Revegetation of a non-acid generating mine tailings pond in boreal Manitoba

By:

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Abstract:

In order to encourage the establishment of an erosion controlling vegetative ground cover, low cost organic amendments and inorganic fertilizers were incorporated into an abandoned, non-acid generating mine tailings pond. By amending these gold mine tailings with a small quantity (3.7 kg m^{-2}) of combined papermill sludge and fertilizer, a consistent and robust grass/*Medicago sativa* sward was established within the first growing season. This tailings amendment and the subsequent plant establishment lead to improvements in substrate fertility characteristics including aggregation, bulk density, as well as cation exchange capacity and organic content. Chemical fertility (available nitrogen and phosphorous) was ameliorated by all initial amendment treatments. However, the effect was relatively short-lived, with all amendment treatments returning to their background levels of these nutrients within one to two years. This study further proves the usefulness of papermill sludge as a low cost amendment for disturbed soil substrates, including mine tailings.

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Table of Contents:

List of Tables:

List of Figures:

Chapter 1. Introduction

Mining activities impact the local and regional environment in many ways throughout exploration, site development, mining, as well as mineral processing. Among the most harmful and long lasting environmental liabilities of many mine operations is the generation of huge quantities of milled and processed waste rock, or tailings and their subsequent dumping in natural environments (Ripley, 1996). This mine waste can vary tremendously in terms of their physical and chemical composition; however some of the common harmful characteristics include an enrichment of some heavy metals or process chemicals, extremes of pH, as well as an overall lack of available nutrients and organic matter (Lottermoser, 2003). Mine tailings impact natural environments in many immediate and long lasting ways, through direct loss of ecological habitat, or through the contamination of interconnected environments through particulate and leachate movement. Many techniques have been developed in order to mitigate some of the environmental impacts of mine tailings and restore ecological and/or economic function to the disturbed land. In situations where erosive transport of the tailings is a significant concern, one of the most popular and practical techniques is to encourage a vegetative cover capable of binding the substrate and controlling erosion. This approach, termed phytostablilization, takes advantage of the physiology and growth morphology of cover vegetation to mechanically bind primary particles, as well as in some cases adsorb and sequester substances present within the soil environment (Mendez & Mainer, 2008).

Given the remote nature of many boreal mining operations as well as the loose environmental regulations governing the industry during much of Canada's mining history, the Canadian boreal forest is littered with thousands of abandoned mine sites (Hogan & Tremblay, 2008). These sites remove land from important ecological functions such as carbon sequestration, nutrient cycling, habitat, water regulation and filtration, as well as erosion control. Low cost, minimal input technologies and approaches to mitigate some environmental impact of these sites are critical in order to encourage the remediation of many ancestral orphaned or abandoned mine sites for which responsible owners are no longer present to undertake appropriate remediation measures. The present study examined the effectiveness of certain low-cost, locally sourced amendment and revegetation strategies based on both overall improvements to mine tailings fertility as well as plant growth and stress measures. Significant success in terms of encouraging the establishment of a vegetative cover capable of limiting tailings erosion was witnessed in the following study with the use of a minimal quantity of locally sourced amendments. This minimal input lead to improvements in terms of substrate fertility that was, in some cases, comparable to that witnessed after many decades of natural revegetation.

Chapter 2. Literature Review

2.1 Introduction

Canada's longstanding and continued position as one of the world's largest and most diverse mineral producing countries has generated much prosperity in addition to much environmental disturbance. The Canadian boreal forest is home to a large proportion of the countries' primary resource industries, including mining, forestry, as well as the various energy production industries. While all of these industries impose some level of disturbance, without considerable forethought and long-term efforts, mining activities may negatively affect local and regional environments well after the lifetime of the mine. Given the remote nature of many boreal mining operations as well as the loose environmental regulations governing the industry during much of Canada's mining history, this region is littered with thousands of abandoned mine sites requiring various levels of remediation effort.

Among the most harmful and long lasting environmental liabilities of many mine operations is the generation of huge quantities of milled and processed waste rock, or tailings. This mine waste can vary tremendously in terms of their physical and chemical composition. Some of the common harmful characteristics include an enrichment of some heavy metals or process chemicals, extremes of pH, as well as an overall lack of available nutrients and organic matter. Many techniques have been developed in order to mitigate some of the environmental impacts of mine tailings and restore ecological and/or economic function to the disturbed land. The range of remediation options range from complete isolation of the waste, to varying levels of amendment and revegetation, and in some situations it may be appropriate to do nothing and allow natural successional processes to reclaim the land.

The following review examines the importance and the impact of the Canadian mining industry on the boreal forest region. Mine tailings are described, specifically as it relates to the various methods of remediating of the wastes to support a vegetative community. In addition, by describing and studying the successional processes that have, in many cases, naturally reclaimed many ancestral mine sites, an appreciation for the elements of technical restoration projects can be developed; as such the final section of this review will provide an examination of naturalized mine tailings.

2.2 Mining in Canada and its impact on the environment

2.2.1 Boreal Forest biome

The boreal forest is one of the world's largest biomes and represents one quarter of the remaining intact forest ecosystems in the world (Carlson et al., 2009). It occupies approximately 11.6 million km^2 throughout the arctic and sub-arctic regions of North America, Asia and Europe (Runesson, 2011). In Canada, the boreal forest covers more than 55% of the total area, stretching from Newfoundland to the Yukon (Canadian Boreal Initiative, 2005) with its northern and southern limits correlating to the July 13°C and July 18°C isotherms, respectively (Bonan et al., 1992). The Canadian boreal forest is characterized by a continental climate with short, warm growing seasons, long cold winters and a moderate level of precipitation (550-1000 mm, increasing to the east and south). Three major vegetation zones are common in this biome: 1) the taiga (the most northern region) is described as a forest-tundra mosaic, 2) the Northern boreal zone (the central boreal region) is visualized as open crown or lichen-woodland that may have a strong shrub presence and 3) the Southern boreal zone (the southernmost regions) is described as a closed-crown forest with increased diversity relative to other boreal zones in terms of understory plant community (Rowe & Scotter, 1973). In all vegetation zones, coniferous trees are the most common dominant canopy members, though broad leaf deciduous trees can dominate in southern or recently disturbed or exposed areas (Rowe & Scotter, 1973; Canadian Boreal Initiative, 2005). Fire is the predominant natural type of large-scale disturbance throughout the boreal forest, and has a significant influence on the overall community composition and development (Niemela, 1999). The Canadian boreal forest can be further divided into seven distinct ecozones, these are: the Boreal Shield, Boreal Cordillera, Boreal Plains, Taiga Shield, Taiga Cordillera, Taiga Plains and Hudson Plains (Northwatch et al., 2008). Approximately 30% of Canada's boreal region is composed of wetland ecosystems, and with over 1.5 million lakes and some of the largest river systems in the world; a landscape rich with water features is characteristic of the Canadian boreal forest (Canadian Boreal Initiative, 2011).

The importance of the Canadian boreal forest on the local and national scale cannot be understated. Fourteen percent of Canada's population (3.5 million) (Northwatch et al., 2008) resides within the boreal forest, including over 600 first nations communities with populations exceeding a million people (CBI, 2005). The forest has long provided all resources necessary for the survival of its inhabitants;

boreal plant species are utilized for food, medicine, shelter as well as transportation and can hold spiritual and cultural significance (Karst, 2010). Activities throughout the boreal forest contribute significantly to our national economy; most notably these are the forestry, mining and energy as well as recreation services industries (Anielski & Wilson, 2002). Approximately half of all Canadian forestry harvesting occurs in the boreal forest, and almost 400,000 jobs are directly or indirectly provided by the boreal forestry sector (Anielski & Wilson, 2002). In Canada's boreal forest, 583,000 km^2 is currently staked for mining activities and 105 mines are currently active within this area (Canadian Boreal Initiative, 2008). More than 150,000 oil or gas wells are active within the boreal forest, and nearly 40% of Canadian generated hydroelectric power comes from rivers that run throughout the region (Wells el al., 2010). Additionally, on an annual basis, over six million Canadians use the boreal forest for recreational purposes, thereby generating approximately 4.5 billion dollars for many associated industries (Anielski & Wilson, 2002). All in all, the annual net market value of the boreal forest to the national economy has been estimated at 37.8 billion dollars (Anielski & Wilson, 2002).

