a mine site in North Dakota, USA.

While organic amendments had no significant effect on EC in our study, Page-Dumroese et al. (2018) reported significantly lower EC in the 0-3 cm layer in plots amended with biochar than those amended with biosolids 19 mo after amendment application. Overall, the EC values in our study were generally low (< 4 dS $m^{-1}$) and within the acceptable range for most plants and, therefore, unlikely to cause adverse effects on vegetation performance.

There was a significant (P < 0.0001) horizon by year interaction for SAR (Table 2.4). While there was no horizon effect on SAR in 2015, SAR in 2019 was significantly higher in the USS horizon than in the TS and LSS horizons. By comparison, SAR in 2017 was significantly greater in the USS than the TS but did not differ significantly between the USS and the LSS (Fig. 2.2b). Across the sampling period, SAR in the USS horizon increased by 43% in 2019 relative to 2015, while there was no significant change in SAR in the TS and LSS horizons over time (Fig. 2.2b). Similar to our findings, (Leskiw et al., 2012) reported low SAR (< 7) in the leaf litter and A horizons and significantly higher SAR (> 14) in the B and C horizons of a boreal soil 5 yr after reclamation. Mackenzie and Naeth (2019) observed an increase in SAR with increasing soil depth. In our study, SAR did not vary significantly with TRD and organic amendment treatment (Table 2.4). Overall, SAR values were low, ranging from 0.5 to 0.8, and fell within acceptable limits for most plants (Alberta Environment, 2001).

Water-extractable Ca, Mg, and Na concentrations did not vary significantly with TRD and organic amendment treatment (Table 2.4). Water-extractable Na concentration was significantly lower in the TS horizon than in the USS and LSS horizons. There was a significant horizon × year interaction for water-extractable Ca and Mg. In the last year of monitoring (2019), water-extractable Ca and Mg concentrations in the LSS horizon were significantly higher than in the TS and USS horizons while there was no significant difference between horizons in 2015 and 2017. Water-extractable Ca and Mg concentrations increased in the LSS horizon and decreased in the USS horizon over time, but the difference between the two horizons was only significant in 2019 (Figs. 2.2c and 2.2d). The high concentrations of Ca and Mg in the LSS than in the USS may be due to mineralization of the underlying parent material, which may have increased the Ca and Mg levels over time. Kabrick et al. (2011) reported an overall increase of Ca and Mg concentrations at depths less than 1 m as a result of the weathering of the underlying dolomite bedrock in forest soils. Additionally, the decrease in these concentrations in the USS indicates that the soluble cations were leaching down the soil profile over time.

Table 2. 4 Changes in selected chemical properties with topsoil replacement depth and organic amendment at the reclaimed borrow pit site.

| **Effect** | **EC[a]** | **SAR** | **pH** | **Moisture at saturation** | **Soluble Ca** | **Soluble Mg** | **Soluble Na** |
|---|---|---|---|---|---|---|---|
| | $dSm^{-1}$ | | | % | mg $L^{-1}$ | | |
| Treatment | | | | | | | |
| TRD80 | 0.39 | 0.66 | 7.2 | 36 | 38.1 | 12.8 | 17.9 |
| TRD50 | 0.38 | 0.7 | 7.2 | 37 | 35.1 | 12.4 | 18.2 |
| PTRD50 | 0.4 | 0.73 | 7.2 | 41 | 35.4 | 11.7 | 18.5 |
| BTRD50 | 0.43 | 0.73 | 7.3 | 39 | 39.1 | 12.7 | 19.5 |
| Horizon | | | | | | | |
| TS | 0.33 | 0.61 | 6.5 | 42 | 29.8 | 11.2 | 15b[b] |
| USS | 0.36 | 0.8 | 7.6 | 35 | 32.4 | 9.9 | 19.5a |
| LSS | 0.52 | 0.69 | 7.6 | 37 | 51.3 | 16.9 | 21.6a |
| Year | | | | | | | |
| 2015 | 0.41 | 0.59 | 7.2 | 36 | 35.7 | 12.2 | 15.4 |
| 2017 | 0.39 | 0.75 | 7.3 | 38 | 38 | 13.1 | 20.7 |
| 2019 | 0.39 | 0.8 | 7.2 | 41 | 36.8 | 11.8 | 19.9 |
| | P value | | | | | | |
| Treatment | 0.81 | 0.21 | 0.48 | < **0.001** | 0.92 | 0.96 | 0.71 |
| Horizon | < **0.001** | < **0.001** | < **0.001** | < **0.001** | < **0.001** | < **0.001** | < **0.001** |
| Year | 0.85 | **0.01** | 0.23 | < **0.001** | 0.94 | 0.85 | 0.07 |
| Treatment × Year | 0.72 | 0.08 | 0.9 | 0.21 | 0.15 | 0.12 | 0.26 |
| Treatment × Horizon | 0.31 | 0.68 | **0.01** | < **0.001** | 0.4 | 0.42 | 0.14 |
| Horizon × Year | **0.01** | < **0.001** | 0.9 | < **0.001** | **0.003** | **0.003** | 0.17 |
| Treatment × Horizon × Year | 0.74 | 0.45 | 0.8 | 0.89 | 0.84 | 0.69 | 0.08 |

[a] EC, electrical conductivity; SAR**,** sodium adsorption ratio; TRD80, 80% topsoil replacement depth, TRD50, 50% topsoil replacement depth, PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar; TS, topsoil horizon; USS (upper subsoil, i.e., bottom of topsoil horizon to 50 cm); LSS, lower subsoil (i.e., 50 to 100 cm layer).

[b] Means within a column followed by the same letter are not significantly different at α = 0.05 according to the Tukey multiple comparison procedure.

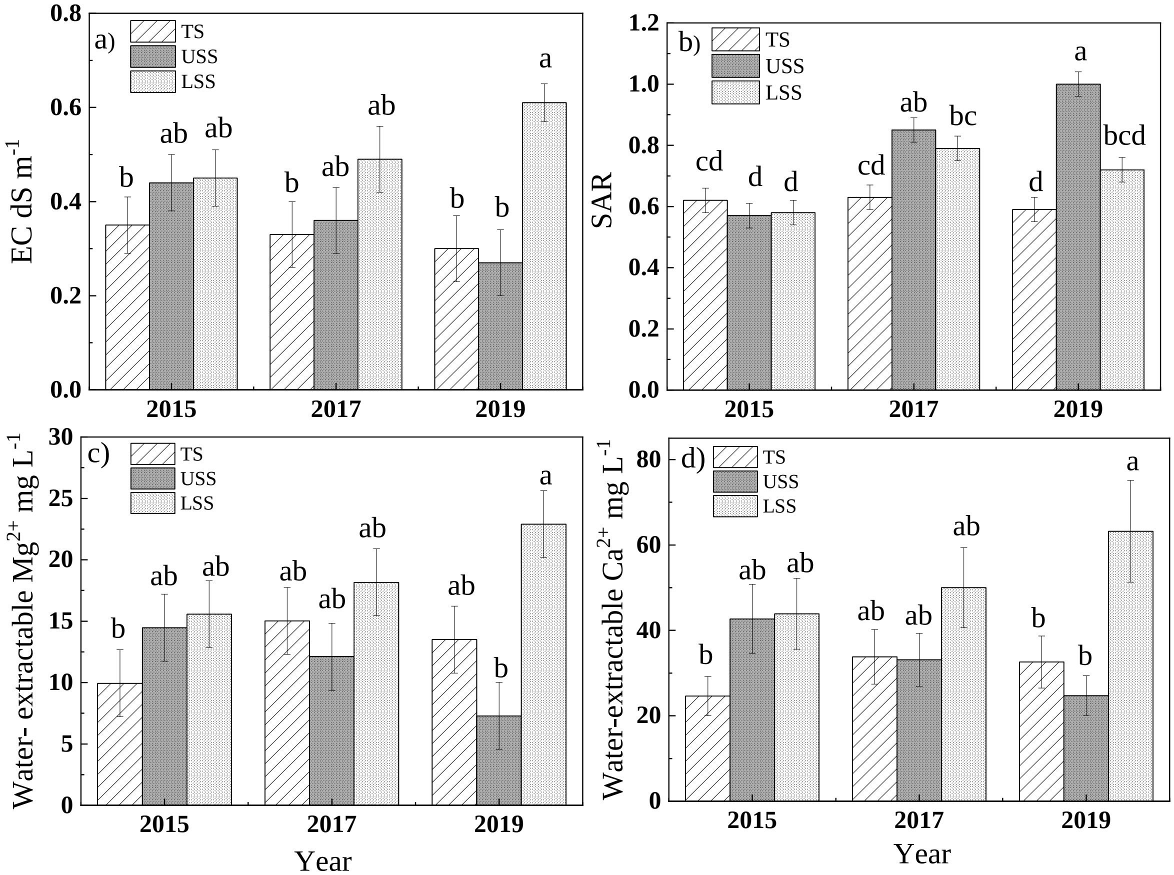


Figure 2. 2 Horizon by year interaction effect on (a) electrical conductivity (EC), (b) sodium adsorption ratio (SAR), (c) water-extractable Mg, and (d) water-extractable Ca following reclamation of a borrow pit. Error bars represent standard errors of the mean. Bars with the same letters are not significantly different at $\alpha = 0.05$ according to the Tukey multiple comparison procedure. LSS, lower subsoil horizon; USS, upper subsoil horizon; TS, topsoil horizon.

The TRD and organic amendment treatments had no significant ($P = 0.48$) effect on soil pH, but there was a significant ($P < 0.0001$) treatment × horizon interaction (Table 2.4). Soil pH was significantly lower for the PTRD50 than the BTRD50 treatment in the TS horizon but there was no significant treatment effect in the USS and LSS horizons (Fig. 2.3a). Although the PTRD50 treatment had a lower soil pH, it did not differ significantly from the TRD80 and TRD50 treatments in the TS horizon. By comparison, Vano et al. (2011) reported a decrease in soil pH to optimum levels of 4.0 to 5.2 in peat moss, sawdust compost, and ferrous sulfate amended soils relative to the soils amended with ammonium sulphate only. In contrast, biochar has been reported to increase soil pH due to its liming properties (Ippolito et al., 2012, 2014; Bednik et al., 2020). Another mechanism by which biochar increases soil pH is through chemical reactions between its oxygen-containing functional groups and $H^+$ ions in the soil, which results in the formation of OH groups, lowering the $H^+$ concentration and raising the soil pH (Chintala et al., 2014). The soil pH across all treatments averaged 7.2, which is suitable for optimum growth of most plants (Sheoran et al., 2010).

### 2.4.6 Soil moisture content at saturation

There were significant ($P < 0.0001$) treatment × horizon and horizon × year interactions for soil moisture content at saturation (Table 2.4). The moisture content at saturation was significantly higher for the PTRD50 and BTRD50 than the TRD50 and TRD80 treatments in the TS horizon but did not vary significantly among treatments in the USS and LSS horizons (Fig. 2.3b). There was no significant difference in moisture saturation between horizons for the TRD50 and TRD80 treatments (Fig. 2.3b). Peat and biochar addition to the 50% TRD treatment, on the other hand, significantly increased moisture content at saturation in the TS horizon by 48 and 32%, respectively, relative to the TRD80 treatment. Greater moisture at saturation observed in the PTRD50 reflects the large pore volume and a high specific surface area of peat, which increases water retention capacity to a greater extent than biochar (Vano et al., 2011; Petelina et al., 2014; Rezanezhad et al., 2016). In situ and laboratory studies by Moskal et al. (2001) showed that increasing the ratio of peat to mineral soil from 1:3 to 3:1 increased field capacity, plant available water, and water holding capacity in reclaimed soils in the Oil Sands Region of Alberta. Petelina et al. (2014) reported eight times greater water holding capacity in peat-amended than in biochar-amended reclaimed soils in field trials evaluating amendments for land reclamation near Lake Athabasca, Alberta.

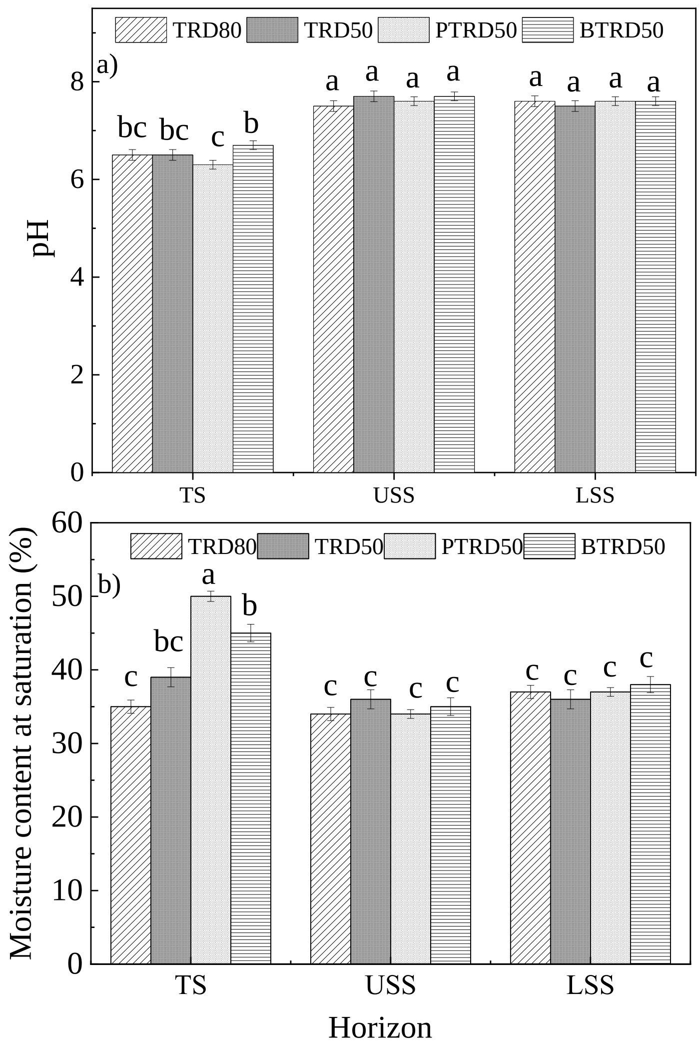


Figure 2. 3 Effect of TRD and organic amendment treatments on (a) soil pH and (b) moisture content at saturation in the TS, USS and LSS horizons averaged over time (2015 to 2019) following reclamation of a borrow pit near Cold Lake, Alberta. Error bars represent standard errors of the mean. Bars with different letters are significantly different according to the Tukey multiple comparison procedure. TS, topsoil horizon; USS, upper subsoil (bottom of topsoil horizon to 50 cm); LSS, lower subsoil (50 cm to 100 cm).