While the Canadian boreal forest has provided for great prosperity, the global impact of this region in terms of climate regulation, water quality protection and wildlife habitat is significantly more important (Anielski & Wilson, 2002). The boreal forest is the largest global carbon sink, storing over 208 billion tones of C in its forest and peatland systems, and accounting for more than 80% of the carbon stored in all of Canada's forest ecosystems (Carlson et al., 2009; Carlson et al., 2010). Carbon is stored in vegetation, soils, permafrost, wetlands and peatlands, lake sediment as well as alluvial deposits (Carlson et al., 2009; Wells et al., 2010). That being said, the vast majority or boreal carbon is stored in the soils (85 and 98.5% for forest and peatlands, respectively) (Carlson et al, 2009). Carbon dynamics within the boreal forest is the primary mechanism through which it affects global climate regulation and fluctuations in incidence of disturbances will affect its status as a sink or source of carbon (Carlson et al., 2010). Additionally, freshwater flows from the Canadian boreal forest into three oceans as well as Hudson bay, having profound effects on oceanic currents as well as sea ice condition; both significant contributors to global temperature dynamics (Wells et al., 2010). One third of the world's and 93% of Canada's peatlands are contained within the boreal region (Wells et al., 2010; Carlson et al., 2010). Peatlands, as well as other wetland areas are essential to the protection of water quality as they can bind and absorb particulate matter and contaminants, play a role in recharging aquifers, and regulate water flow in rivers (Wells et al., 2010). The diverse habitats established throughout the boreal forest region are essential to billions of migrating birds (more than 30% of North American migrating birds)(Canadian Boreal Initiative, 2011), some of the world's richest fish populations (including Atlantic salmon, chum salmon, brook trout, Arctic char, inconnu, lake whitefish, Arctic grayling, lake trout and northern pike)(Wells et al., 2010), as well as animals such as the woodland caribou, moose, deer, bear, wolf, lynx, fox, beaver, muskrat as well as may more (Wells et al, 2010; Canadian Boreal Initiative, 2011).

2.2.2 Mining in Canada

Owing to its expansive geography and diverse geological formations, Canada has been among the world's largest mineral producing countries for over a hundred years (Wallace, 1996). The mining industry, which comprises metal, non-metal and oil sands mining, as well as operations related to extraction, refinement and fabrication of mined materials, contributed over 32 billion dollars towards the gross domestic product (GDP) of Canada in 2009 (MAC, 2011). Provinces including Alberta, Saskatchewan, British Columbia, Manitoba, Newfoundland and Labrador as well as New Brunswick derive significant portions of their revenue from the mining sector (MAC, 2011). Additionally, mining activities employ over 300 000 workers throughout rural and urban Canadian communities; approximately 1/50 Canadian jobs are provided by the mining industry and over the next decade, an estimated 10,000 mining related jobs would need to be filled on a yearly basis (MAC, 2011). Canada is home to 19 of the worlds''s 'top 100' mining companies (more than any other county), and it remains the top destination for foreign exploration, attracting 16% of the world's exploration investment (MAC, 2011).

Historically, native north-American cultures have utilized mineral resources in the production of tools, weapons and decorative objects for thousands of years. However mining on a large scale in Canada began following European colonization (Cranstone, 2002; Innis, 1948). Initially, mineral extraction and use was limited to locally available materials used for building (such as stone, brick clay, sand, gravel as well as lime) as well as the mining of rocks to be used as grindstones (Cranstone, 2002; MIEN, 2011). Deposits of coal were discovered in Cape Breton, Nova Scotia, and by the year 1711, British settlers had begun coal extraction (Cranstone, 2002). The first long-term mining lease was issued in 1828 to the General Mining Company, allowing 36 years of coal mining in Cape Breton, Nova Scotia (Innis, 1948). Other important early mining activity included utilizing copper reserves of the Lake Superior region and later Lake Huron, gypsum beds in Nova Scotia and later Ontario, iron from along the St. Lawrence (most notably as in Three Rivers region) as well as petroleum reserves near Lake Erie (Innis, 1948; Cranstone, 2002). The discovery of placer gold (alluvial depositions of a precious metal) in the Californian Cordilleran region in 1848 brought about a new era in mineral exploration (Innis, 1948). The ensuing gold rush resulted in a massive population and economic boom, requiring substantial coal reserves that were partially satiated by mining operations of Vancouver Island (where coal mining begun around 1835) (Cranstone, 2002). Additionally, northward explorations of the Pacific coast Cordilleran region lead to gold discoveries and subsequent rushes of population and industry throughout southern British Columbia. Placer gold was discovered in the Queen Charlotte Islands in the late 1850s, followed by the Fraser Canyon (1858) and Cariboo river (1861) regions of southern British Columbia (Innis, 1948; Cranstone, 2002). The gold rushes that followed new discoveries drew interest from many mine developers, hastening the development of major transportation routes as well as other infrastructure and technologies necessary for further mineral development throughout the Pacific coast (Innis, 1948). The climax of placer mining occurred in 1896 when gold deposits were discovered along the Klondike River in the Yukon and Alaska, initiating the Klondike Gold Rush. The extensive prospecting for gold that took place in southern B.C. lead to the discovery and production of a number of other base metals, including silver, copper, zinc and lead (Cranstone, 2002). Elsewhere in Canada, phosphate was being mined for agricultural purposes in western Quebec beginning in the 1870s. Additionally, huge quantities of asbestos were discovered in eastern Quebec in the 1880s; this region has remained a top supplier of asbestos ever since (Cranstone, 2002). The development, as well as the eventual completion, of the Canadian Pacific Railway (CPR) effectively linked eastern and western Canadian trade, and advanced the mining industry in Canada considerably. Not only did the new railway mean that mineral resources could more effectively be transported long distances, but many potential mine claims were discovered throughout the establishment of the railway (Cranstone, 2002). In 1883, a massive nickel-copper deposit, which has been mined ever since, was discovered in Sudbury, Ontario as a result of railway construction. Similarly, one of the world's largest silver deposits was discovered near Cobalt, Ontario in 1903 during railway construction (Cranstone, 2002). With a capable freight system in place, mining, refinement and fabrication operations could operate more independently, allowing for greater efficiency and autonomy. Even today, more than half of the total rail and port freight shipped by Canada is derived from the mining industry (MAC, 2011).

Prior to the early 1900s, the majority of mining discoveries were accidental; however, beginning in the 1920s, mining companies began employing more sophisticated exploration teams, which would include trained geologists, in order to prospect regions of the country (Cranstone, 2002). Unfortunately, during the Great Depression of the 1930s there was little mineral exploration and mining taking place, the exception being for gold and silver. When the Second World War began in 1939, many mining operations were forced to cut back significantly or even cease operation fully. Other operations switched to, or were already in production of, materials for war and thus played an important military role (Northwatch et al., 2008; Cranstone, 2002). At this time, several technological innovations occurred that would allow for much more effective mineral exploration. The invention of the Geiger counter, followed by the advancement of the technology in the form of scintillometer and later gamma ray spectrometer, allowed for the detection of radioactive materials. These technologies lead to the discovery of uranium deposits beginning in the 1930s and 40s, which include several situated in Northern Saskatchewan and Ontario. Additionally, the invention of an airborne magnetometer (whose military application was the detection of submarines) allowed for the discovery of ore that exist within magnetically charged mineral deposits such as magnetite and pyrrhotite. Ground electromagnetic (EM) systems were developed during the 1930s based on the concept that sulphide and vein-type ore bodies are electrically conductive, and by the 1950s this technology, too, had been adapted for aerial use (Cranstone, 2002). With the use of aerial EM and magnetometer systems, explorations occurred much more rapidly and lead to the development of many base metal mining operations throughout Canada during the second half of the 20th century. While prospecting had been limited to areas of outcropping rock features, these new technologies have allowed for the sub-surface detection of magnetically or electrically conductive indicator minerals. As a result, the extensive areas of Canada that is covered by over-burden or permafrost are subject to mineral explorations and possible mining. Since 1946, over two thousand mining discoveries leading to actual mining have been made in Canada, an average of over 40 discoveries per year (Cranstone, 2002).

2.2.3 Mining in Manitoba

Mining is a very important primary resource industry within the province of Manitoba, second only to agriculture in terms of contribution to the provincial economy (MIEM, 2011). Seven percent of the provincial gross domestic product (GDP) as well as 11% of the provincial exports are derived from the mining industry (Government of Manitoba, 2008). The mineral industry in Manitoba directly provides approximately 6,100 jobs (MIEM, 2011) as well as an estimated 14,000 in indirect, spin-off jobs (Government of Manitoba, 2008). Additionally, the development of mining operations in northern Manitoba has lead to the establishment of many communities and cities including Flin Flon, Thompson, Lynn Lake, Bissett, Wabowden, Leaf Rapids and Snow Lake (MIEM, 2011).

Historically, salt was produced from the western springs of Lake Manitoba and Winnipegosis, and aggregate production (sand, gravel, crushed stone) dates back to some of the earliest settled communities (MIEM, 2011). Oil has been produced in the southwestern corner of the province since 1951, and current production provides significant economy and employment to the towns of Virden, Melita, Waskada and Pierson (MIEM, 2011). Metal mining operations are concentrated in the western portion of the province, north of Lake Winnipeg, as well as in the southeast, near Bissett, MB. Nickel, copper, zinc, gold, cobalt, silver, as well as lithium, cesium, selenium and tantalum are all mined in Manitoba, and in the case of lithium and cesium, this constitutes the only Canadian reserve (MIEM, 2011; MAC, 2011). Given the large landmass (almost 550,000 km^2), and diverse geological formations, Manitoba is still considered to be relatively under-explored (Government of Manitoba, 2008). The potential for diamond, uranium, platinum group metals (PGMs), rare-earth elements (REEs), iron-oxide-copper-gold (IOCG) and potash mining operations should ensure that this industry remains one of Manitoba's most important in the years to come (Government of Manitoba, 2008).