Although moisture content at saturation was significantly greater for peat than biochar in our study, biochar has been shown to increase water retention properties of mineral soils due to its high pore volume and specific area (Zimmerman et al., 2011; Page-Dumroese et al., 2016, 2018). Ramlow et al. (2018) reported a 26% increase in soil moisture retention in soils amended with wood biochar compared to the control (unamended). Additionally, Mukherjee et al. (2014) reported a 63% increase in plant-available water holding capacity in oakwood biochar-amended silty loamy soils in a field study, in Ohio, USA. Hardie et al. (2014) reported an increased total porosity and saturated water content in soils amended with green waste biochar at 47 Mg $ha^{-1}$.

Moisture content at saturation across all treatments increased significantly across from 2015 through 2017 to 2019 in the TS horizon but did not change significantly with year in the USS and LSS horizons (Fig. 2.4), indicating the importance of numerous pores with the ability to store moisture in the topsoil relative to the subsoil horizons (Dhar et al., 2022).

Our findings indicate that amending disturbed soils with organic amendments may improve water retention and saturation capacity compared to unamended soils. Similarly, Leatherdale et al. (2012) concluded that soils with high amounts of organic matter have higher water retention capacity than mineral soils with low organic matter content. Additionally, other organic amendments, such as forest floor, at various rates should be tested for their potential to augment boreal site reclamation using suboptimal topsoil.

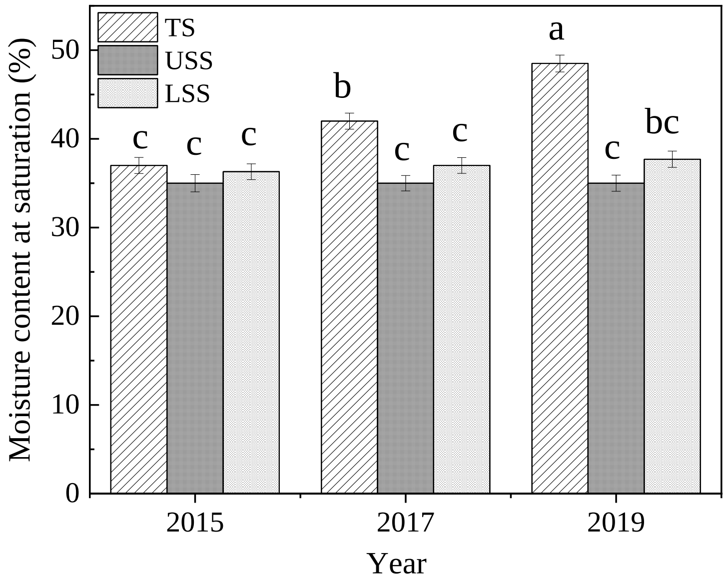


Figure 2. 4 Moisture content at saturation in the TS, USS, and LSS horizons over time (2015 to 2019) following reclamation of a borrow pit near Cold Lake, Alberta. Error bars represent standard errors of the mean. Bars with different letters are significantly different according to the Tukey multiple comparison procedure.

## 2.5 Conclusion

Decreasing topsoil replacement depth from 80% to 50% with no amendment had minimal effects on soil chemical properties following reclamation of the borrow pit. Although some treatments produced higher analyte concentrations in the soil, their effects varied from year to year with no consistent temporal trends. However, amending the suboptimal topsoil with peat (PTRD50) and biochar (BTRD50) significantly improved TKN, TOC, and moisture content at saturation in the topsoil compared with the TRD80 and TRD50 treatments. There was an increase in EC in the LSS horizon and SAR in the USS horizon after 5 yr, indicating some level of leaching of salts down the profile over time. Nonetheless, the EC and SAR values were low and not expected to adversely impact vegetation performance. Comparing our findings to the pre-disturbance site soil characteristics assessment using the 2010 criteria for forest ecosystem restoration, we found that all topsoil and subsoil measurements passed level assessment except for topsoil depth, which was less than optimal for the 50% TRD treatments. Despite the topsoil depth limitation, our findings indicate that reclamation with suboptimal (50%) TRD with or without organic amendments can produce acceptable soil property outcomes. However, the addition of peat and biochar does improve soil quality relative to unamended topsoil during reclamation when salvaged soil is insufficient to achieve the optimal TRD80. Since soil nutrient status evolves over time, continued monitoring will provide valuable insights into the long-term effects of TRD and these organic amendments on soil properties and function.

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# 3 REVEGETATION OF WELLSITES RECLAIMED WITH SUBOPTIMAL TOPSOIL REPLACEMENT DEPTH AND ORGANIC AMENDMENTS

## 3.1 Abstract

Reclamation success of disturbed boreal sites depends primarily on the availability of salvaged topsoil that can be used for reclamation. Alberta's reclamation regulations require a minimum of 80% topsoil replacement depth (TRD80) for successful reclamation of disturbed wellsites. However, this is not always attainable for some wellsites with insufficient salvaged topsoil. This 5-yr study examined the efficacy of organic amendments to augment revegetation of wellsites reclaimed with suboptimal salvaged topsoil. Specifically, we evaluated the response of vegetation performance and survival to 50% topsoil replacement depth without organic amendment (TRD50) or amended with either peat (PTRD50) or biochar (BTRD50), relative to the mandatory TRD80 treatment, following wellsite reclamation at Cold Lake, Alberta, Canada. Tree and shrub seedling mixes were transplanted into all plots. Native plant canopy cover was significantly greater for the PTRD50 and the TRD80 treatments than the BTRD50 treatment. Canopy cover for forb and non-native species decreased significantly whereas those for graminoid and native species increased over time. Our results showed that, across all treatments, native species richness increased by 10% per year while non-native species richness decreased by 19% per year. Tree height increased significantly with time and was significantly greater for TRD80 and PTRD50 than for BTRD50 and TRD50. Throughout the study, aspen and green alder species had survival rates below 50% while other tree and shrub species had high survival rates (69 to 98%). Overall, the peat treatment (PTRD50) produced similar vegetation performance results to the mandatory TRD80 treatment, indicating that peat amendment can improve reclamation success at disturbed boreal sites where salvaged soil is insufficient to achieve the optimal 80% TRD.

### 3.2 Introduction

Canada is the fourth-largest oil producer after the United States, Saudi Arabia, and Russia (EIA, 2021). Approximately 165 billion barrels of recoverable bitumen occupying a total area of 142 000 $km^2$ underlies the boreal forest region of Alberta, Canada (Alberta Energy 2014; Government of Alberta 2021; Natural resources Canada 2017). However, the extraction of these non-renewable energy resources results in significant soil and vegetation disturbances (Aerts and Honnay, 2011; Macdonald et al., 2015; Bork et al.,2021).

During oil well site development, disturbances, including vegetation removal and disruption of wildlife habitats, increasingly disrupt the ecological integrity and biodiversity of Alberta's boreal forest region (Schneider & Dyer, 2006). This necessitates reclamation of the disturbed land to a function level that existed before the disturbance. Plant succession is one of the measures of successful restoration of disturbed forests (Řehounková and Prach 2008; Dhar et al., 2018, 2020b), while other researchers consider species diversity and richness as important measures (Rokich et al., 2000; Rivera et al., 2014; Das Gupta and Pinno, 2020). Alteration of soil physiochemical properties due to disturbance slows down natural succession with the result that it takes several decades to recover the ecosystem (Harper and Kershaw, 1996; Řehounková and Prach, 2008). Some reclamation techniques can quicken the restoration process of disturbed forest ecosystems. However, the reclamation of anthropogenically disturbed forest sites is complex and requires the availability of appropriate resources to support successful restoration to a level of equivalent land capability (Fridley, 2002; Macdonald et al., 2012; ESRD, 2013).

The main goal of reclaiming disturbed well sites is to develop and re-establish fully functional and self-sustaining forest plant communities. Rebuilding the plant community and ecological resilience often entails planting native trees and understory vegetation that will support the diversity of the ecosystem (Mackenzie and Naeth, 2009; Macdonald et al., 2012, 2015a). However, revegetation efforts at disturbed sites are hampered by the scarcity of quality topsoil and organic matter needed to sustain plant growth during early establishment (Bowen et al., 2005; Mackenzie and Naeth, 2009; MacKenzie et al., 2012). During the early 1990s, oil companies were not mandated to implement pre-disturbance measures such as topsoil salvage and stockpiling. Instead, the topsoil removed during wellsite development was used for other purposes instead of being stored for future reclamation (Powter et al., 2012). Consequently, many older well sites have insufficient or no salvaged topsoil for use during reclamation to promote plant growth during the revegetation process.

The amount of topsoil returned to the site during reclamation is the primary factor influencing successful revegetation, as soil quantity and fertility influence plant growth (Yang et al., 2019). Several studies have demonstrated reduced plant growth following reclamation using suboptimal topsoil volumes (Larney et al., 2003; Yang et al., 2019). Suboptimal topsoil depths, particularly those attained using stockpiled soil, are often characterized by nutrient deficiencies, low organic carbon content, poor drainage, and limited availability of viable native seeds or propagules, all of which can impede plant establishment and growth (Mackenzie and Naeth, 2009; MacKenzie et al., 2012; Yang et al., 2019). On the other hand, exceeding the optimum topsoil depth may be expensive as it may not result in any significant increase in soil biological or microbial activity or plant growth (ESRD, 2013; Yang et al., 2019). Published research indicates that the determination of topsoil replacement depth in the short term should be based on the plant establishment and growth requirements for successful restoration (Dhar et al., 2018), but in the long term, emphasis should be on sustainable and stable plant communities to minimize invasion of native boreal forests by non-native species and topsoil erosion (Bowen et al., 2005; Wick et al., 2011).

Current regulations in western Canada require the replacement of at least 80% of the pre-disturbance topsoil depth (ESRD, 2013; Powter et al., 2012). The provision of adequate topsoil is essential for the re-establishment of functional forest ecosystems (Angel et al., 2005, 2017; Burger et al., 2005). Topsoil substrate that supplies and retains adequate nutrients and water is critical in early vegetation establishment and survival in reclaimed forests (Macdonald et al., 2012; Dhar et al., 2019). While most of the soil present at disturbed wellsites is typically of low quality, alternative cost-effective reclamation efforts using organic amendments to augment restoration of disturbed forest sites have been implemented (Rowland et al., 2009; Brown and Naeth, 2014; Page-Dumroese et al., 2018).

Several organic amendments have been used for reclamation, with peat being the most used to restore disturbed forest ecosystems. The abundance of peat in the oil sands region has attracted its use in upland reclamation and revegetation practices (Rowland et al., 2009; Errington and Pinno, 2015; Calver et al., 2019). Peat supplies soil nutrients like nitrogen (N) and phosphorus (P) (Hemstock, 2008; Hemstock et al., 2010; Calver et al., 2019) and improves soil water holding capacity (Moskal et al. 2001; Li et al. 2020). While peat is commonly used in reclamation, the majority of previous studies focused on the use of peat mineral mix in restoration following oil sand mining (Rowland et al., 2009; Hahn and Quideau, 2013; Dhar et al., 2020a). Pinno et al. (2012) reported a significant increase in tree (trembling aspen) height, biomass, and foliar N concentration on reclaimed soils amended with peat following oilsands reclamation in Alberta.

Recently, biochar has attracted greater attention from forest researchers as a potential cost-effective organic amendment that can be used in the reclamation and revegetation of disturbed sites (Page-Dumroese et al., 2016, 2018). Previous studies showed increased water holding capacity (Page-Dumroese et al., 2016, 2018), sustainable nutrient supply, high organic carbon (Mukherjee and Zimmerman, 2013), and increased CEC (Glaser et al., 2002; Joseph et al., 2010) from soils amended with biochar during reclamation. Page-Dumroese et al. (2018) reported improved total percent vegetation cover when biochar was applied, with a smaller percentage of bare ground (23%) two years after reclamation at a mine site in northeastern Oregon, USA. Dietrich and MacKenzie (2018) reported a significant increase in soil K and trembling aspen (*Populus tremuloides*) seedling growth on reclaimed peat-mineral mix cover soil mixed with biochar in the Athabasca oil sands region. Similarly, Thomas and Gale (2015) reported improved growth and establishment of vegetation and increased tree biomass in soils amended with biochar.