Over the past 100 years, more than seventy mines were active within Manitoba, and within the past decade 31 mines, two smelters and two refineries have operated within the province (Government of Manitoba, 2008). Currently, 13 mines are actively producing within Manitoba (MAC, 2010) and 209 million tons of mine tailings (fine grade processing wastes) as well as 102 million tons of waste rock cover an area of approximately 2,400 ha within the province (MSSC, 1996). As a result of a general lack of regulation during ancestral mining activities relating to the disposal of mine wastes throughout the province, and largely within the boreal region, many former and current mine sites present significant environmental liabilities (Slivitzky, 1996). These mine sites may be designated as orphaned or abandoned if the mine operator(s) cannot be identified or are financially unable to undertake remediation efforts, in which case the impetus for reclamation falls on governmental agencies (NOAMI, 2004). In 2000, the Manitoba Orphaned and Abandoned Mine Site Rehabilitation Program was established with a goal of identifying and remediating all orphaned and abandoned mine sites within the province. One hundred and forty nine former mine sites were identified, with five being classified as hi-priority (Gods Lake, Snow Lake, Lynn Lake, Sherridon Mine and Baker Patton) and another 31 as hi-hazard (Northwatch et al., 2008). Current remediation goals aim to resolve the environmental hazard established at the hi-priority and hi-hazard mine sites by the end of 2012 (Mines Branch, 2009).

2.2.4 Wastes generated by mining activities:

Mining is an intrinsically destructive process that requires huge volumes of earth to be cleared, large volumes of rock to be extracted and processed by physical and chemical means, and ultimately the vast majority of what is mined must be disposed of as waste. At any given metal mining operation, common activities can include mining, or mineral extraction, as well as mineral processing and metallurgical extraction (Lottermoser, 2003). Throughout the process, waste products are generated in the forms of solid waste, gaseous and particulate emissions as well as impacted water and process chemicals (MSSC, 1996).

Large volumes of solid wastes are generated during both the mining and the processing of the mineral ores (Ripley et al., 1996). Mining wastes comprise vegetation, top soils, rocks and loose sediments that were removed from the mining area in order to establish facilities and infrastructure, as well as unwanted rocks extracted in order to access the desired mineral ores (Lottermoser, 2003). This type of solid mine waste is often termed mine spoil, and open-pit mining (employed when the desired ore is within ~ 100 m of the surface) tends to generate much more of this waste as compared to sub-surface operations (Dudka& Adriano, 1997). As the mineral ore is treated physically and chemically in order to release the desired mineral from the unwanted (gangue) matrix, often upwards of 95% of the extracted rock body may become waste. The solid wastes generated through mineral processing are in the form of tailings and sludge, with mine tailings being the most voluminous waste product generated at most mine sites (Lottermoser, 2003). Mining facilities in Canada generally discharge approximately 98% of the mined materials as solid waste with ~ 52% as tailings, 42% as waste rock, and 4% as slag (waste produced from the smelting or metalliferous ores) (Northwatch et al., 2008). The processing of mineral ores through mechanical and chemical means will generate fine grade waste rock that is often enriched with heavy metals as well as process chemicals (Lottermoser, 2003). This waste, called tailings, can range from colloidal to coarse gravel sized particles and, like mine spoil, their chemical and physical nature depends on factors such as mineralogy and geochemistry as well as the processes performed to liberate the desired mineral (Lottermoser, 2003). Unlike waste rock, these tailings often undergo extensive treatment with process chemicals that can rarely be recovered efficiently, or in an economically viable fashion. The types of process chemicals often employed include acids and bases, oils, oxidizing agents as well as metallic compounds. The remnant levels of these contaminants in the tailings will influence the potential environmental impact (Lottermoser, 2003). Additionally, metal mineral ores are naturally rich in some desired metal, and in most cases these will co-occur with other metal minerals. As a result, following processing to recover a certain desired metal, the unwanted mine tailings will often remain enriched in other metals that were present within the ore bodies (Ye et al., 2002).

Mineral processing in the form of smelting and refining also introduces large amounts of gaseous compounds (including oxides of sulfur) as well as particulate matter into the atmosphere (Ripley et al., 1996). In fact, smelters are the main source of arsenic, copper, cadmium, antimony and zinc and a significant contributor to chromium, lead, selenium and nickel in the atmosphere (Dudka & Adriano, 1997). In recent past, Manitoba has the distinction of having two of the three worst air polluters in Canada; during operation the smelters in Thompson and Flin Flon accounted for almost half of Canada's sulfur dioxide emissions, and 95% of Manitoba's emissions (Northwatch et al., 2008). Great improvements have been made in terms of reducing and treating emissions from smelters over the past decades, however they remain significant pathways for harmful gaseous compounds and particulate matter to enter the atmosphere (SCNR, 1996). Among metal refining operations, it tends to be the copper, lead and zinc processing facilities that generate the greatest amount of this air borne waste (Dudka & Adriano, 1997). Furthermore, as mine tailings dumps dry, wind forces can erode the fine, unaggregated surface particles, lifting them into the atmosphere. During active mining, much of the tailings impoundments tend to remain under ponded water (Blowes et al., 2003). Without continual input of water, depending on the regional climate and the position of the water table, the tailings surface will dry leaving them susceptible to erosion (Barbour, 1994). In fact, Environment Canada estimated 63,000 tons of particulate emissions generated from mine tailings dumps in Canada in one given year (MSSC, 1997).

When discussing waste products generated through mining activities it is critical to consider the water that has been adversely affected, and like solid wastes, the mine water characteristics are unique to the mine site (Pentreath, 1994). Sub-surface water, or water which has collected in open-pit mines or in mine shafts may be classified as waste water if it exhibits elevated levels of metals (leached from exposed rock) or other chemicals (such as fuels or explosive residues) (Lottermoser, 2003). Various types of mineral processing may degrade water by exposing it to contaminants; these include grinding of ore, hydrometallurgical processes as well as the movement and disposal of milled tailings in the form of slurry (Ripley et al., 1996; Lottermoser, 2003). Additionally, new water sources (precipitation for example) are exposed to different solid and gaseous forms of mine wastes; they too may become impacted by harmful contaminants, and in turn become wastewater.

Mining as well as mineral processing often require the use of various chemical substances, including fuels, acids and bases, oils, oxidizing agents as well as metallic compounds (Lottermoser, 2003). The use of these chemicals can generate waste substances that must be handled appropriately in order to avoid environmental impact.

Finally, when considering the wastes generated throughout mining operations and activities, it is important not to overlook the impact of human wastes. Many mining operations situated in remote locations require the establishment of housing and other facilities that serve the mining workforce, not to mention transportation and energy providing infrastructure (Northwatch et al., 2008). The generation of human wastes throughout the lifetime of remote mining operations requires the construction of landfills or other waste facilities in order to appropriately treat the material. When mining operations come to an end at a particular site, due to economic reasons and/or the depletion of desired ores, many of these facilities become waste products themselves, often requiring significant effort to remove (Lottermoser, 2003).

2.2.5 Environmental impacts of mining activities:

The environmental impacts of mining activities can often be observed even prior to the development of a mine, and if mine wastes are not treated appropriately, their effects may persist for decades if not centuries. The process of exploration for mineral deposits may require the clearing of vegetation through survey lines, the establishment of drill sites as well as the use of heavy equipment (Lottermoser, 2003). While these disturbances are relatively small in comparison to extractive mining, the impact on the local community can be significant. The dissection of forested land by survey lines and exploration roads establishes conditions that favor the introduction of weedy, potentially invasive plant species to these forest corridors (Gelbard & Belnap, 2003). Additionally, studies have shown a negative effect of anthropogenic noise, specifically that produced by the mining sector, on the density of some boreal songbirds (Bayne et al., 2008). Of the 23 boreal birds included in the study, one third exhibited decreased density in habitat where anthropogenic noise was present, and the overall passerine density was 1.5 times greater in habitat without the noise. The same study observed negative forest edge effects on density in four of the 23 boreal bird species considered (Bayne et al., 2008).

Mine site development includes the clearing of vegetation, the construction of roads, facilities and other infrastructure (Lottermoser, 2003). Lands cleared for development of roads and other facilities may require considerable landform change in order to house the desired structure. Therefore, the impact on the local environment can include a reduction in primary productivity due to a loss of vegetation and soil biota (Ripley et al., 1996) as well as disturbed local hydrology of the region (Johnson & Miyanishi, 2008). Cleared materials must be disposed of or stockpiled for alternative uses. Neither of these options is ideal from an environmental perspective, as disposal requires a dumpsite as well as the effort of transportation, while on site stockpiling will lead to degradation of this material (Stronhmayer, 1999) and a greater land requirement. These common mine site development practices will require extensive use of heavy equipment that can have detrimental effects on the local environment through soil compaction (Kozlowski, 1999), air quality (Northwatch et al., 2008) and habitat degradation (Bayne et al., 2008) as well as globally in terms of greenhouse gas emission (MAC, 2003). Additionally, large mining operations established in remote regions requiring a live-in workforce would generate considerable wastes throughout its lifetime. Disposal of human generated wastes of mining operations may require the establishment of treatment facilities and landfills (Ripley et al., 1996).