While most published research has addressed the use of organic amendments in the revegetation of forest ecosystems following disturbances (Hahn and Quideau 2013; Dhar et al. 2019, 2020b), there has been little attention to the effect of suboptimal topsoil alone or amended with organic amendments (peat and biochar) on the recovery of reclaimed well sites. Therefore, the objective of this research was to determine the effects of TRD and organic amendments on early vegetation establishment and plant community development. We hypothesized that vegetation attributes will vary with TRD and amendment type. Results from the research will provide guidelines on the use of suboptimal TRD in conjunction with organic amendments (peat and biochar) to augment revegetation of disturbed boreal forest sites.

## 3.3 Methods and Materials

### 3.3.1 Study site

The study was located at a borrow pit near Cold Lake in northeastern Alberta (54°36′22.26″ N, 110°29′28.24″ W). The area is characterized by Luvisolic soils, glacial till material with loam (sandy loam and sandy clay loam), surface textured soils, and subsurface texture with medium-sized gravel. The soil structure varies from fine-grained blocky to somewhat massive. Additionally, the area is characterized by irregular drainage to well-drained patterns and slope classifications ranging from nearly level to very gentle slopes. The region is located within the moist rich d ecosite of the Central Mixedwood Region (moist-rich species). The dominant tree species in the area are trembling aspen (*Populus tremuloides* Michx.), white spruce (*Picea glauca*), and basalm poplar (*Populus balsamifera* L.), as well as woody shrub species such as green alder (*Alus veridis*), low bush cranberry (*Viburnum edule*), red-osier dogwood (*Cornus sericea*), and prickly rose (*Rosa acicularis* Lindl) (Natural Regions Committee, 2006). Dominant native graminoid species in the area include bluejoint (*Calamagrostis canadensis*) and native herbaceous forbs, common fireweed (*Chamerion angustifolium)*, wild strawberry (*Fragaria virginiana*), and common horsetail (*Equisetum arvense*).

### 3.3.2 Experiment layout and treatments

The experiment was laid out in a randomized complete block design with three blocks as replicates (Fig S1). Each block was divided into four plots, each measuring 20 m × 20 m with 2 m buffers. The treatments were the optimal topsoil depth (TRD80) as the control, the suboptimal topsoil depth 50% TRD (TRD50), and the 50% TRD amended with peat (PTRD50) and biochar (BTRD50), respectively. The control had a topsoil thickness layer of 22 cm, while the 50% TRD had a topsoil thickness layer of 14 cm. These were calculated from the original pre-disturbance topsoil depth, which was 28 cm (Solstice, 2010).

### 3.3.3 Amendment properties

Peat and biochar were applied at the beginning of the experiment at organic C rates equivalent to those in the TRD80 treatment (control). The peat had been stockpiled for one year at a nearby well pad at the site prior to placement into the plots. Biochar was procured from Lorris Laboratories Ltd., Calgary, Alberta, who prepared it from raw pine tree logs using slow pyrolysis in a kiln at temperatures between 550 °C and 650 °C for 6 to 12 h. The peat and biochar amendments were mixed with topsoil and uniformly spread in all the plots using a tractor and rototiller. Peat and biochar were applied at calculated rates of 20 kg $m^{-2}$ and 4.75 kg $m^{-2}$, respectively, to bring the TOC mass $m^{-2}$ in the TRD50 up to the level comparable to that in the TRD80. Amendments were incorporated in all plots in May 2015. Baseline chemical characteristics of the amendments were determined at the start of the experiment prior to application in the plots were described in chapter 2. (Table 2.2).

### 3.3.4 Tree and shrub seedling planting

Following plot preparation and treatment application, approximately 3,125 stems $ha^{-1}$, equivalent to 125 trees and shrub seedlings, were planted in each plot in May 2015. The seedlings (8 to 12 mo old) were obtained from Boreal Horticulture Ltd., Bonnyville, Alberta. Tree seedlings [balsam poplar (*Populus balsamifera* L), trembling aspen (*Populus tremuloides* Michx), white spruce (*Picea glauca*), and white birch (*Betula papyrifera* Marsh.)] were individually planted and uniformly distributed in each plot while shrub seedlings [saskatoon (*Amelachier alnifonia*), green alder (*Alus veridis*), red osier dogwood (*Cornus sericea*)] were planted in clusters in accordance with standard reclamation practice (Table S1). Additional tree and shrub seedlings were planted in buffer zones between the plots and along the perimeter of plot areas. The forest vegetation understory ingress community naturally grew in each plot. Stem mapping was used to map and tag the seedlings for identification purposes. Vegetation assessments were conducted using 10 m × 10 m quadrats.

### 3.3.5 Vegetation measurements

Canopy cover and vegetation health were assessed annually starting in August 2015. Plot centers were divided into equal sections equivalent to four 100 $m^2$ quadrats. Azimuth and distance from the plot center were used for stem mapping of the planted tree/shrub species while observing quadrat boundaries (Man and Yang, 2015). Stem mapping and vegetation assessments of planted species were conducted concurrently. Vegetation assessments recorded included tag number, height (cm), species type, and planting location (hummock, level, woody debris, hollow).

Following stem mapping and tree/shrub assessments, identification of all naturally-established species (Moss et al., 1983; Johnson et al., 1995; Bubar et al., 2000) in each quadrat was done in conjunction with visual assessment of canopy cover for each individual species. Cover class (<1 %, 1-2 %, 2-5 %, 5-10 %, 10-25 %, 25-50 %, 50-75 %, 75-95 %, and 95-100 %) was determined according to Braun-Blanquet (1965), with midpoints of the cover classes used to calculate the total canopy cover of all the species in each plot.

Total canopy cover was enumerated for different functional groups (graminoids, forbs, woody (trees and shrubs), native, and non-native species) and exceeded 100% for all except for woody species, due to overlap when cover estimates of species within a functional group were summed.

Species richness, Shannon Diversity Index (H′), species evenness (E), and percent survival were calculated using vegetation assessment data collected from the quadrats.

Species diversity in all plots was calculated using the Shannon Diversity Index (Kent and Coker 1992):

$$H' = -\sum_{i=1}^{s} p_i \ln p_i \qquad [1]$$

where H′ = species diversity, S = number of species, $\sum =$ the total sum of species (S), $p_i$ = proportion of individual species or abundance of the $i^{th}$ species expressed as a proportion of total cover, and ln = natural log.

Species evenness (E) was calculated as

$$E = \frac{H'}{\ln S} \qquad [2]$$

$$\text{Percent survival} = \frac{\text{number of survived trees}}{\text{total number of trees planted}} \times 100 \qquad [3]$$

### 3.3.6 Statistical analysis

Data were analyzed using the generalized linear mixed model procedure (PROC GLIMMIX) of SAS version 9.4 (SAS Institute, 2013) to determine treatment effects on vegetation (woody, graminoid, forb, native and non-native) canopy cover, woody plant height, species richness, H′, evenness, and percent survival. Treatment was modeled as a fixed effect and block as a random effect, while year was a repeated measures factor. For tree and shrub survival, species was also modeled as a fixed effect. The first-order autoregressive [AR1] covariance structure was selected as the most suitable for repeated measures analysis of vegetation attributes data, based on the corrected Akaike information criterion (Littell et al., 1996). Data for vegetation cover except woody cover, H′, and seedling height were normally distributed, whereas woody species cover, and evenness followed a beta distribution percent survival followed a binomial distribution, and species richness data followed the Poisson distribution. Analysis of covariance (ANCOVA) was used to determine treatment effects on the seedling height of each species, with initial height as a covariate. Means were compared using the Tukey multiple comparison procedure at $\alpha = 0.05$. When year main and interaction effects on species richness, woody plant height, and percent survival were significant, orthogonal polynomial contrasts were tested, and appropriate regressions were fitted and compared where applicable.

## 3.4 Results and Discussion

### 3.4.1 Weather conditions

Annual total precipitation was 423 mm in 2015, 462 mm in 2016, 495 mm in 2017, 501 mm in 2018 and 403 mm in 2019. The 30-year (1981–2010) average annual precipitation is 421 mm of which 319 mm is rainfall (Environment Canada, 2020). Monthly total precipitation was highest between June and July in all years (Fig. 3.1). The highest monthly mean temperatures ranging from 16.9 to 18.7 °C were recorded in July, while the lowest monthly mean temperatures ranging between -11.7 and -22.6 °C were recorded in January in all years (Fig. 3.1) (Environment Canada, 2020). In most months, the monthly mean temperatures were slightly warmer than normal in 2015, 2016, and 2019, while slightly cooler than the normal in 2018 (Fig. 3.1).

### 3.4.2 Canopy cover

Total cover differed significantly among treatments ($P = 0.04$) and years ($P < 0.0001$) (Table 3.1). Total cover was significantly greater for the TRD80 (mean, 154%) than the BTRD50 (132%) treatment but did not differ significantly between the TRD50 and the PTRD50 treatments. This could indicate the significance of a thicker topsoil layer as a medium with high moisture retention and nutrient availability for plant growth (Yang et al., 2019). Additionally, high nutrient availability and moisture retention properties of peat in the 50% TRD promoted total vegetation cover to a greater extent than biochar. Petelina et al. (2014) concluded that peat outperformed biochar in terms of vegetation cover during field trials evaluating amendments for land reclamation in Lake Athabasca, Alberta.

Although canopy cover of forbs, graminoids, and woody plants did not differ significantly among treatments, it varied significantly with time elapsed (year) since the start of the experiment ($P < 0.0001$ for the three variables) (Table 3.1). The canopy cover averaged across treatments for total vegetation, forbs, non-natives, and bare ground declined over time, whereas woody cover, native and graminoid cover increased (Table 3.1). Averaged across treatments, the total cover average was significantly greater in 2016 (mean 199%) than in 2015 (mean 51.1%) and 2019 (mean 138%). Similarly, forb cover (mean 129%) was significantly greater in 2016 relative to all other years across all treatments (Table 3.1). Thereafter forb cover decreased to 22.2% after 5 yr. of reclamation (Table 3.1). On the other hand, graminoid cover was significantly greater in 2019 (mean 87%) relative to the preceding years, and significantly increased by 82% between 2015 to 2019. The inverse relationship between forb and graminoid covers may reflect increased competition from graminoids, which may have germinated from the stockpiled soil in the final years of reclamation.

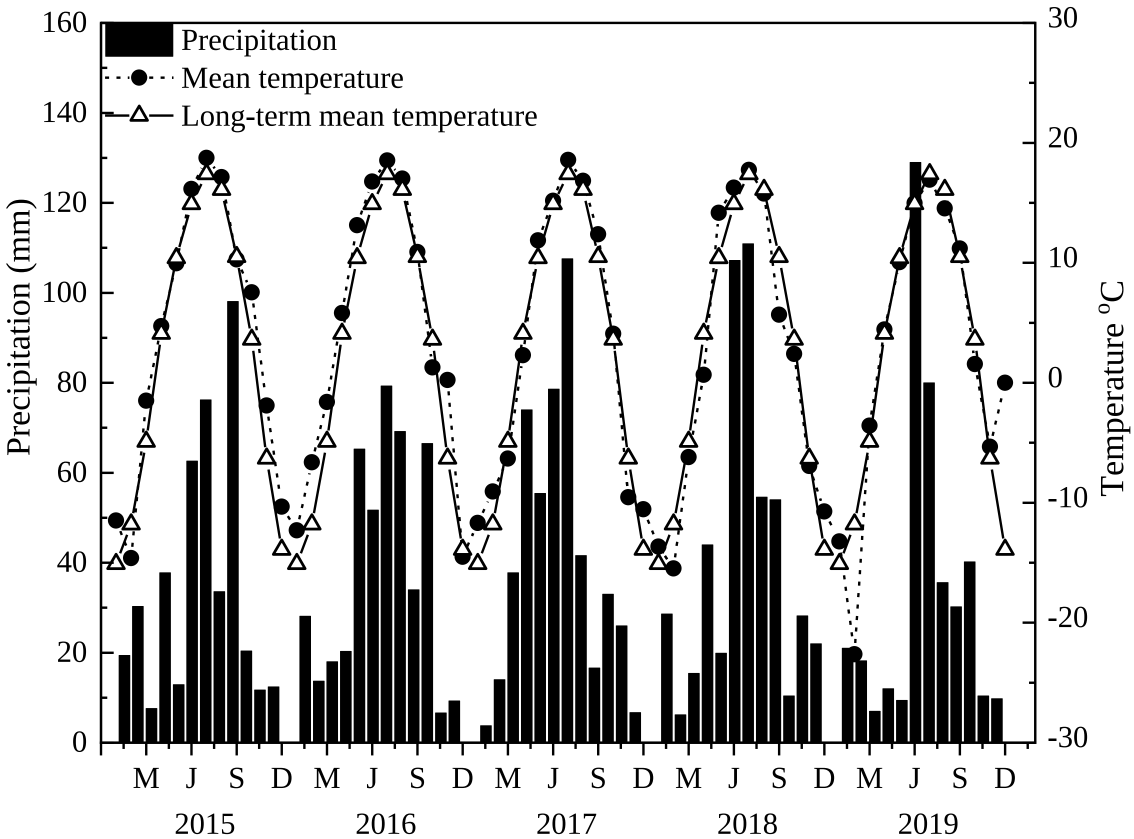


Figure 3. 1 Monthly cumulative precipitation and mean temperature during 2015-2019 at the borrow pit near Cold Lake, Alberta, Canada.

Dhar et al. (2019) reported that using stockpiled soil for reclamation had approx. four times graminoid cover relative to direct placement of cover soil. Thus, as graminoid abundance increases, competition for resources with other plants, including forbs, intensifies. Additionally, in the first few years following reclamation, newly disturbed boreal forests are dominated by annual understory forbs, which gradually decline as perennial species establish (Pinno and Hawkes, 2015; Dhar et al., 2020a; 2020b).