Mineral extraction and processing will impact the local environment in a number of ways. The most readily observable impact is the creation of large voids where rocks have been extracted, and the subsequent dumping of unwanted waste rock (Lottermoser, 2003). This extensive movement of earth results in the disturbance and possibly the destruction of landforms and vegetation, and introduces potentially toxic wastes to the soils, atmosphere as well as surface and ground waters (Ripley et al., 1996). Surface mining operations must move much larger volumes of earth to access desired ore bodies as compared to underground operations (Lottermoser, 2003). In most of the world, underground mining operations account for more than half of the total mining activities. However in the United States, where coal mining dominates all other forms of mining, more than 90% of minerals are accessed are through surface operations (Dudka & Adriano, 1997). The excavated rock, which contains no economically valuable minerals, has traditionally been used for backfill and construction purposes when possible, (Ripley at al., 1996) but the majority of this mine waste will normally be dumped near to the mine, somewhere on the mine lease (Blowes et al., 2003). Physical and chemical properties of these wastes vary considerably based on the mineralogy and geochemistry of the parent rock material as well as the types of equipment used (Lottermoser, 2003). Mine spoil can range from boulder to clay sized particles and are typically low in all plant available nutrients as well as devoid of organic matter and any living, biological component (Lottermoser, 2003). As a result, mine spoils often resist natural recolonization by native plants, and may require considerable remediation to encourage plant cover (Ripley et al., 1996).

Waste rock and tailings exposed to the aerobic surface environment will undergo spontaneous weathering processes, which include acid producing, acid buffering as well as non-acid generating, or consuming reactions (Lottermoser, 2003). When waste rich in sulfide minerals are exposed to an aerobic environment, this change causes very rapid (as little as a few days) oxidation (Tremblay & Hogan, 2001), a process that can release two moles of acidity for each mole of sulfur reduced (Blowes & Ptacek, 1994). The relative proportion of acid neutralizing minerals, such as carbonates and silicates to acid generating minerals, most notably pyrite and pyrhotite (Blowes & Ptacek, 1994), as well as their structure, the particle size of the waste, oxygen and moisture content and presence of bacteria capable of catalyzing the oxidation processes determine whether a waste will generate acid or not (U.S. EPA, 1994). Local site characteristics such as location, climate, ecology, geology and hydrology dictate what environmental receptors are vulnerable to the effects of acid generation (Jennings et al., 2008). That being said, aquatic ecosystems are commonly affected when contaminated surface and sub-surface waters drain into connected waterways through a process termed acid mine drainage (AMD) (Jennings et al., 2008). Acid mine drainage is the single largest environmental liability of the mining industry, in Canada alone it is estimated between 2 – 5 billion dollars, and worldwide the liability due to acid generating mine wastes is worth upwards of 100$ billion US (Tremblay & Hogan, 2001). Gray (1997) identified metal toxicity, sedimentation, production of acidity, as well as salinization as the four main mechanisms through which AMD impacts lotic ecosystems.

Metallurgical processes, including smelting and refinement, release large quantities of gaseous and particulate matter wastes into the atmosphere (Ripley et al., 1996). Sulfur dioxide and other gaseous compounds, as well as metals such as arsenic, copper, cadmium, zinc, chromium, lead and nickel can be introduced to the atmosphere through metallurgical operations (Dudka & Adriano, 1997). These compounds can be transported great distances once airborne, and upon re-deposition through precipitation can lead to aquatic and terrestrial degradation (Ripley et al., 1996). Respectively, the Flin Flon and Thompson, MB operations have released 595 and 83 tones of heavy metals into the atmosphere on an annual basis (Northwatch et al., 2008). The most commonly observed effects of atmospheric deposition of mining wastes are soil and water acidification as well as contamination with heavy metals leading to decreased fertility, biological activity and overall productivity (Dudka & Adriano, 1997).

2.2.6 Mine tailings and their associated hazards:

Mine tailings are the most voluminous waste produced at most mining operations, and in many cases represent upwards of 99% of the rock that has been extracted (Lottermoser, 2003). Their physical and chemical properties vary from mine site to mine site due to differences in parent material mineralogy and geochemistry as well as the equipment and techniques used to process the ores. Due to their consistent mechanical processing, mine tailings tend to exhibit a fairly homogenous particle size, and most mine tailings dumps display a texture containing ~ 70-80% sand sized particle (Lottermoser, 2003). Further processing may include a variety of techniques depending on the target mineral and the matrix from which it must be separated. Additionally, processing techniques have changed considerably over the history of mining, constantly seeking more efficient and economically viable methods of extraction (Tordoff et al., 2000). Simple washing or sorting by gravity, magnetic, electrical or optical means are common techniques that an ore may undergo prior to treatment with process chemicals during mineral processing (Lottermoser, 2003). Process chemicals are classified based on a range of functions and include flotation reagents, modifiers such as pH regulators, activators and depressants, flocculants and coagulants, hydrometallurgical agents as well as oxidizers (Lottermoser, 2003). These functions are accomplished by a great array of chemicals including but not exclusively: organic and fatty acids, alcohols, xanthates, strong inorganic acids and bases, salts, clays, metal hydroxides and sulfates, polysaccharides and starch derivatives as well as cyanides and strong oxidizing agents (e.g. hydrogen peroxide) (Lottermoser, 2003). When mine tailings have been fully processed, they are most often pumped and dumped as slurry in natural or man-made impoundments near to the mill site, most often somewhere on the mining lease (Blowes et al., 2003). When left to dewater and dry, mine tailings may become susceptible to erosion to wind and water erosion leading to contamination of adjacent, undisturbed lands (Barbour, 1994).

When mine tailings are dumped over undisturbed landscapes, they suffocate the native vegetation and soils and will significantly impede the recolonization of the area by native organisms (Tordoff et al., 2000). The extent of the environmental degradation will depend on the characteristics of the tailings. However the immediate disturbance of productive land due to tailings dumping eliminates valuable ecosystem functions such as wildlife habitat, carbon sequestration as well as water and nutrient cycling (Ripley et al., 1996; Anielski & Wilson, 2002). Tailings rich in certain heavy metals or process chemicals pose a particularly great risk to the environment as precipitation events may leach these compounds through the waste piles and underlying soil strata and into the ground water aquifer (Barbour, 1994). Additionally, aerobically exposed mine wastes rich in sulfidic minerals such as pyrite (FeS_2) and pyrrhotite ($Fe_{x-1}S$), chalcopyrite ($CuFeS_2$), arsenopyrite (FeAsS), sphalerite (ZnS), galena (PbS) as well as many others, undergo spontaneous weathering reactions which can liberate large quantities of acidity as well as metals into solution (Blowes et al, 2003). This process, which has been termed acid mine drainage (AMD), can lead to acidification as well as heavy metal loading in adjacent water bodies and aquifers (Ripley, et al., 1996). Acidification of water bodies as well as soils can lead to a decrease in biological activity as well as diversity not only due to direct effects of pH on metabolic function, but because of the interactive effect of pH on metal speciation and availability (Jennings et al., 2000; Ripley et al., 1996). As a result, soils and water bodies affected by acidification in conjunction with metal loading can exhibit toxicity levels that prevent many native organisms from persisting in that habitat (Jennings et al., 2000).

While specific properties vary considerably from site to site, mine tailings show many physical and chemical properties that impede the establishment of plants and soil organisms (Lottermoser, 2003). The fairly homogenous texture and lack of organic matter and aggregation exhibited by mine tailings promotes compaction as well as poor surface structure, ultimately limiting water infiltration and retention (Ripley et al., 1996). These features leave mine tailings highly susceptible to erosion by surface waters and wind (Tordoff et al., 2000), and present a difficult substrate for the germination and establishment of plant seedlings. The most important chemical characteristic limiting plant establishment is the pH and acid generating potential of the tailings (Ripley et al., 1996). Acid generation can occur within days of mine tailings oxidation, or after a lag period of up to several decades (Tremblay & Hogan, 2010), and while excess acidity has deleterious direct effects on plant growth, interactive effects of pH on the availability of heavy metals, soluble salts, and mineral nutrients can present greater obstacles (Ripley et al., 1996). Most tailings exhibit extremely low levels of plant available macro and micronutrients (Ripley et al., 1996). This deficiency can be further exacerbated under extremes of pH, for example at moderately alkaline pH, phosphorous availability is diminished while under acidic conditions, the availability of nitrogen, potassium, calcium, magnesium as well as phosphorous is decreased (Brady & Weil, 2008). Additionally, many micronutrients essential for plant growth in small quantities become significantly more available under acidic conditions, leading to potential toxicity (Brady & Weil, 2008). On the other hand, high levels of non-essential elements (heavy metals) as well as other harmful process chemical pose issues of toxicity if present at significantly elevated levels; however great variation exists among plant species in terms of their tolerances (Tordoff et al., 2000). Finally, strong acidity can lead to the liberation of soluble salts during weathering of mineral fractions, potentially imposing direct ionic stresses as well as osmotic stress (Ripley et al., 1996).

2.2.7 Regulatory framework for mining and mine site reclamation in Canada

In Canada, all mineral (or sub-surface) rights are property of the Crown, regardless of the surface land ownership situation (Northwatch et al., 2008). Provincially, the Crown refers to the provincial government while for the Yukon and Northwest Territories the Crown refers to the federal government. For over 150 years, Canada has employed an antiquated system for mineral claim staking called 'free-entry' which is based upon the premise that mining is the best possible use of Crown land (Northwatch et al., 2008). The Free Entry system developed in the 1500s, was used in feudal England as well as during the colonization of North America, and allows mineral claims to be staked without consideration for other values or potential land uses (Northwatch et al., 2008). Largely as a result of this system, in 2006, almost 57 million hectares, (approximately 5.7% of Canada's total land area) had been staked for mineral claim, including over 12% of Alberta and 11.4% of Saskatchewan (Northwatch et al., 2008). This system differs considerably from the traditional application and review procedures that must be taken to establish forestry, agriculture or transportation projects, and has resulted in considerable conflicts between surface and sub-surface interests (Canadian Boreal Initiative, 2008). The environmental consequences of this Free Entry system is also considerable; sensitive ecological features as well as natural values were completely unprotected during early mine development (Northwatch et al., 2008). Additionally, the past performance of mine operators is not considered under this system, allowing for operators with track records of environmentally destructive practices to be treated like any other potential prospector (Northwatch et al., 2008).

Federal and provincial regulatory legislations control mining activities as well as any associated impacts to land, air and water quality, with provincial level jurisdiction deliberating on the majority of mining issues (Northwatch et al., 2008). That being said, the federal government retains jurisdiction over ocean and inland fisheries, navigable waterways, criminal law, inter-provincial trade and commerce, national park establishment, uranium mining, as well as aboriginal land issues (Northwatch et al., 2008).

At the federal level, the Canadian Environmental Protection Act (CEPA), the Canadian Environmental Assessment Act (CEAA) as well as the Fisheries Act are the main legislature ensuring mine wastes are appropriately treated. The Canadian Environmental Protection Act aims to protect air, water and land from harmful substances and ensure the highest overall levels of environmental quality. This act sets limits for emission of toxins such as mercury, as well as any smelter emissions (Northwatch et al., 2008). The Canadian Environmental Assessment Act establishes the framework for environmentally responsible planning, and though provincial and federal environmental assessments may not be legally binding, they are often an important pre-requisite for granting a mine lease (Northwatch et al., 2008). The Fisheries Act legislates for the conservation and protection of fish habitat, and is therefore concerned with the various ways mining operations could negatively affect aquatic ecosystems. More specifically, the Fisheries Act's Metal Mining

Effluent Regulations (MMER) establishes limits for nine pollutants in metal mine effluent in accordance with the "Best Available Technology Economically Available" approach (Northwatch et al., 2008). Provisions within the act, such as the 'no net loss' for fish habitat, aim to at very least mitigate the inevitable impacts to aquatic habitat (SCNR, 1996), which are extensive and can include draining lakes and diverting rivers (Wells et al., 2010), changes in rate and volume of water flow (Pentreath, 1994), increased turbidity and sedimentation (Northwatch et al., 2008), as well as water quality degradation (Barbour, 1994). Natural Resources Canada (NRC) is an important federal contributor with secretariats supporting initiatives such as Mine Environmental Neutral Drainage (MEND), which has produced more than 200 technical documents relating to acid mine drainage mitigation (Tremblay & Hogan, 2001), as well as the National Abandoned/Orphaned Mines Initiative (NAOMI). Environment Canada, the Department of Fisheries and Oceans (DFO), the Mining Association of Canada, Mines Ministers as well as the Canadian Committee of Ministers of Environment (CCME) are other important contributors to the federal level of influence (Northwatch et al., 2008). The CCME is a national council consisting of federal, provincial and territorial ministers of the environment whose chief role is to develop consistent environmental objectives and standards suitable for use as nationwide guidelines. Some of the valuable guidance documents generated by the CCME include: Canadian Environmental Quality Guidelines, the National Classification System for Contaminated Sites, A Framework for Ecological Risk Assessment: General Guidance, as well as Canadian Drinking Water Quality Guidelines.

At the provincial level, here in Manitoba, the Mines and Minerals Act provides the framework for all mining activities (Government of Manitoba, 2008), while remediation of disturbed lands is regulated by four main acts. The Environment Act is an overarching statue for environmental compliance; the Dangerous Goods Handling and Transportation Act governs the handling, transport and disposal of harmful wastes; the Contaminated Sites Remediation Act seeks to decrease future environmental risks and remediated current contaminated sites; and finally the Water Resources Conservation and Protection Act is concerned with the maintenance of water resource quality. In 1999, an important new regulation was instituted to the Mines and Mineral Act entitled the Mines Closure Regulation (Northwatch et al., 2008). This regulation not only requires mine operators to develop an appropriate mine closure plan prior to receiving operating permit, but it also holds the operator liable, and requires financial surety, for any and all environmental remediation following mine closure (Hart et al., 2012).

Given the vast territory and varied geology within Canada, it is not surprising that it has long been the world's largest and most diverse mineral exporter (Wallace, 1996). Furthermore, the tremendous economic input from this industry on the national, provincial and local levels (Northwatch et al., 2008) in addition to the promising future of mineral exploration and mining in Canada (Cranstone, 2002) suggest that this industry will remain important going forward. That being the case, sustainable practices must be employed throughout the mining process to minimize conflicts that arise due to interacting interests (e.g. mine operators, stakeholders, local and indigenous communities as well as the public at large). Hilson and Murck

(2000) present some important guidelines for encouraging sustainable development in the mining industry, these were: improved planning, environmental, and waste management (including the implementation of cleaner technologies); as well as to emphasize training programs, to address the needs of communities and stakeholders, as well as to develop sustainable partnerships. It is therefore the role of both mine operators as well as provincial and federal governments (acting with direction from public interest as well as educated agencies, associations and committees) to ensure that mining throughout Canada progresses in a fashion that minimizes impacts to the environment and conflicts between human and industrial land usage.

2.3 Current approaches to mine tailings reclamation

2.3.1 Tailings characterization

Mine tailings can exhibit tremendous variation in terms of their physical characteristics, mineralogy and geochemistry and as a result vary considerably in terms of their environmental hazard (Ripley et al., 1996). The four main environmental concerns related to mine tailings are the loss of ecologically valuable land to tailings disposal and management, water quality and aquatic ecosystem degradation, as well as air quality impacts, primarily through dust emission (Northwatch et al., 2008). Given the site-specific nature of mine tailings and other wastes (Bradshaw et al., 1978; Pentreath, 1994), detailed characterization of the mine wastes is a critical first step in the development of a remediation plan. Remediation options can range from a completely passive treatment, when the wastes pose no environmental risk and can naturally revert to a vegetated state within a reasonable timeframe, to extremely complex and intensive treatments where wastes are isolated and any potential leachates are actively treated (Ripley et al, 1996; Johnson & Bradshaw, 1977). Table 2.1 presents various properties of mine wastes that will affect its environmental hazard (acid generation and erosive potential) as well as its capacity for revegetation.

Of primary importance when describing a mine waste in terms of its potential environmental impact is its capacity to generate acid as well as potential pathways through which acidic or heavy metal laden discharge can enter adjacent waterways and ecosystems (U.S. EPA, 1994; Pentreath, 1994; Ripley et al., 1996). The capacity of a mine waste to generate acid is related to the quantities of acid producing sulfidic minerals (primarily pyrites and pyrrhotites) (Blowes et al., 2003) and acid neutralizing minerals (largely carbonates and silicates) (Lottermoser, 2003), as well as levels of residual contaminants (U.S. EPA, 1994). Factors that contribute to the rate of acid generation include the type of sulfide and carbonate minerals (as well as their form), the available surface area (related to both particle size and spatial occurrences of grains), the availability of water and oxygen, temperature, as well as the abundance of oxidizing bacteria (e.g. *Thiobacillus ferrooxidans*) that are capable of enhancing the rate of oxidation (U.S. EPA, 1994; Lottermoser, 2003).

Physical and geochemical properties of the mine wastes, individually and interactively, present impediments to natural revegetation; in order to establish a suitable environment for a sustainable plant cover, these factors must be carefully

Table 2.1. Important properties to consider during mine waste characterization.