Thus, our findings indicate that the decrease in forb cover and increase in graminoid cover could be explained by the fact that the majority of dominant forbs in our study were annuals that declined after five years, whereas perennial graminoids (*Phalaris arundinacea, Agrostis scabra and Calamagrostis canadensis*) increased and dominated at the end the study. Despite the fact that graminoids are highly competitive for nutrients, moisture, and space which suppresses the growth of desirable woody plants following reclamation, their canopy and roots help reduce erosion and stabilize exposed soil (Naeth and Wilkinson, 2004).

Higher forb and total cover in 2016 and 2018 may have been a result of increased precipitation and warm temperatures, which likely stimulated microbial activity and enhanced availability of soil nutrients such as N, P, and S, which promoted greater vegetation development. (Hart and Chen, 2006; Dhar et al., 2019). The elevated nutrient concentrations in 2016 may also have been due to the increased transformations expected in the year following topsoil placement and amendment application, which may have resulted in the germination and rapid establishment of annual species from stockpiled seedbanks and seed dispersal from adjacent land (Carpenter and Fernandez, 2000). Carpenter and Fernandez (2000) found that topsoil manufactured with pulp sludge increased soil mineralization, P availability, CEC, pH, and cumulative grass yields relative to the unamended (control) topsoil 15 mo after topsoil placement.

Previous studies have demonstrated an increase in forb cover dominance shortly after disturbance during the early stages of reclamation, with trees and shrubs eventually becoming a significant component of the ecosystem as succession progresses (Hart and Chen, 2008; Pinno and Hawkes, 2015). Dominant forb species observed in our study were the competitive agronomic species alsike clover (*Trifolium hybridum*), red clover (*Trifolium repens*), and alfalfa (*Medicago sativa*).

Table 3. 1 Effects of topsoil replacement depth and organic amendments on forb, graminoid, woody, shrub, native and non-native cover over a 5-year period in northeastern Alberta.

| **Effect** | **Total cover** | **Woody cover** | **Forb cover** | **Graminoid cover** | **Native cover** | **Non-native cover** | **Bare ground cover** | **Woody debris** |
|---|---|---|---|---|---|---|---|---|
| | % | | | | | | | |
| Treatments | | | | | | | | |
| TRD80[a] | 154a [b] | 9.5 | 57.9 | 66.3 | 93.1 | 54.9 | 5.9 | 13.4 |
| TRD50 | 141ab | 8.9 | 62.7 | 49.6 | 73 | 62.3 | 3.1 | 14.5 |
| PTRD50 | 147ab | 10.6 | 60.2 | 60.9 | 94.1 | 51.9 | 6.4 | 13.2 |
| BTRD50 | 132b | 7.5 | 61.1 | 41.9 | 63.4 | 63.8 | 2.6 | 12.2 |
| Year | | | | | | | | |
| 2015 | 51.5c | 3.8e | 40.5c | 5.4d | 16.3 | 36.2 | 56.6a | 20.1 |
| 2016 | 199a | 5.9d | 129a | 50.3c | 97.8 | 90.8 | 2.2b | -[d] |
| 2017 | 164ab | 9.2c | 80.3b | 69.4b | 94.3 | 66.4 | 0.8b | 17.5 |
| 2018 | 165ab | 13.4b | 87b | 60.3b | 80.3 | 82.6 | 1.5b | 12.7 |
| 2019 | 138b | 20.3a | 22.2d | 87.8a | 116 | 15.1 | -[c] | 9.4 |
| | P value | | | | | | | |
| Treatment | **0.04** | 0.18 | 0.82 | 0.15 | < **0.0001** | 0.47 | 0.19 | 0.99 |
| Year | < **0.0001** | < **0.0001** | < **0.0001** | < **0.0001** | < **0.0001** | < **0.0001** | < **0.0001** | 0.78 |
| Treatment × Year | 0.38 | 0.67 | 0.17 | 0.59 | **0.01** | **0.003** | 0.74 | 1.00 |

[a] TRD80, 80% topsoil replacement depth; TRD50, 50% topsoil replacement depth; PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar.

[b] Means within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Tukey's multiple comparison procedure.

[c] Bare ground fully covered by canopy cover.

[d] Missing values for coarse woody debris.

Although the dominant graminoids were native species and characteristic species (Table S2), including reed canary grass (*Phalaris arundinacea*) and blue joint (*Calamagrostis canadensis*), they may be extremely competitive and therefore inhibit the establishment and growth of desirable native and woody species (Landhäusser et al., 1996; Hart and Chen, 2006; Dhar et al., 2019). While in our study TRD did not significantly affect forb and graminoid cover, Bowen et al. (2005) reported an increase in graminoid cover with increasing TRD whereas forb cover decreased with increasing TRD at reclaimed old mines in Wyoming, USA. This indicates that forb seedbanks require shallow depth for successful germination, whereas graminoids can survive greater burial depths (MacKenzie et al., 2012). Woody canopy cover varied significantly ($P < 0.0001$) with year but was not significantly affected by treatment (Table 3.1). In 2015, woody canopy cover averaged across all treatments was significantly lower than in the subsequent years, which significantly increased to reach 20.3% at the end of the study (Table 3.1). Although woody cover varied significantly in all the years, the lowest and the highest woody cover was observed in 2015 (3.8%) and 2019 (20.3%), respectively.

Canopy covers of tree and shrub species were less than those of forbs and graminoids. The low woody cover observed during the early years of this study could be a result of increased competition for light, moisture, and nutrients from dominant forb and graminoid species (Maron & Marler, 2008; Hart and Chen, 2008; Dhar et al., 2020b). However, consistent with previous studies (Messier et al. 1998; Dhar et al., 2020a, 2020b), as tree and shrub species matured and their canopies expanded over time, they shaded out understory forbs and graminoid species, decreasing their growth and promoting the establishment of desirable characteristic species. Other studies have demonstrated a similar pattern of woody cover progression ( Pinno and Hawkes 2015; Dhar et al., 2019, 2020a). Pinno & Hawkes (2015) reported an increase in forb (~60%) and graminoid

(~35%) canopy cover during the first five years after oil sands mine reclamation, followed by a decline 20 yr following reclamation. On the other hand, shrub cover increased significantly over time but remained below 20%, 20 yr following boreal forest reclamation in Fort McMurray, Alberta. Similarly, five years following boreal reclamation, Dhar et al. (2020a) observed a high forb canopy cover (~60%) and low tree canopy cover (less than 20%). However, sixteen years later, the authors reported a significant double reduction in forb canopy cover and a significant four-fold increase in tree canopy cover (80%). The indicator species, *Rubus idaeus*, observed in our study may signify reclamation success and a trajectory towards the natural boreal forest ecosystem (Hart and Chen, 2006; Dhar et al., 2020a, 2020b).

There was a significant treatment × year interaction for native ($P = 0.01$) and non-native species cover ($P = 0.003$) (Table 3.1). Native species canopy cover was significantly lower for all treatments in 2015 but was higher in subsequent years (Fig. 3.2a). In 2017, the TRD80 treatment had significantly higher native species cover than the BTRD50 and TRD50 treatments whereas in 2019, there was no significant difference in native species canopy cover among the three treatments (Fig. 3.2a). Notably, in 2019, native species canopy cover increased by approximately 100% for the TRD50 and 45% for the BTRD50 treatment relative to 2018. Additionally, when comparing the organic-amended plots, the PTRD50 treatment produced a significantly greater native species canopy cover than the BTRD50 treatment in 2018, but there were no significant differences between the treatments in 2019. This suggests that peat improves native canopy cover and plant community development to a greater extent than biochar during the early stages of vegetation establishment. The slow response of native canopy cover to biochar may be due to its low nutrient content during the early establishment of plants. Previous research demonstrated that, due to the high C/N ratio (945) of biochar, immobilization of plant-available nutrients is likely to occur during early plant establishment stages (Chan and Xu, 2009). However, its positive effect may become evident in later years, as indicated by increased native cover in year 5 of our study. However, additional research is needed to investigate whether the immobilization and mineralization of biochar is a short-term, mid- or long-term occurrence in forest ecosystem restoration studies (McElligott, 2011).

Non-native species canopy cover was significantly greater in 2016 and 2018 than in 2015 and 2019 for all treatments (Fig. 3.2b). In 2015, all treatments had significantly lower non-native canopy cover (less than 40%) than in 2016, but in 2017, the BTRD50 treatment had significantly higher non-native canopy cover than the PTRD50 treatment (Fig. 3.2b). Additionally, the BTRD50 treatment had a significantly higher non-native canopy cover than TRD80 in 2018, but this difference was not significant in 2019. Notably, non-native species cover declined significantly (to < 20%) in 2019 relative to the preceding three years, indicating a decrease in their dominance over time. Thus, during the early stages following reclamation of disturbed boreal sites with sparse vegetation cover and abundant bare ground, non-native species may be dominant, reflecting their highly competitive nature at exploiting available resources immediately following reclamation (Pinno and Hawkes 2015; Dhar et al. 2020a, 2020b). Importantly, non-native species can impair the growth of native species, altering their successional trajectory and reducing the diversity of desirable native species (MacDougall and Turkington, 2005). Nonetheless, because the majority of non-native species are annuals, they may be eliminated over time as perennial native species develop (Dhar et al., 2019, 2020a; 2020b).

The 20% decrease in non-native species cover in year 5 of our study corroborates findings by Dhar et al. (2019), who observed a decline in non-native species cover from 32% to 10% within 16-24 yr following reclamation. Similarly, Pinno and Hawkes (2015) reported a mean non-native species canopy cover of 10% 20 yr after reclamation in the oil sands region of Alberta.

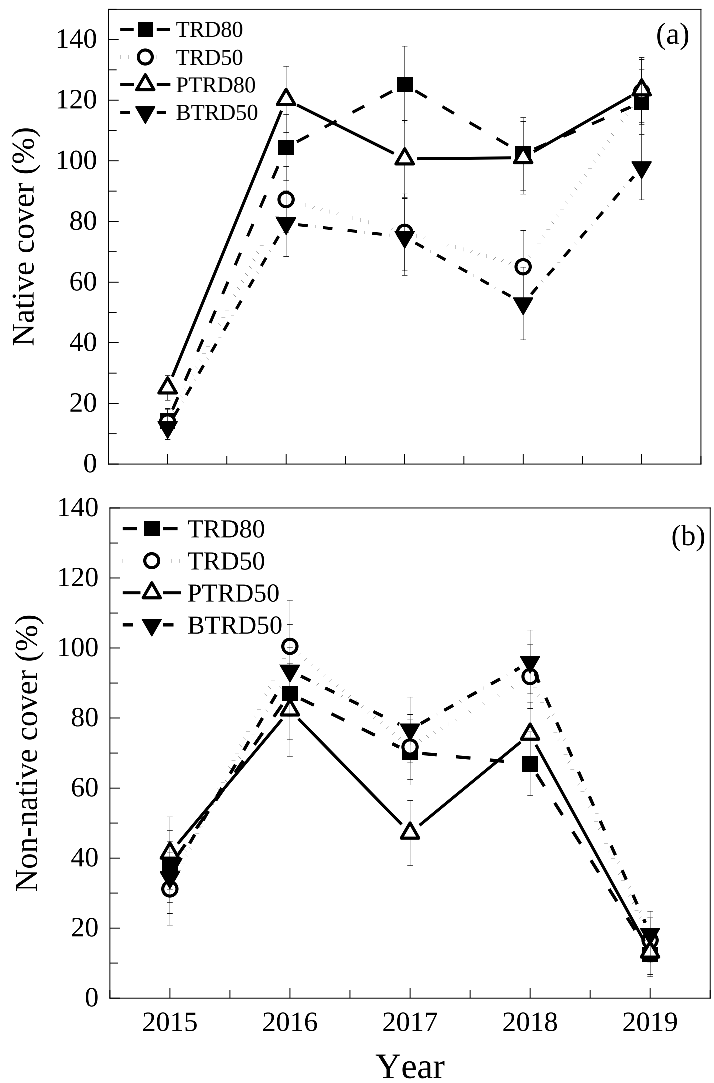


Figure 3. 2 Treatment by year interaction effect on (a) native canopy cover and (b) non-native canopy cover following reclamation of a borrow pit in northeastern Alberta. Vertical bars represent standard errors of the mean.

While non-native species are undesirable, their early establishment provides the cover needed to reduce soil erosion, enhance organic matter accumulation, sustain nutrient cycling and support the development of soil structure (Macdougall and Wilson 2011; Dhar et al., 2019). The majority of non-native species in our study were forbs dominated by *Melilotus alba*, *Trifolium hybridum*, and *Trifolium pratense*, with a few graminoids, which included *Bromus inermis*, *Elymus repens*, and *Phleum pratense*. Sylvain et al. (2019) also observed a high proportion of forbs among non-native or ruderal species at reclaimed rangeland sites in the Great Plains of northern USA.

Bare ground and coarse woody debris cover did not vary significantly with treatment. However, there was a significant change in bare ground over time ($P < 0.0001$) (Table 3.1). Bare ground decreased significantly after 2015 (56.6%) and averaged 1.5% during 2016 through 2019. The abundant bare ground during the first year was expected since canopy cover was not fully developed, and invasive and other native herbaceous plants were not fully established. However, consistent with previous research (Groninger, 2005), as canopy cover expanded, the bare ground gradually disappeared. Zvomuya et al. (2011) reported an inverse relationship between bare ground and lichen cover, which is consistent with observations in our study.

### 3.4.3 Species richness, diversity, and evenness

There was a significant treatment × year interaction ($P = 0.01$) for total species richness (Table 3.2). Orthogonal polynomial contrasts revealed a significant treatment × year linear interaction ($P = 0.0003$); however, Poisson regression analysis showed no significant difference in regression coefficients between treatments. Therefore, a common linear regression was fit for all treatments ($P = 0.0004$) and indicated a 3% increase in total species richness per year (Table S3).

While forb species richness did not differ significantly ($P = 0.19$) with treatment, woody species richness for the TRD50 treatment was significantly greater than that for the BTRD50 treatment but was not significantly different from those for the TRD80 and PTRD50 treatments (Table 3.2). Both forb ($P < 0.0001$) and woody ($P < 0.0001$) species richness showed significant change over time. Forb species averaged across treatments, were significantly more abundant during early years (2015 and 2016) but declined significantly in subsequent years. In comparison, woody species richness increased by 30% in 2019 relative to 2015. Regression analysis showed a 9% per year decrease in forb species richness ($P < 0.0001$), whereas woody species richness increased by 6% per year (Table S3.). This inverse relationship between forbs and woody species richness in plant community development following reclamation has been demonstrated in previous studies (Rowland et al., 2009; Dhar et al., 2020a; 2020b). Thus, during the early years of reclamation, understory forb species typically predominate over slow-growing woody species, taking advantage of nutrients and water at newly disturbed sites (Hart and Chen 2006; Dhar et al. 2020a, 2020b). However, trees and shrubs eventually outgrow and shade the forb species, thus out-competing them (Lieffers et al., 1999), thereby limiting competition and enhancing recolonization by desirable perennial forb and woody species (Zhang et al. 2017; Dhar et al. 2020b). This pattern is consistent with that of early successional community development in a typical natural boreal forest ecosystem (Landhäusser et al. 1996; Hart and Chen 2008; Dhar et al., 2020a).

There was a significant treatment × year interaction ($P = 0.03$) for graminoid species richness (Table3). While orthogonal polynomial contrast analysis revealed a significant treatment × year linear interaction ($P = 0.01$), further analysis using Poisson regression indicated a linear temporal trend that did not vary significantly among treatments. A common regression was therefore fitted and showed that graminoid richness increased by 21% per year (Table S3).

Table 3. 2 Species richness, species diversity index, and evenness as affected by topsoil replacement depth and organic amendment at a reclaimed borrow site in northeastern Alberta.

| **Effect** | **Total richness** | **Forb richness** | **Woody richness** | **Graminoid richness** | **Native richness** | **Non-native richness** | **H′ [a]** | **E** |
|---|---|---|---|---|---|---|---|---|
| Treatment | | | | | | | | |
| TRD80 | 44 | 22 | 9.6ab [b] | 11 | 32 | 10 | 2.6 | 0.79 |
| TRD50 | 45 | 22 | 9.8a | 11 | 32 | 11 | 2.7 | 0.81 |
| PTRD50 | 45 | 24 | 9.6ab | 10 | 33 | 10 | 2.6 | 0.76 |
| BTRD50 | 43 | 22 | 8.7b | 9 | 29 | 11 | 2.7 | 0.84 |
| Year | | | | | | | | |
| 2015 | 41 | 27a | 7.7c | 5 | 23 | 17 | 2.9a | 0.93a |
| 2016 | 47 | 26a | 9.8ab | 10 | 33 | 13 | 2.3b | 0.66b |
| 2017 | 44 | 21b | 10.1a | 12 | 32 | 10 | 2.6ab | 0.75ab |
| 2018 | 39 | 18c | 8.8b | 11 | 29 | 9 | 2.8a | 0.84ab |
| 2019 | 52 | 22b | 11a | 15 | 41 | 7 | 2.6ab | 0.71b |
| | | | | P-value | | | | |
| Treatment | 0.57 | 0.19 | **0.03** | 0.13 | **0.03** | 0.18 | 0.80 | 0.88 |
| Year | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **0.0004** | **0.01** |
| Linear (yearlin) | **0.0004** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | 0.63 | 0.06 |
| Quad (yearquad | **0.01** | **<0.0001** | 0.20 | **0.0002** | 0.34 | 0.44 | **0.01** | 0.12 |
| Treatment × Year | **0.01** | 0.08 | 0.66 | **0.03** | **0.03** | **0.03** | 0.52 | 0.98 |
| Treatment × yearlin | **0.0003** | 0.20 | 0.56 | **0.01** | **0.03** | **0.002** | 0.33 | 0.76 |
| Treatment × yearquad | 0.86 | 0.64 | 0.67 | 0.28 | 0.78 | 0.20 | 0.38 | 0.86 |

[a] H′, Shannon Diversity Index; E, evenness index; TRD80, 80% topsoil replacement depth; TRD50, 50% topsoil replacement depth; PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar.

[b] Means within a column followed by different letters are significantly different at $\alpha = 0.05$ according to Tukey's multiple comparison procedure.

While graminoid richness is usually dominant during early years of reclaimed boreal sites, followed by a declining trend as succession progresses (Rowland et al. 2009; Pinno and Hawkes 2015; Dhar et al. 2020a), graminoid richness for all treatments in our study showed an increasing trend, reaching 15 graminoid species five years following reclamation. We suspect that these graminoids were established out of the seed bank of the stockpiled topsoil used in this study, and later became more competitive for space as forb species richness decreased. Based on previous research, stockpiling of cover soil may increase graminoid species abundance after reclamation of boreal sites as some of the graminoids can maintain their viability at greater depths during stockpiling (MacKenzie et al., 2012; Dhar et al., 2019). Although reclaimed sites have been reported to have a high proportion of graminoids than undisturbed natural forests (Rowland et al., 2009; Rapai et al., 2021), we anticipate that these graminoids will decline over time (Dhar et al., 2020a). However, there is some evidence that graminoids may persist in reclaimed sites up to 14 yr following reclamation (Norman et al., 2006) and 16-24 yr (Dhar et al., 2020a).

Treatment effects on native species richness varied significantly with year as indicated by a significant ($P = 0.03$) treatment × year interaction (Table 3.2). Although orthogonal polynomial contrast analysis showed a significant treatment × year linear interaction for native species richness ($P = 0.03$), Poisson regression analysis indicated a temporary linear trend that did not differ significantly among treatments. Averaged across treatments, native species richness increased by 10% per year ($P < 0.0001$) over the 5-yr study period (Table S3).

There was a significant treatment × year interaction ($P = 0.03$) for non-native species richness (Table 3.2). In contrast to native species richness, all treatments had a significantly greater non-native richness in the first year (2015) than in Year 5 (2019), indicating a significant declining trend over time (Table 3.2). Orthogonal polynomial contrast analysis revealed a significant treatment × year linear interaction ($P = 0.002$) (Table 3.2). However, regression analysis showed that, across all treatments, non-native species richness decreased by 19% per year. Previous studies have demonstrated that non-native species occupy a greater amount of space during the early post-reclamation years due to seed dispersal, resulting in competition for space with native species. Non-native species have been shown to decline over time as perennial native species develop (MacDougall and Turkington, 2005; Pinno and Hawkes, 2015; Dhar et al., 2020a). This was evident in our study, where native species richness increased over time in all plots while non-native species richness decreased with time. The decrease in non-native species and dominance of native species 5 yr after reclamation translates to a dissipation in competition of non-native species for available resources as desirable perennial native species become more established (Rowland et al. 2009; Pinno and Hawkes 2015; Dhar et al. 2020a). Similarly, Dhar et al. (2020a) reported a decrease in non-native species 16 yr after reclamation of a disturbed boreal forest site in the oil sands region of Alberta. While non-native species may negatively impact species diversity and delay the successional progression of ecosystem recovery, in some cases, they can prevent soil erosion and contribute to soil formation processes (Dhar et al., 2018; Rapai et al., 2021). Although non-native species were present in our study, they were significantly less abundant than perennial native species, indicating that the trajectory for all treatments was towards the desired plant community development of a typical natural boreal forest moving from invasive or non-native and annual species to perennial species (Hart and Chen, 2006; Dhar et al., 2019).

Species evenness and H′ did not differ significantly among treatments (Table 3.2). However, both indices were significantly higher in 2015 (2.9 for H′ and 0.93 for evenness) than in 2016 (2.3 and 0.66, respectively). Although orthogonal polynomial contrasts revealed a significant ($P = 0.01$) quadratic trend (Table 3.2), regression analysis showed no significant temporal change in H′ across all treatments. Five years after reclamation, $H'$ decreased to 2.6 from 2.9 in 2015. This corroborates the findings of Corns and Roi (1979), who reported high species diversity during the early years following reclamation, followed by subsequent declines as overstory canopy cover increased. Similarly, Dhar et al. (2020a) observed a slight decrease in species richness and diversity 16 yr after reclamation. Overall, $H'$ was high and within the range ($1.5 < H' < 2.5$) considered normal (Kent and Cooker, 1992). Diversity indices of 1.5 or less indicate low diversity whereas those of 2.5 or greater indicate high diversity (Kent and Cooker, 1992). Similarly, E in our study (0.66 to 0.93) was high (Dhar et al., 2019).

Species diversity is considered one of the drivers for ecological restoration in reclaimed boreal sites where it plays a critical role in the resilience and productivity of healthy forest ecosystems (Tilman, 1994; Drever et al., 2006). Additionally, reclaimed sites with high species diversity are regarded as stable and capable of rapid recovery following disturbance (Hooper et al., 2005; Dhar et al., 2018). Thus, the higher species diversity and presence of characteristic species (Table S2) in our study suggest that the reclaimed site may recover to a stable and resilient plant community (Dhar et al., 2018).

Our findings indicate that reclamation using a 50% TRD negatively impacts revegetation relative to the 80% TRD required by Alberta's regulatory mandates. Reclamation and ecosystem recovery of disturbed sites are often difficult with a higher proportion of desirable native species because reclaimed sites are characterized by a significant proportion of competitive undesirable graminoids and non-native species during plant community development.

### 3.4.4 Tree and shrub height

There was a significant treatment × year interaction effect on the seedling heights of aspen ($P < 0.0001$), birch ($P = 0.01$), poplar ($P = 0.02$) and spruce heights ($P = 0.001$), indicating that the response of heights to years differed among treatments (Table 3.3). Orthogonal polynomial contrasts revealed significant quadratic temporal trends in seedling height that varied among treatments with years as indicated by quadratic × treatment interaction for aspen heights ($P < 0.0001$), birch ($P = 0.01$), poplar ($P = 0.04$), and spruce ($P = 0.004$) (Table 3.3). Regression analysis showed that aspen seedling height for the TRD80 and PTRD50 treatments was described by a common regression line and increased as a quadratic function of years after reclamation (aspen heights = 37.1 – 6.85 × year + 5.64 × years$^2$ , $r^2 = 0.93$, $P < 0.0001$) but, increased linearly with years for BTRD50 (aspen heights = 13.2 + 12.5 × years , $r^2 = 0.82$, $P < 0.0001$) and TRD50 (aspen height = 16.7 – 6.56 × year, $r^2 = 0.74$, $P < 0.0001$) (Fig 3.3a). Thus, while aspen heights increased from 10.1 cm in 2015 to 49.5 cm in 2019 in TRD50 plots and from 25.7 to 75.7 cm in BTRD50 plots, it increased from 35.9 to 144 cm in the TRD80 and PTRD50 plots at a slower rate in the TRD80 and PTRD50 plots during the first 2 yr. as indicated by a lag phase, but significantly increased over time to reaching a maximum height of 169 cm at year 5 (Fig. 3.3a). While aspen heights increased by 12 cm per year in the BTRD50 plots, nearly twice the rate at which aspen heights increased in the TRD50 plots, which increased by 6.56 cm per year (Fig. 3.3a).

The temporal increase in birch height was described by a common regression for the TRD80 and PTRD50 treatments and increased as a quadratic function of year (height = 43.3 – 15.7 × year + 7.69 × year$^2$, $r^2 = 0.96$, $P < 0.0001$), whereas it increased linearly for TRD50 and BTRD50 (common regression: height = 12.8 + 8.15 × years; $r^2 = 0.80$, $P < 0.0001$) (Fig. 3.3b). Similarly, spruce height increased quadratically with time for the TRD80 and PTRD50 treatments (common regression: height = 31.3 – 3.79 × year + 4.34 × year$^2$, $r^2 = 0.91$, $P < 0.0001$), whereas it increased linearly for TRD50 and BTRD50 (common regression: height = 18.5 + 5.85 × year; $r^2 = 0.53$, $P < 0.0001$) (Fig. 3.3c). The increase in poplar seedling height was described by two quadratic functions: one for the TRD80 and PTRD50 treatments (height = 38.6 – 9.47 × years + 6.87 × years$^2$, $r^2 = 0.87$, $P < 0.002$) and the other for the BTRD50 treatment (height = 18.3 – 4.13 × years + 2.59 × years$^2$, $r^2 = 0.96$, $P < 0.01$), whereas there was no significant trend in seedling height for the TRD50 treatment. Poplar tree is heights for the TRD50 treatment were significantly lower than those for the other three treatments in Years 3 (2017) through 5 (2019) (Fig. 3.3d). After 5 yr, poplar heights reached the maximum height of (213 cm) while spruce had the shortest height of (132 cm) (Fig. 3.3c; 3.3d), indicating the relatively fast growth rate of poplar and the slow growth rate of spruce.