Concern:	Characteristic:
Acid Generation	
Total Capacity	Amount and type of sulfide minerals
	Amount and type of carbonate minerals
	Volume of waste, settling and layering
	Leaching properties
Rate of Generation	Particle size
	Mineral form
	Presence of catalyzing bacteria
	Oxygen and moisture content
	Temperature
Erosive potential	Particle size distribution
	Climate and position of water table
	Surface characteristics
	Topography
Capacity for Revegetation	
Physical Properties	Texture and gravel content
	Bulk density and porosity
	Rate of infiltration
	Hydraulic conductivity
	Moisture retention
	Structure and aggregation
	Colour and thermal regime
	Plasticity, consolidation behavior
	Crusting and soil strength
	Compaction
	Erosion potential
Geochemical and Mineralogical Properties	pH
	Electrical conductivity
	Cation exchange capacity
	Base saturation
	Organic carbon content
	Available nutrients
	Metal contents
	Carbon to nitrogen ratio (C:N)
	Clay minerology

considered (Johnson et al., 1994; Tordoff et al., 2000). One of the most important factors affecting the hydrology of mine wastes dumps is their particle size distribution, and texture (Smith & Beckie, 2003). Mine tailings exhibit a fairlyhomogenous particle size due to their consistent mechanical processing (Lottermoser, 2003); however both within and between mine tailings dumps considerable variation can be witnessed (Winterhalder, 1995). Since fresh mine tailings are pumped as a slurry (typically with ~ 20 - 40% solid)(Lottermoser, 2003), upon deposition, settling patterns are dictated largely by particle size; with finer particles being transported a greater distance from the discharge point than larger particles (Blowes et al., 2003). As a result, mine tailings dumps often contain distinct zones where differences in texture and mineralogy favor conditions conducive to the establishment of markedly different communities (Winterhalder, 1995). Infiltration, hydraulic conductivity and water holding capacity are all directly influenced by the texture and rock fragment content of mine wastes (Dollhopf & Postle, 1988). Coarse grained mine tailings allow rapid infiltration and drainage, while fine textured tailings limit water infiltration and hold moisture with greater force than coarser substrates (Ripley et al., 1996). Under local climatic factors, coarse-grained tailings may be susceptible to rapid surface drying, and fine-grained tailings may hold water at the surface leading to surface runoff and gulley erosion (Bradshaw et al., 1978; Ripley et al., 1996). In addition, fine textured soils (with greater proportion of clay minerals) exhibit a greater capacity to hold not only water, but also mineral nutrients, retaining valuable plant resources within the rooting substrate (Ripley et al, 1996). The ability of a soil to bind, store and release mineral nutrients (measured as cation exchange capacity or base saturation) are related to the proportion of clay minerals and organic matter within the substrate (Brady & Weil, 2008). Given the lack of an organic component in fresh, as well as in many historic mine tailings that have resisted re-colonization, the proportion of clay minerals within the dump can have significant influence on the overall fertility of the site (Johnson et al., 1994; Ripley et al., 1996).

Geochemical properties of mine tailings often have the greatest influence on their suitability for revegetation, and these characteristics vary considerably from site to site (Bradshaw et al., 1978; Ripley et al., 1996). The most important features relate to heavy metal as well as nutrient availability; however due to the interrelated nature of both of these characteristics with pH, they must be discussed with specific reference to the ambient pH. Mineral ores, and their resultant tailings are, by nature, enriched with some metallic minerals. Additionally, untargeted metals, which may include zinc, lead, iron, copper and arsenic may be liberated during the weathering of exposed mine tailings (U.S. EPA, 1994). The techniques used to extract the desired commodity from the parent rock have changed over time leading to more efficient methods of recovery which generate a waste with less residual levels of that metal than ancestral techniques would generate (Johnson et al., 1994). All heavy metals have the potential to induce symptoms of toxicity in plants and soil organisms, though the degree of hazard varies greatly depending on the metal. The level at which these symptoms are displayed varies between species, between tissues, as well as in relation to other factors, including developmental stage and nutritional status (Foy et al., 1978; Fageria et al., 2011). While specific symptoms vary from metal to metal, common mechanisms through which metals affect plant growth include negative effects on transpiration, photosynthesis, respiration and nodulation, interference with cell division in roots, enzymatic function, and the uptake, transport and use of various important elements (Foy et al., 1978). Furthermore, synergistic and antagonistic interactions may occur between different ions, which may result in a greater observed level of toxicity that would have been expected given the individual components (Tordoff et al., 2000). For example, in several plant types it has been shown that increased zinc levels in the soil caused an increase in translocation of manganese to foliar tissues, potentially leading to manganese toxicity (Foy et al., 1978). Alternatively, some available metals will compete with desirable mineral nutrients for plant uptake. For example, at low pH the trivalent form of aluminum directly competes with calcium for uptake potentially leading to nutrient deficiencies (Meriño-Gergichevich et al., 2010). While mine tailings may be rich in some micronutrients (such as iron, manganese, copper, zinc or boron), most macronutrients including nitrogen, phosphorous, potassium, calcium and magnesium are generally lacking in sufficient quantities (Bradshaw et al., 1978; Ripley et al., 1996). Further exacerbating the nutrient deficient nature of mine tailings is the general lack of organic matter and the low proportion of clay minerals (Johnson & Bradshaw, 1977; Tordoff et al., 2000). The organic and clay fractions, together account for the great majority of active ion exchange sites in a soil, a factor which allows nutrients to resist leaching and be held in the rooting zone (Brady & Weil, 2008). Lacking adequate nutrient holding capacity, inorganic fertilizers applied to mine tailings are susceptible to rapid loss through leaching and will require regular reapplication to sustain vegetative cover (Johnson et al., 1977). pH affects the speciation and the solubility of metallic compounds as well as important mineral nutrients (Ripley et al., 1996). Under acidic conditions, metals including iron, lead, copper and zinc become considerably more available both through increased weathering of metallic minerals as well as interactions of pH with metal speciation and solubility (Ripley et al., 1996; Caruccio et al., 1988). Under these conditions, the availability of all macronutrients as well as a couple micronutrients (molybdenum and boron) is significantly curtailed (Brady & Weil, 2008). On the other hand, under alkaline conditions, the uptake of some nutrients may be restricted due to the formation of hydroxide, carbonate, and calcium – heavy metal complexes (Bradshaw et al., 1978).

The mine tailings surface can present a particularly inhospitable environment for the establishment of a plant community given the issues of erosion, surface compaction as well as exposure (Bradshaw et al., 1978; Ripley et al., 1996). The large area and flat topography characteristic of mine tailings dumps in addition to the fine, un-aggregated nature of mine tailings contribute to their susceptibility to wind erosion (Bradshaw et al., 1978; Johnson et al., 1994). Low levels of continual water erosion are common in mine tailings dumps as a result of poor infiltration rates associated with a dense substrate lacking organic matter and aggregation (Bradshaw et al., 1978; Ripley et al., 1996). Aggregation is influenced by physiochemical processes relating to clay mineralogy and organic matter content as well as biological processes mediated by plant roots, fungal hyphae and soil organisms (Brady & Weil, 2008). Level of aggregation as well as aggregate stability are extremely important soil characteristics because of their effect on surface hydrology, erosive potential, as well as microbial habitat (Ripley et al., 1996; Chenu & Cosentino, 2011). The stable aggregation of soil particles generates structures far less easily transported by wind and water erosion due their increased size and mass. Also, the aggregation of surface particles creates pore space both within and between aggregates through which water and oxygen transmission can occur. This improved surface structure results in an improved rate of infiltration and, consequently, less surface run-off and erosion (Brady & Weil, 2008). The erosion of mine tailings will not only has negative environmental impacts on the receiving area, but also hinders plant establishment and soil development for the area of loss (Ripley et al., 1996; Tordoff et al., 2000). The fresh exposure of tailings to the surface may result in increased oxidation of sulfidic minerals, potentially liberating additional quantities of metals and acidity (Johnson et al., 1977). Surface erosion results in the removal of fine organic matter as well as seeds, spores and other important biological components and may result in damage to plants through sandblasting, burying as well as undermining (Ripley et al., 1996). Surface characteristics (e.g. roughness) as well as topographical features (e.g. aspect) are significant contributing factors on the erosive potential of a substrate (Brady & Weil, 2008), and can have a determinant influence on future land use options (Ripley et al., 1996). The uniform texture and lack of aggregation at the tailings surface leaves them vulnerable to compaction, especially from equipment traffic, establishing a harsh mechanical impediment to plant roots as well as soil organisms. Additionally, under certain geochemical and climatic conditions, soluble salts may concentrate at or near the tailings surface, leading to ionic and osmotic issues as well as possible surface crusting (Lottermoser, 2003). Finally, the tailings surface is directly exposed to atmospheric conditions, and depending on local climate, the inhabiting plant community may be exposed to extremes of radiation, temperature, moisture and wind (Ripley et al., 1996).