There was a significant treatment effect on dogwood ($P = 0.01$) and saskatoon ($P = 0.003$) but not on alder ($P = 0.45$) shrub height (Table 3.3). Averaged across years, dogwood height was significantly lower for the BTRD50 treatment (mean = 44.5 cm) than the other three treatments (mean = 52.9 cm for TRD80, TRD50, and PTRD50). Similarly, saskatoon height (mean of the 5 yr) was significantly lower for BTRD50 (24.4 cm) than for TRD80 and PTRD50 (mean = 32.5 for the two treatments) but did not differ significantly between BTRD50 and TRD50 (26.2 cm).

There was also a significant year effect on the height of all three shrub species (Table 3.3). Orthogonal polynomial analysis and subsequent regression analysis revealed that, averaged across all treatments, shrub height increased as a linear function of year for alder (height = 2.19 + 0.47 × year, $r^2 = 0.72$, $P < 0.0001$) and saskatoon (height = 21.1 + 2.57 × year, $r^2 = 0.20$, $P < 0.001$) and as a quadratic function of year for dogwood (height = 14.7 + 17.7 × year – 1.52 × year$^2$, $r^2 = 0.81$, $P = 0.03$) (Fig. S1).

The greater tree heights for the TRD80 treatment relative to the TRD50 indicates the importance of sufficient topsoil replacement, which is needed to provide adequate moisture and nutrients for tree growth (Dietrich and MacKenzie 2018). Notably, our results show that, in the absence of sufficient topsoil, peat application can improve tree growth to levels similar to those for the mandatory TRD80. The superior performance of peat relative to biochar may also be related to greater moisture retention (saturation) properties (50%) of peat-amended soils relative to biochar-amended soils observed in this and other research (Petelina et al., 2014), which likely resulted in faster tree growth for the PTRD50 than the BTRD80 treatment.

Nitrogen availability is among the critical drivers for tree growth and is typically limiting in boreal forest soils (Turkington et al., 1998); therefore, organic amendments with low C/N ratio, such as peat (C/N = 17.6), enhance nutrient availability, thereby promoting the early establishment of trees following reclamation. It has been suggested that biochar alone may result in the immobilization of some nutrients due to its recalcitrant and sorptive nature, which may negatively impact tree growth (Chan and Xu, 2009). Previous studies have demonstrated that the addition of fertilizers together with biochar enhances nutrient availability and therefore promotes the early establishment of trees during the restoration of forest ecosystems (Pinno and Errington, 2015; Tremblay et al., 2019; Hogberg et al., 2020).

Table 3. 3 Tree and shrub height response to topsoil replacement depth and organic amendments at a reclaimed borrow site in northeastern Alberta.

| Effect | Aspen | Birch | Poplar | Spruce | Alder | Dogwood | Saskatoon |
|---|---|---|---|---|---|---|---|
| | | | | cm | | | |
| Treatment [a] | | | | | | | |
| TRD80 | 82.6 | 89.1 | 76.1 | 68.9 | 36.9 | 54.5a[b] | 34.3a |
| TRD50 | 36.4 | 50.3 | 37.1 | 33.6 | 31.1 | 51.4a | 26.2ab |
| PTRD50 | 74.6 | 80.7 | 95.5 | 66.5 | 39.3 | 53.0a | 30.6a |
| BTRD50 | 50.6 | 59.2 | 59.2 | 35.4 | 41.4 | 44.5b | 24.4b |
| Year | | | | | | | |
| 2015 | 31.3 | 29.6 | 28.9 | 28.6 | 12.9d | 32.2d | 23b |
| 2016 | 39.5 | 40.7 | 42 | 36.4 | 26.6c | 40.1c | 27.4ab |
| 2017 | 53.7 | 54 | 58.9 | 44.7 | 38.7bc | 56.3b | 28.7a |
| 2018 | 76.7 | 78 | 84.7 | 65.1 | 62.5ab | 61.4ab | 30.4a |
| 2019 | 104 | 107 | 120 | 81 | 85.2a | 64.2a | 34.8a |
| | | | | P value | | | |
| Treatment | **<0.0001** | **<0.0001** | **0.003** | **0.0001** | 0.45 | **0.01** | **0.003** |
| Year | **<0.0001** | **0.0007** | **<0.0001** | **<0.0001** | **0.002** | **<0.0001** | **0.001** |
| Linear (year lin) | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** | **<0.0001** |
| Quadratic (year quad) | **<0.0001** | **0.04** | **0.01** | **0.02** | 0.18 | **<0.0001** | 0.95 |
| Treatment × Year | **<0.0001** | **<0.0001** | **0.03** | **0.001** | 0.97 | 0.75 | 0.13 |
| Treatment × year lin | **<0.0001** | **<0.0001** | **0.001** | **<0.0001** | 0.87 | 0.59 | **0.01** |
| Treatment × year quad | **<0.0001** | **0.01** | **0.04** | **0.004** | 0.88 | 0.22 | 0.28 |

[a] TRD80, 80% topsoil replacement depth; TRD50, 50% topsoil replacement depth; PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar.

[b] Means within a column followed by the same letter are not significantly different at $\alpha = 0.05$ according to the Tukey multiple comparison procedure.

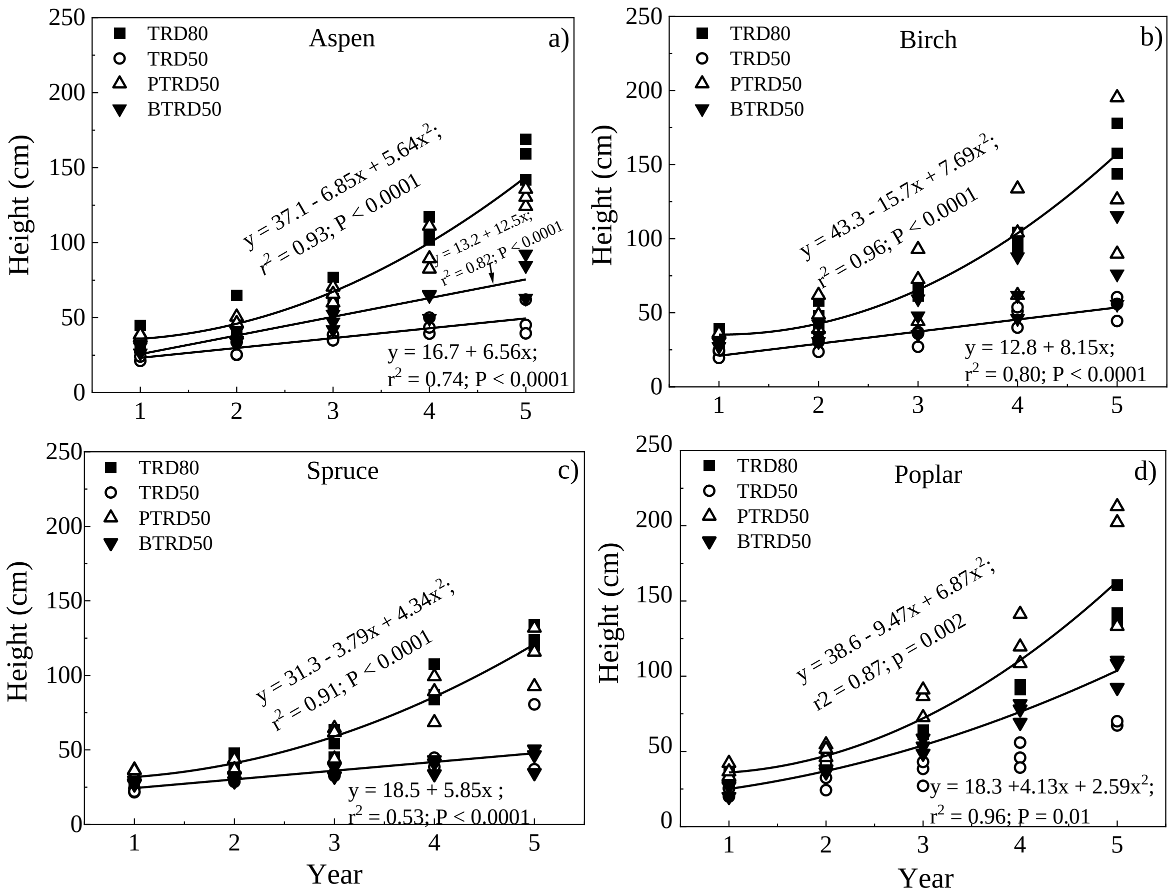


Figure 3. 3 Effects of treatment and year on the height of (a) aspen, (b) birch, c) spruce, and (d) poplar during reclamation of a borrow pit in northeastern Alberta. Year 1, 2015; Year 2, 2016; Year 3, 2017; Year 4, 2018; Year 5, 2019.

While a recent meta-analysis on the effect of different woody biochars on tree growth showed improved woody plant growth for all biochars relative to the unamended control, with the greatest responses occurring during the early growth stages (Thomas and Gale, 2015), our results showed poor tree growth response for the biochar treatment relative to the peat treatment. Thomas and Gale (2015) concluded that biochar may have a greater potential for forest restoration as it provides organic matter and enhances nutrient availability and moisture retention for the growth of woody plants. Additionally, Palviainen et al. (2020) reported tree heights increased by 12% for the biochar amendment applied at 5 Mg $ha^{-1}$ relative to the control during three years of their study in a clear cut boreal zone in southern Finland. In contrast, Bieser and Thomas (2019) revealed no significant difference in seedling height growth between poplar biochar and high-wood ash biochar amendments during their 3-yr boreal forest restoration study in northwestern Ontario. Biochar properties differ widely, depending on feedstock type. The pine wood-based biochar used in our study may not support as much vegetation growth as the wood biochars examined by Thomas and Gale (2015) and Palviainen et al. (2020). This underlies the need for future research to evaluate different biochars alone or in combination with other amendments for their effects on early plant establishment.

Dietrich and MacKenzie (2018) observed an increase in aspen seedling height for a peat mineral mix compared to a peat treatment amended with biochar. Although competition for resources and space from other functional groups such as forbs and graminoids typically has a negative effect on tree and shrub seedling growth (Pokharel and Chang, 2016; Dhar et al., 2019, 2020b), it is noteworthy that the shrub species alder, which was included in our study, is a nitrogen-fixer that can increase N availability and organic matter in the soil, thereby promoting the growth of other vegetation species, including trees (Balandier et al., 2006; Lefrançois et al., 2010; Pinno and Hawkes, 2015). Additionally, shrubs promote tree growth by retaining moisture or snow, thereby reducing seedling mortality and mitigating drought stress (Rowland et al., 2009).

The smaller heights observed for white spruce in our study were likely due to its determinate growth habit, which makes it a slow-growing tree species (Nienstaedt and Zasada, 1990); this is exacerbated during early establishment by competition for space and resources from other vegetation functional groups. Although understory herbaceous and graminoid species are important for plant community development and ecological diversity restoration, they have a detrimental effect on the growth of desirable planted woody species via shading and resource exploitation (Dhar et al., 2020a; 2020b).

### 3.4.5 Percent survival

There was a significant treatment × species interaction effect ($P = 0.02$) on tree survival (Table 3.4). Across all years, spruce maintained a high rate of survival for all treatments while birch showed a significant decrease in percent survival for the BTRD50 treatment relative to the TRD80, TRD50, and PTRD50 treatments (Fig. 3.4). Birch survival was significantly greater than poplar survival for the TRD80, TRD50, and PTRD50 treatments, but the survival of the two species did not differ significantly in the BTRD50 treatment.

In comparison to other tree species, aspen had the lowest survival (< 50%) across all treatments (Fig. 3.4). Aspen trees are shade-intolerant and negatively affected by low light conditions (Loach, 1970; Ung et al., 2001), which may also explain the low survival rate of aspen in our study. Additionally, the high mortality rate of aspen could be a result of high competition for resources from other vegetation functional groups, such as herbaceous and graminoid species (Franklin et al., 2012).

Tree survival differed significantly ($P < 0.0001$) with year (Table 3.4). Orthogonal polynomial contrasts showed a significant ($P < 0.0001$) year linear function for tree survival (Table 3.4). Regression analysis revealed a significant temporal linear decrease of 28% per year for all treatments. Thus, a significant decrease in tree survival with time is a typical pattern on disturbed sites. Tree survival is typically initially high and declines over time, approximately from 5 yr, depending on the species, before attaining a constant level as the woody canopy closes over the ruderal understory species (Skousen et al., 2009). Skousen et al. (2009) reported high survival rates for black cherry (82%) and red oak trees (96%), which declined to 36% and 47%, respectively, after 7 yr. The authors also noted that it took approximately 5-8 yr for the same species to reach a constant population post reclamation. Although we used different tree species from those in their study, we observed a similar pattern of declining tree species survival with time. Another factor affecting tree species survival is the presence of a dense herbaceous and graminoid cover (Torbert and Burger, 2000; Holl, 2002; Dhar et al., 2020a). These species occupy most of the available space on disturbed sites and compete with woody plants for nutrients and moisture (Pinno and Hawkes, 2015). Additionally, other invasive forbs and graminoids can obscure the planted trees and reduce their chances of survival (Torbert, 1995).

Shrub survival did not differ significantly among treatments ($P = 0.85$) and years ($P = 0.07$), but it varied significantly among species ($P < 0.001$). Survival of dogwood (98%) was significantly greater than that of saskatoon (77%) and alder (35%) (Table 3.4). Koropchak et al. (2013) reported high survival rates (> 80%) for gray dogwood and green hawthorn after one growing season during reclamation in West Virginia, USA. Similarly, Zipper et al. (2011) reported dogwood species survival rates of 86% nine years following reclamation in southwestern Virginia. These studies corroborate our findings of greater survival rates for dogwood across the 5 yr following reclamation, which indicates desirable shrub characteristics in degraded soils during early establishment, making it an ideal shrub for reclamation (Gucker, 2012). We observed low survival rates (< 50%) for green alder. In contrast, Pietrzykowski et al. (2018) reported green alder survival rates of 72% and 93% after 5 yr of restoration of fly ash disposal sites in central Poland. Although green alder is regarded as a moderately shade-tolerant species (Krajina et al., 1982), we speculate that the greater grass canopy cover observed in our plots may have partially negatively impacted the early establishment, growth, and survival of green alder.

Overall, the survival rates of aspen (24%) and alder (35%) were less than 50%, indicating a high mortality rate (Skousken et al., 2009). By comparison, spruce (91%), birch (86%), poplar (69%), dogwood (98%), and saskatoon (77%) had survival rates greater than 50%, which are considered high.

Table 3. 4 Topsoil replacement depth and organic amendment effects on tree and shrub species survival following reclamation of a borrow pit in northeastern Alberta.

| Effect | Tree survival | Shrub survival |
|---|---|---|
| | % | |
| Treatment [a] | | |
| TRD80 | 79 | 80 |
| TRD50 | 70 | 81 |
| PTRD50 | 69 | 77 |
| BTRD50 | 69 | 85 |
| Species | | |
| Poplar | 69 | - [c] |
| Aspen | 24 | - |
| Birch | 86 | - |
| Spruce | 91 | - |
| Alder | - [b] | 35c |
| Dogwood | - | 98a |
| Saskatoon | - | 77b |
| Year | | |
| 2015 | 88a | 89 |
| 2017 | 70b | 83 |
| 2018 | 68b | 79 |
| 2019 | 52c | 71 |
| | P value | |
| Treatment | 0.54 | 0.85 |
| Species | **<0.0001** | **<0.0001** |
| Year | **<0.0001** | 0.07 |
| Year (lin) | **<0.0001** | 0.01 |
| Year (quad) | 0.81 | 0.97 |
| Treatment × Species | **0.002** | 0.66 |
| Treatment × Year | 0.35 | 0.96 |
| Species × Year | 0.05 | 0.10 |
| Treatment × Species × Year | 0.79 | 0.98 |

[a] TRD80, 80% topsoil replacement depth; TRD50, 50% topsoil replacement depth; PTRD50, 50% topsoil replacement depth plus peat; BTRD50, 50% topsoil replacement depth plus biochar

[b] Means within a column followed by same letters are not significantly different at $\alpha = 0.05$ according to Tukey's multiple comparison procedure.

[c] Missing elements for non-tree species.

[d] Missing elements for non-shrub species.

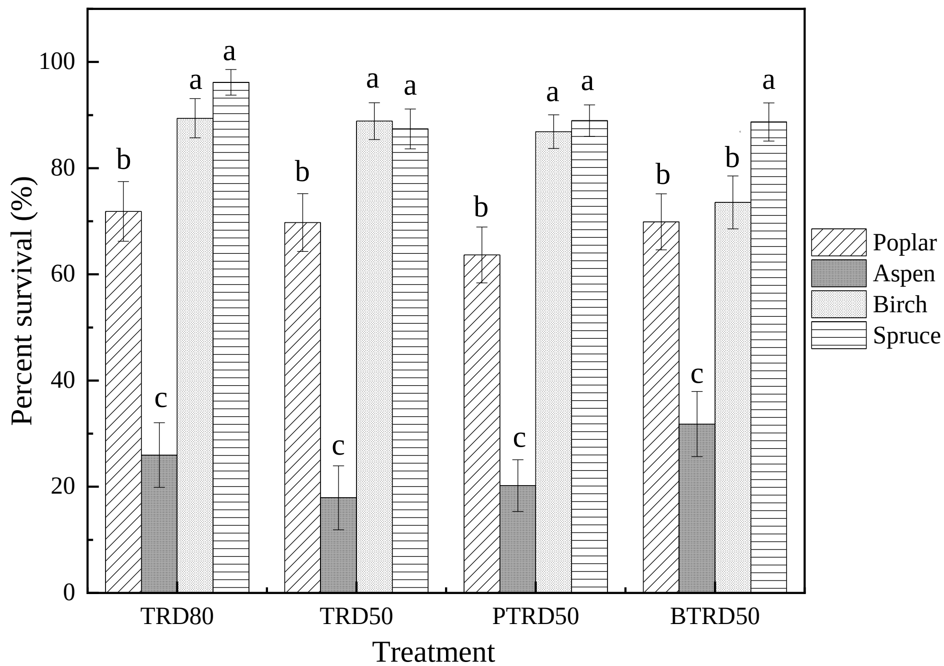


Figure 3. 4 Treatment by species effect on tree percent survival during reclamation of a borrow pit from 2015 to 2019 in Cold Lake. Error bars represent the standard error of the mean. Bars with different letters are significantly different following Turkey mean separation procedure.

### 3.5 Conclusion

Vegetation variables (canopy cover, species richness, tree height, and survival) differed significantly with topsoil replacement depth and organic amendment treatments. All treatments produced significantly greater native cover, while the TRD50 and BTRD50 treatments resulted in greater non-native cover in Year 5 of the study. Canopy cover and species richness of forbs, graminoids, and woody species did not vary with treatment but varied with time. While forb and non-native species cover and richness decreased with time, graminoid, native and woody cover and richness increased with time. Although diversity indices did not differ significantly among treatments, they were within the range considered normal for the boreal forest ecosystem. The heights of all tree species were significantly greater for the TRD80 and PTRD50 than the TRD50 and BTRD50 treatments. On the other hand, tree and shrub survival showed a declining trend from 2015 to 2019. While aspen and green alder had survival rates below 50%, other tree and shrub species had high survival rates. Our results indicate that peat can be used to improve vegetation establishment and plant community development when topsoil available for reclamation is insufficient (50% in this case) to attain the mandatory 80% TRD. By comparison, biochar showed no significant improvement in the vegetation variables when applied with insufficient topsoil (50% TRD). Nonetheless, all treatments met the characteristic species thresholds, indicating progression towards the desired moist rich d ecosite. Overall, our results point to the beneficial effects of peat in augmenting reclamation success when insufficient (50%) topsoil is available for reclamation of disturbed boreal sites. Continued monitoring would be beneficial in determining the long-term effects of topsoil replacement depth and amendments on ecosystem recovery as communities may shift over time in response to treatments.

### 3.6 References

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# 4 OVERALL SYNTHESIS

## 4.1 Summary of findings and Implications of research

Global population growth and accelerated industrialization have led to an increase in global demand for oil and natural gas. However, extraction of these non-renewable resources results in land disturbances, altering the aesthetics of the landscape, ecological biodiversity, and integrity of the boreal forest. This necessitates reclamation, which is the process of restoring disturbed land to achieve land capability equivalent to the pre-disturbed condition. This process requires sufficient topsoil, which serves as a foundation and a growth medium for plants during the revegetation process (Strohmayer, 1999; Larney et al., 2005).

Current recommended reclamation practices in western Canada call for replacing 80% of the original, pre-disturbance topsoil depth (ESRD, 2013). However, compliance with these regulations is not feasible for older wellsites that have insufficient or no salvaged topsoil to achieve the optimal topsoil depth unless the required topsoil can be imported from elsewhere. The scarcity of topsoil for reclamation and the fact that disturbed soils are typically deficient in organic matter necessitate alternative strategies, such as the use of organic amendments to reconstruct and improve the quality of disturbed topsoil, thereby promoting revegetation (Larney et al., 2005; Rowland et al., 2009). There is, therefore, a need to investigate the effects of TRD and organic amendments (peat and biochar) on the productivity of boreal region wellsites reclaimed using insufficient topsoil. The primary objective of this research was to determine the effect of suboptimal topsoil replacement depth with or without organic amendments (peat or biochar) on soil properties (Chapter 2) and early vegetation establishment (Chapter 3) relative to the recommended optimal topsoil depth (TRD80) during reclamation.

The soil properties study in Chapter 2 demonstrated that reducing topsoil replacement depth from 80% to 50% with no amendment showed minor variable effects on topsoil and subsoil chemical properties. While some treatments increased nutrient and other analyte concentrations in the soil, their effects varied from year to year and exhibited no consistent temporal trends. However, amending the suboptimal topsoil depth with peat (PTRD50) and biochar (BTRD50) increased TOC concentrations by 83% and 88%, respectively, relative to the mean of the unamended TRD80 and TRD50 treatments. Additionally, the peat amendment also resulted in a 113% increase in TKN concentration relative to the mean of all the other treatments (Chapter 2.). This implies that peat amendment increases plant availability of N, which is the most limiting nutrient in boreal forests. Our results clearly indicate that characteristics of peat and biochar, including high organic C and N and high-water holding capacity enhanced the quality of suboptimal (50% TRD) disturbed topsoil compared with the recommended optimal 80% topsoil replacement depth.

All topsoil and subsoil measurements passed the reference pre-disturbance soil characteristics assessment based on the 2010 criteria for forest ecosystem restoration (ESRD, 2013). An exception was topsoil depth, which was purposefully less than optimal for the 50 %TRD treatments. Given the large number of unreclaimed legacy and other oil and natural gas wellsites in Alberta with suboptimal topsoil volumes, our findings may assist reclamation specialists in enhancing or improving soil nutrient availability and quality through peat and biochar amendments. Additionally, because soil quality is primarily important for revegetation purposes (Chapter 3), using organic amendments such as peat and biochar helps the rapid restoration of these disturbed wellsites following reclamation.

The revegetation process is viewed as an important indicator of successful reclamation and restoration of forest ecosystems, with a goal of achieving equivalent land capability (ESRD, 2013).

Therefore, Chapter 3 examined the effects of peat and biochar on revegetation of oil wellsites reclaimed with suboptimal topsoil replacement depth. The study showed that species performance differed significantly with topsoil replacement depth and organic amendment treatment. The TRD80 and PTRD50 treatments had greater native cover than the BTRD50 treatment, while non-native cover decreased in all treatment plots 5 years after reclamation. Canopy cover and species richness of forbs and woody species did not vary with treatment but varied with time. While forb and non-native species cover and richness decreased with time, graminoid, native and woody cover and richness increased with time. Five years after reclamation, biochar amended plots had greater graminoid and non-native species than peat-amended plots. These species are undesirable in native boreal forest reclamation as they inhibit and delay the growth of desirable native perennial species. Species diversity indices (H' = 2.6; E = 0.71) were considered high across all treatments five years after reclamation. Reclaimed sites with high species diversity are regarded as stable and capable of rapid recovery following disturbance (Hooper et al., 2005; Dhar et al., 2018). Thus, the higher species diversity observed in our study may indicate that the reclaimed site is on trajectory towards a stable and resilient plant community (Dhar et al., 2018).

Within the five-year time frame of our investigation, the TRD80 and peat amendment treatments demonstrated exponential tree growth compared to the biochar and TRD50 treatments, which demonstrated slow tree growth. With the expectation of rapid tree and vegetation establishment in reclamation, peat amendment improved rapid tree establishment and growth to a level comparable to the mandatory TRD80, in comparison to the biochar treatment which showed slow growth responses. Thus, due to the recalcitrant nature of biochar, it can act as a slow-release and sustainable amendment over time. However, because of the exponential plant growth associated with the peat amendment treatment increases biomass in which the plants shed their leaves in the fall, thereby adding organic matter to the soil and establishing a self-sustaining ecosystem. Therefore, it is important to assess the performance of biochar and peat in long-term studies as treatment effects may shift over time.

While all treatments had no significant effect on tree and shrub survival, aspen and green alder had low survival rates (< 50%), whereas survival rates for the other tree and shrub species were in the high range (> 50%). The low survival rate of aspen was likely due to its shade intolerance, hence susceptibility to suppression by the highly competitive blue joint (*Calamagrostis canadensis*) (Landhäusser et al., 1996). Thus, the key reclamation goal is to rapidly establish trees to allow for a rapid canopy cover development, which can help in the control of shade-intolerant invasive species and competitive grasses that negatively impact the growth and survival of desired perennial native species. Reclamation specialists should, therefore, critically select species with high growth and survival rates when reclaiming disturbed forests to ensure a rapid and successful ecosystem recovery. Additionally, natural ingress control and management techniques, particularly for competitive graminoids and non-native species, may be beneficial for the growth and support of desired perennial woody species on reclaimed boreal sites.

Since undisturbed native boreal forests are characterized by perennial native forbs and the absence of graminoids and non-native species, it can be inferred that reclamation success for plant community development, species richness, woody plant height, and invasive species reduction was greater on peat-amended plots and TRD80 than on unamended 50% TRD and biochar-amended plots. Therefore, TRD80 and PTRD50 treatments demonstrate a greater rate of recovery and a trajectory towards the natural boreal forest ecosystem. This study provides reclamation specialists and policy-makers with additional tools to refine guidelines for forested land reclamation.

## 4.2 Recommendations

This study is one of the first to examine the reclamation of boreal wellsites using suboptimal topsoil replacement depth in conjunction with organic amendments. Findings from the study indicated that while reduced topsoil replacement depth had no effect on soil nutrient availability or soil quality five years following reclamation, adding peat and biochar improved soil nutrient availability. Our study focused more on topsoil nutrient availability and some chemical elements such as EC, SAR, and pH. However, only bulk density and moisture percent saturation were the physical properties measured. We recommend that future studies should examine TRD and amendment effects on additional soil physical properties, including saturated hydraulic conductivity, infiltration rate, resistance to penetration, and soil thermal properties. Given the short duration of this study, measurements of labile soil nutrients such as light fraction carbon, light fraction N, and potentially mineralizable C and N could have provided a more sensitive indication of the progression in soil quality.