Following the exhaustive description of the mine waste and their local setting, remediation goals can be set. These goals will depend upon the desired end land use of the impacted property as well as the immediacy of the environmental or human health risk (Bradshaw & Johnson, 1992; Johnson et al., 1994). In order to return drastically disturbed lands to a healthy ecological state, the physical and geochemical characteristics of the mine tailings must be carefully considered with regard to biological inhabitability both on short and long-term timelines. Many of the factors discussed can act individually, or in concert under the local climate to create limit or entirely impede the establishment of native vegetation and wildlife from the area for centuries (Mendez & Mainer, 2008). Careful and complete tailings characterization is the key to implementing successful remediation plans that will mitigate the environmental impacts and potentially return some ecological or environmental function to the disturbed area.

2.3.2 Cap and cover systems (avoidance)

When it is determined that the mine waste exhibits properties that are likely to preclude the establishment of a plant cover, or when a site must be rapidly reclaimed to mitigate some immediate hazard (generally acid drainage or the associated metal loading), capping or covering the waste may represent the most effective solution (Johnson et al., 1994). Since cover systems are very often employed in an attempt to mitigate acid drainage, the most effective mechanism by which this can be accomplished is by limiting the influx of water and oxygen (Huag & Pauls, 2001). Many techniques have been developed to accomplish these functions including both simple and complex systems employing wet and dry covers (Tremblay & Hogan, 2009). Covers are used to isolate toxic materials, to establish barriers or breaks to water and oxygen, as store and release reservoirs of sub-surface water, and even as insulation in cold environments where wastes are kept in a frozen state (Rykaart & Hockley, 2009).

Water provides an inexpensive capping substance that creates an effective barrier to oxygen, and as a result, sub-aqueous disposal of tailings is a relatively common practice in Canada. In Canada's boreal forest, fifteen natural bodies of water are being used for tailings disposal, with five having been approved between 2002-2009 (Wells et al., 2010). While the concentration of oxygen in water is temperature dependant, the maximal oxygen concentration in water is three orders of magnitude smaller than the maximal atmospheric concentration (Lottermoser, 2003). Research has been conducted in Manitoba where two northern lakes (Anderson Lake in the Snow Lake region and Schist Lake near Flin Flon) have been monitored to determine the effect of sub-aqueous tailings disposal (Rescan 1990). At these sites, as well as many others, encouraging results have been obtained. According to Sly (1996), while the short-term effect of sub-aqueous tailings disposal may be unfavorable, due to the long-term stability of the waste in this condition, this disposal technique warrants consideration. The ability of water to limit oxidation is exploited in many cover systems, through the flooding of a tailings impoundments, by artificially raising the water table, as well as by dumping sulfidic wastes below the water table to keep the potential oxidizing materials submerged (Northwatch et al., 2008; Lottermoser, 2003; SENES, 1996). Problems that can arise when employing water covers relate to the difficulty in maintaining effective water coverage over time, as well as maintaining the integrity of the containment structures (Tremblay & Hogan, 2009).

Dry cover systems are also employed as a means to limit oxygen influx as well as moisture infiltration through sulfide rich mine wastes. However they may serve an additional function as rooting medium for overlying vegetation (Tremblay & Hogan, 2009). Simple cap or cover systems are generally composed of a single layer of material placed atop the mine wastes, designed to separate the toxic material from the active, surface environment (Johnson et al., 1994). Cover materials may be characterized as ameliorative (providing for a suitable rooting medium), or inert (Tordoff et al., 2000). Inert, coarse textured material can be used as a simple capping material when the intent is to limit erosion of fine wastes, to isolate particularly toxic materials, or to create a hydraulic break that would limit upward migration of contaminated tailings water (Bradshaw & Johnson, 1992; Johnson et al., 1994). Examples of inert covers include readily available materials such as mine wastes (e.g. low-sulfide and oxide waste rock, non-acid generating tailings), as well as epoxy resins, chemical caps, fly ash mixes, asphalt, cementitious materials, wax barriers, synthetic materials and modified soils (Huag & Pauls, 2001;

Lotermosser, 2003). If the purpose of the cover is ameliorative, it is most often some form of soil (stockpiled or borrowed top soil, sub-soil or peat) or another organic substitute (sewage or other organic sludge, manures, mulches)(Johnson & Bradshaw, 1976; Ripley et al., 1996). In order to provide adequate rooting volume for selected cover plants, it is recommended to place a minimum of 30 cm of rooting medium as simple cover (Johnson et al., 1994). However, depending on the end land use, suitable rooting depths greater of 120 cm may be recommended (Burger et al., 2005). For example, during the revegetation of copper mine tailings in Ontario, papermill sludge was applied as a cover of two different thickness (15 cm and 50 cm) in a large-scale field experiment. The authors found that the thicker cover maintained a neutral pH and allowed for increasing crop yield over the three year trial period while the thinner cover underwent acidification as well as a loss of fertility (measured as cation exchange capacity) leading to decreasing crop yields over the course of the experiment (Okonski et al., 2003). This type of system provides an improved rooting medium for the introduction of plant cover; however, it does not isolate the plant community from the potentially toxic substance underneath the surface strata nor does it establish a hydraulic break to water within the waste. As a result, problems with this system have arisen due to contamination of the cover by upward migration of toxic metals as well as symptoms of toxicity when plant roots penetrate into the underlying waste (Johnson et al., 1994).

Compound cover systems employ multiple materials to accomplish many functions including, but not exclusively: encapsulation or isolation, hydraulic or oxidative barriers and breaks, to function as reservoirs, as well as to provide drainage or a suitable rooting medium (Wilson, 2003; Rykaart & Hockley, 2009). Low hydraulic conductivity materials (including compacted clay, bentonite modified soils, synthetic liners and covers as well as geo-membranes) are used as barriers or isolating layers, while materials with high conductivities (coarse aggregate, graded non-acid generating mine wastes, as well as topsoil or other organic rich materials) are used for drainage or rooting purposes (Johnson et al., 1994; Huag & Pauls, 2001). The successful design of a mine waste cover system depends on site-specific atmospheric factors (i.e. precipitation and evaporation) as well as soil characteristics (primarily those relating to infiltration and hydraulic conductivity), as well as plant and animal effects (such as root growth or burrowing)(Wilson, 2003). Common design of a compound cover system will include a low conductivity or impervious layer directly above the mine waste, a thick (30-300 mm) layer of inert, granular material above that, as well as a suitable rooting medium (> 20cm) at the soil surface (Lottermoser, 2003; CANMET, 1999). The low conductivity material limits the ingress of water and oxygen through the waste while the coarse material creates a capillary break between layers, ensuring that water-soluble waste products do not migrate upwards during water table fluctuations. As a result, the superficial rooting medium remains uncontaminated by the potentially harmful substances present at depths (Bradshaw & Johnson, 1992). The desired end land use must be considered when selecting your rooting substrate as the quality and quantity of the rooting medium will have a great influence on the composition and health of the plant community (Jansen & Melsted, 1988). For example, cereals and forbes may prefer finer textured soils while trees and shrubs are more suited to coarser soils (Ripley et al., 1996). Additionally, the nutrient requirements provided by the soil substrate differ markedly between agricultural and forest plant species. The Forestry Reclamation Approach (FRA) was established by the Appalachian Regional Reforestation Initiative, and provides guidelines for the establishment of productive managed forest stands on reclaimed mine spoil lands. Under this approach, it is recommended to provide at minimum ~120 cm of rooting medium, which includes an active soil component which is loosely graded at the surface to allow water infiltration, root penetration and to limit erosion (Burger et al., 2005). Additional recommendations for reclamation to this land use include the use of a low nitrogen, high phosphorous content fertilizer, as well as seeding a tree compatible ground cover in order to limit resource competition (Burger et al., 2009). Regardless of the desired land use or the rooting medium, the overall goal of a revegetation effort should be to establish a self-sustaining nutrient pool sufficiently large to fulfill the nutrient demand of the vegetative community in perpetuity (Ripley et al., 1996).

2.3.3 Tailings amendment (ameliorative)

While some mine tailings and wastes require elaborate measures to isolate and mitigate their potential liability, depending on site-specific characteristics, others may have the potential to be revegetated with minimal amelioration (Ripley et al, 1996). Techniques exist for the treatment of the various constraints established by the mine tailings environment, which include elevated levels of heavy metals, extremes of pH, lack of available nutrients, soil structure and biota, as well compaction and water availability issues (Tordoff et al., 2000). Given the site-specific nature of mine wastes, no single amendment technique is universally applicable. The edaphic conditions of the spoil as well as the regional climate and the desired land use must be considered in conjunction in order to develop amendment strategies (Johnson et al., 1994). Mine wastes have been successfully amended to house all types of land uses, including both managed and unmanaged systems (Johnson & Bradshaw 1977; Kramer et al, 2000; Reid & Naeth, 2005b).