Vegetation measurements showed that reducing TRD from 80% to 50% had a significant effect on revegetation. With certain vegetation parameters, such as tree height, the TRD80 and peat amendment was greater than TRD50. Despite the cost associated with the production and application of biochar, biochar had no apparent benefit on vegetation variables at the rate tested relative to peat. Biochar performance is dependent on the feedstock material used in biochar preparation (Thomas and Gale, 2015; Page-Dumroese et al., 2016); thus, it is possible that the woody-pine biochar used in our study may not support or improve vegetation establishment and growth of desirable species. Additionally, the effectiveness of biochar in our study may have been delayed as biochar might have temporarily immobilized some nutrients such as nitrogen during the five-year period. Therefore, we recommend that future research should evaluate the use of different types of biochar at different rates as this may yield a different result. Additionally, continuous monitoring beyond five years may be necessary to determine the long-term effect of biochar on reclamation success.

While undisturbed native boreal forests are devoid of non-native species and competitive graminoids, reclaimed boreal sites exhibit non-native species dominance, particularly during the early years of reclamation, due to the high proportion of bare ground. This often delays establishment of perennial native species. Therefore, future studies should examine how reclamation strategies such as planting of trees at high densities may assist in reducing non-native species dominance by reducing bare ground during reclamation of boreal sites. One of the requirements of revegetation of boreal forests is the establishment of a woody stem count greater than 2,000 stems/ha (ESRD, 2013). In our study, we observed a decrease in woody stem count from 3125 stems/ha to 1728 stems/ha, which is less than the minimum requirement. Therefore, we recommend selection of species with high growth and survival rates in combination with nitrogen-fixing native shrubs and forbs to enhance N availability as N is the scarcest nutrient. Additionally, further investigation on aspen and green alder is needed to determine the factors affecting the survival of these tree species during boreal site reclamation.

Some researchers suggest that the use of forest floor as an amendment can significantly accelerate ecosystem recovery due to its high native seedbank sourced from the upland forest (Mackenzie and Naeth, 2009; Dhar et al., 2020). Thus, future studies should explore the potential for other cost-effective organic amendments, such as forest floor, biosolids and papermill sludge, at various rates to augment boreal site reclamation using suboptimal topsoil.

Since soil nutrient status and plant community evolve over time, ongoing monitoring is necessary to provide valuable insights into the long-term effects of TRD and organic amendments on soil properties and soil function as well as ecosystem recovery. Overall, regardless of the topsoil depth limitation, our short-term findings indicate that, based on soil and vegetation properties, reclamation goals can be achieved with suboptimal (50%) TRD amended with peat to a level comparable to the recommended 80% TRD in cases where topsoil is not readily available to achieve the recommended TRD.

## 4.3 References


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**APPENDICES**

**Appendix A: Supplementary tables**

Table S. 1 Number of seedlings transplanted per plot

| **Type** | **Species** | **Common name** | **Number of plants planted per plot** |
|---|---|---|---|
| Trees | *Betula papyrifera* | White birch | 20 |
| | *Populus balsamifera* | Basalm poplar | 20 |
| | *Populus tremuloides* | Trembling aspen | 40 |
| | *Picea glauca* | White spruce | 15 |
| Shrubs | *Alnus viridis* | Green alder | 10 |
| | *Amelanchier alnifolia* | Saskatoon | 10 |
| | *Cornus stolonifera* | Red osier dogwood | 10 |
| Total per plot | | | 125 |

Table S. 2 List of characteristic species in all the treatment plots in a reclaimed borrow pit at Cold Lake, Alberta.

| **Lifeform** | **Scientific Name** | **Common Name** | **Treatment** | | | |
|---|---|---|---|---|---|---|
| | | | **TRD80 (Control)** | **TRD50** | **PTRD50** | **BTRD50** |
| Graminoids | *Calamagrostis canadensis* | bluejoint | X | X | X | X |
| Herbs | *Chamerion angustifolium* | common fireweed | X | X | X | X |
| | *Equisetum arvense* | common horsetail | X | X | X | X |
| | *Equisetum sylvaticum* | woodland horsetail | X | X | X | X |
| | *Fragaria virginiana* | wild strawberry | X | X | X | X |
| | *Galium triflorum* | sweet-scented bedstraw | X | | | |
| | *Lathyrus ochroleucus* | cream-colored vetchling | X | X | X | |
| | *Mertensia paniculata* | tall lungwort | X | | X | |
| | *Rubus pubescens* | dewberry | X | | | |
| Shrubs | ***Alnus viridis*** | green alder | X | X | X | X |
| | ***Amelanchier alnifolia*** | saskatoon | X | X | X | |
| | ***Cornus stolonifera*** | red-osier dogwood | X | X | X | X |
| | *Lonicera involucrata var. involucrata* | bracted honeysuckle | | | X | |
| | *Ribes glandulosum* | skunk currant | X | X | X | |
| | *Rosa acicularis* | prickly rose | X | X | X | |
| | *Rubus idaeus* | wild red raspberry | X | X | X | X |
| | *Salix arbusculoides* | shrubby willow | X | X | | X |
| | *Salix bebbiana* | beaked willow | X | X | X | X |
| | *Salix discolor* | pussy willow | X | X | X | X |
| | *Salix interior* | sandbar willow | X | X | | X |
| | *Salix lasiandra* | shinning willow | | X | | X |
| | *Salix maccalliana* | velvet-fruited willow | | | X | |
| | *Salix pseudomonticola* | false mountain willow | | X | X | |
| | *Symphoricarpos albus* | snowberry | | | X | |
| Trees | ***Betula papyrifera*** | white birch | X | X | X | X |
| | ***Picea glauca*** | white spruce | X | X | X | X |
| | ***Populus balsamifera*** | balsam poplar | X | X | X | X |
| | ***Populus tremuloides*** | Trembling aspen | X | X | X | X |
| ***Total*** | | | ***23*** | ***22*** | ***23*** | ***17*** |

X – present, bolded species represent planted species

Table S. 3 Poisson regression analysis of species richness with year as the explanatory variable.

| **Response variable** | **Intercept** | **Slope** | **Exponential of slope** | **95% Cl** | | **P** |
|---|---|---|---|---|---|---|
| | | | Poisson regression | | | |
| Total richness | 3.70 | 0.031 | 1.031 | 0.004 | 0.058 | **0.02** |
| Forb richness | 3.38 | -0.087 | 0.917 | 0.013 | 0.08 | **<0.0000** |
| Graminoid richness | 1.78 | 0.194 | 1.215 | 0.139 | 0.25 | **<0.0001** |
| Woody richness | 2.07 | 0.059 | 1.061 | 0.002 | 0.118 | **0.04** |
| Native richness | 3.17 | 0.095 | 1.069 | 0.041 | 0.09 | **<0.0001** |
| Non-native richness | 3.01 | -0.216 | 0.806 | 2.85 | 3.17 | **<0.0001** |
| | | | Binomial regression | | | |
| Tree survival | 1.77 | -0.32 | 0.72 | 0.93 | 2.60 | **0.01** |
| Shrub survival | 1.53 | -0.23 | 0.79 | 0.59 | 2.47 | **0.07** |

## Appendix B: Supplementary figures

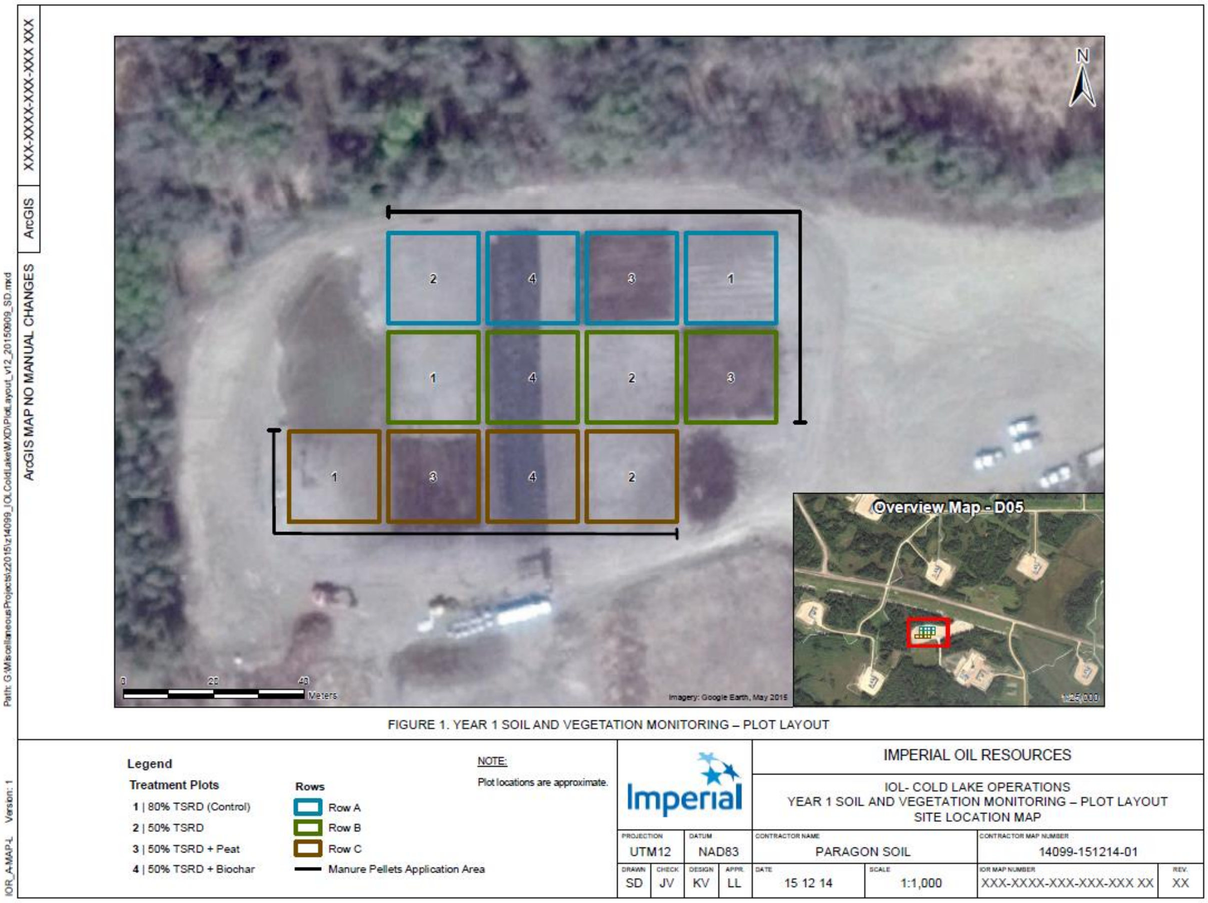


Figure S.1 Experimental layout at Cold Lake Operations, Alberta.

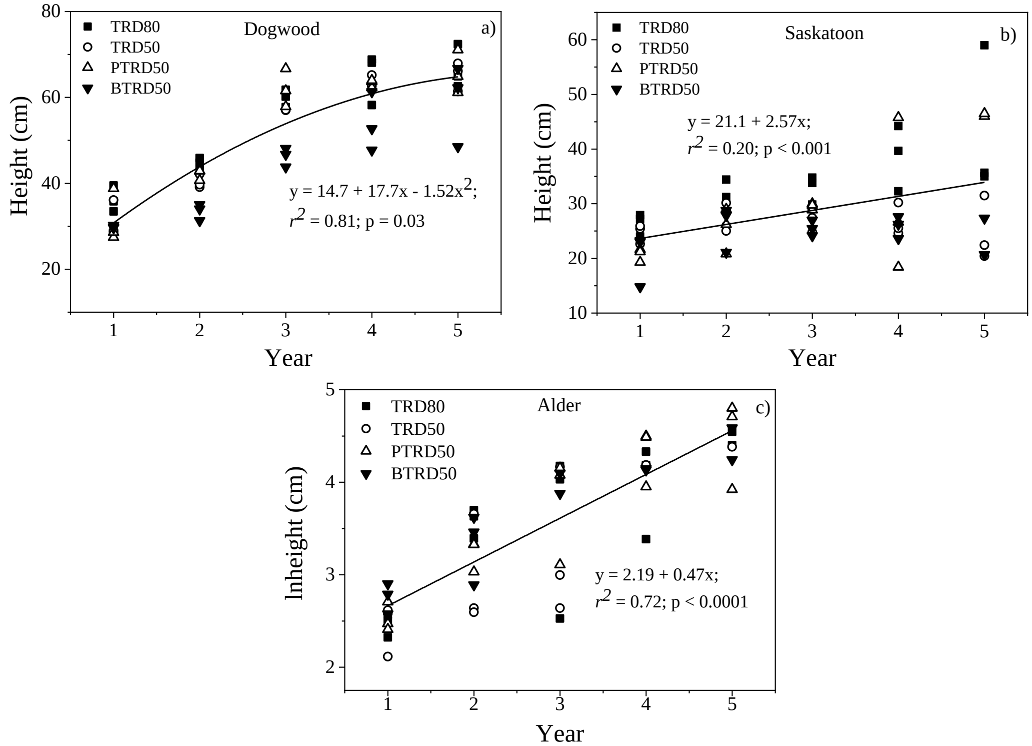


Figure S.2 Effect of treatments and year on the height of the shrubs (a) dogwood, (b) saskatoon, and (c) alder averaged across all treatments during reclamation of a borrow pit in northeastern Alberta. Year 1, 2015; Year 2, 2016; Year 3, 2017; Year 4, 2018; Year 5, 2019.