Among the most common limitation to plant growth exhibited by mine tailings is the drastic lack of available mineral nutrients. Additionally, without active ion exchange sites provided by the clay and organic fractions of a soil substrate, mine tailings are not capable of holding nutrients as they become available. Mine tailings are often amended with inorganic fertilizers of all types, however given their inability to bind and hold nutrients in the active rooting zone, much of what is applied is lost and regular re-fertilization may be required to maintain plant cover (Johnson & Bradshaw, 1977; Tordoff et al, 2000). The application or incorporation of organic matter to the tailings surface can greatly improve fertility during their gradual decomposition as well as through the creation of ion exchange sites. Additionally, this input of organic matter provides a resource for and a source of soil biota including bacteria, fungi as well as invertebrates capable of mineralizing the organic matter into plant available nutrients (Ripley et al., 1996; Barton & Northup, 2011). The growth of the plant community as well as overall soil fertility is dependent upon the presence of a healthy soil community. Soil biota accomplishes and facilitates a host of roles critical to plant nutrition, including nutrient and water foraging, uptake and transmission, atmospheric nitrogen fixation as well as organic matter decomposition and mineralization (Tate, 2000). Inoculation of plant materials or soil amendments with beneficial microbes including mycorrhizal fungi or nitrogen fixing bacteria can ensure that these organisms are present and their important functions can be accomplished. Among the most important characteristics relating to the nutrient status of an organic amendment is its relative proportion of carbon to nitrogen (C: N)(Vagstad et al., 2001). Materials with a carbon to nitrogen ratio greater than ~25:1 will likely lead to net immobilization of any available nitrogen in microbial biomass, while materials with a lower ratio will lead to net mineralization of nitrogen (Brady & Weil, 2008). As a result, materials with extremely high ratios (e.g. straw, peat, wood chips and bark) are generally not suited to the function of improving nutrient availability on the short term, while materials with low C: N ratios (e.g. animal manures, composted materials, as well as some municipal and industrial sludge) can prove very effective.

Organic amendments not only have the potential to provide a nutrient source for plants and soil organisms, but also can also greatly improve aeration, water infiltration and retention, as well as seedbed characteristics (Johnson et al., 1994). Whether applied as mulch or incorporated into the tailings surface, organic materials have great capacity to absorb and store water. Peatmoss for example has a remarkable water absorption capability, able to hold between 20-30 times its weight in water (Yoshikawa et al., 2004). Additionally, organic matter addition will inevitably roughen the surface and improve tilth (Jansen & Melsted, 1988), increasing water infiltration (Smith & Beckie, 2003) and leading to decreased erosion, increased moisture retention as well as an improved capacity to catch windblown propagules and organic matter (Burger et al., 2009). Organic matter is also an important component driving the formation of stable soil aggregates, a characteristic with significant effects on water transmission as well as resistance to compaction and erosion (Lyle, 1987). Inorganic materials such as coarse, inert waste rock can similarly be used to improve water transmission characteristics. Co-disposal of coarse materials with fine grade tailings may establish a texture more conducive to normal drainage patterns, or coarse materials may be employed in the development of drainage features (e.g. toe drain)(Lottermoser, 2003).

Residual levels of potentially toxic heavy metals, extremes in pH, as well as the interaction of these two characteristics often presents the major chemical impediment to plant establishment on mine tailings (Tordoff et al., 2000). Optimal soil pH for normal plant growth is in the range of 5.5-7.5, with considerable variation between species as well as genotypes within a species (Brady & Weil, 2008). While some tailings can exhibit alkaline pH, it is far more common to exhibit strongly acidic pH as a result of the oxidation of sulfide minerals (Lottermoser, 2003). Given that metal availability is largely a function of pH, treating the pH of mine tailings may be sufficient to reduce metal availability to acceptable levels (Ripley et al., 1996). Treatment of acidic mine tailings with liming agents such as ground limestone, fly ash, and calcium carbonate have been extensively employed throughout the world for decades in order to raise pH and buffer the system against further acidification (Johnson et al., 1994, Ripley et al., 1996). By raising the pH and buffering the system, the rate of mineral weathering is reduced and fewer metallic ions are released into available forms (Price, 2003). Another technique that has been extensively used in order to mitigate the impediment established by excessive heavy metal availability is the binding of metals through organic amendment (Calace et al., 2011). Given their high capacity for ion exchange, organic materials (e.g. paper mill sludge) can bind and retain heavy metals, thereby reducing their availability for plant uptake (Calace et al., 2011). While this absorption alleviates some phytotoxicity on the short term, during decomposition these metals may be released back into the soil, leading to regression in plant performance over time (Bradshaw et al, 1978).

Tailings amendment for revegetation is a technique to mitigate, or at very least dilute the constraints for plant establishment, as opposed to cover systems that aim to avoid these issues all together (Johnson et al., 1994). While the method of application (surface application, incorporation, or sub-surface injection), as well as the type of amendment varies from site to site, cost is always an important factor dictating what remediation options will be employed. As a result, it is common practice to attempt to recycle materials generated through mine site operation, or materials easily procured at low cost in the amendment process (Johnson & Bradshaw, 1977; Tordoff et al., 2000). Examples of this include the use of inert waste rock (such as non-pyritic colliery shale, slate quarry waste or limestone chippings), the use of sediment from drained water bodies, stockpiled topsoil or sub-soils, peat or other vegetation cleared during development, as well as fly ash from colliery processes (Johnson & Bradshaw, 1977; Pichel et al., 1994; Tordoff et al., 2000; Rensburg & Morgenthal, 2004; Reid & Naeth, 2005).

2.3.4 Revegetation and phytostabilization

Regardless of the level of effort required ameliorating the tailings substrate, among the most favorable strategies to minimize the impact of the mine tailings on the local and surrounding environment is to re-introduce plant cover (Mendez & Maier, 2008). The establishment of plant cover, particularly one that is in synchrony with the overall vegetative community and ecological landscape, will provide many functions; the most readily visible is the improvement to the overall environmental aesthetic. More importantly, through a process termed phytostabilization, plant cover can greatly reduce the susceptibility of these tailings to unwanted migration by wind and water erosion (Mendez & Maier, 2008). Plant root systems will contribute to a decreased erosive potential in a number of ways. First, root systems can provide considerable soil reinforcement simply as a result of their architecture and spatial extension (Pierret et al., 2011). Additionally, organic inputs to the rhizosphere in the forms of root tissue turnover, root and soil microbe exudates as well as decomposing litter provide positive feedback on soil-plant fertility and act as a binding agent thereby improving soil aggregation. Larger soil aggregates are less easily eroded, and as soil aggregation increases and soil structure improves, water infiltration should occur more rapidly leading to decreased surface water erosion (Brady & Weil, 2008). Furthermore, networks of fungal hyphae sent out from mycorrhizal roots provide an additional level of mechanical binding, as their fibrous systems of hyphae are many times finer and more extensive than their host roots (Chenu&Cosentino, 2011). Above ground plant tissues also contribute to decreased erosion of soil particles in a number of ways. When established in a particularly exposed area, low growing plants will serve windbreak and sediment catching functions. As plants turnover above ground tissues, the organic matter deposition will shelter the soil surface from exposure to direct sunlight and rainfall, helping to protect the surface structure integrity (Lyle, 1987).

Beyond the role that plant cover plays in stabilizing the primary particles of mine tailings, they may also protect ground water aquifers from potential acidification or heavy metal contamination (Johnson & Bradshaw, 1977). The downward leaching of potentially harmful substances from mine tailings dumps is accelerated by water infiltration and percolation of water through the tailings. By establishing a healthy plant cover overlaying the mine waste, surface water inputs can be intercepted, utilized and returned to the atmosphere through transpiration without having the chance to percolate through the contaminated waste. In temperate regions, evapotranspiration by plant cover can return approximately 50% of the precipitation to the atmosphere, alternatively, for wet tropical regions the proportion is less than 25% and for dry tropical regions it can be greater than 75% (Johnson et al., 1994). Considering the boreal region, Black et al., (1996) found that an aspen stand in central Saskatchewan was retuning approximately 88% of the annual precipitation to the atmosphere through evapotranspiration. Conversely, colder regions of the boreal biome that are far less vegetated (e.g. taiga ecozones) would express far lower levels of transpirational return. Additionally, certain plant species as well as organic matter deposited by the cover community can absorb potentially harmful substances present near the soil surface thereby reducing their leaching potential and bioavailability (Mendez & Mainer, 2008; Calace et al., 2011).

2.3.5 Plant species selection and planting techniques

In addition to identifying and improving the site and substrate characteristics that are likely to impede plant growth, the careful selection of plant species and method of planting is key to the success of a revegetation effort (Johnson et al., 1994). Whether plants are introduced directly to the mine waste surface, or to a modified soil substrate designed for rooting (i.e. a cover system), plants often employed during revegetation can be described as agronomic, native, as well as metal and otherwise tolerant populations (Tordoff et al., 2000). Furthermore, the method by which these selected plants are seeded or planted can significantly affect the establishment as well as survival and overall health of the plant populations (Johnson & Bradshaw, 1977; Lyle, 1987).

Agronomic plant species have generally developed through agricultural type selective breeding processes and are chosen based on desirable characteristics for a reclamation objective at a specific site (Bell & Meidinger, 1977). Most commonly agronomic plant species are herbaceous (grasses and forbs are common components of reclamation plans) and alien to the area where they are being planted (Brink, 1980; Skousen & Zipper, 1996). Seed mixes often include fast growing grass species to establish plant cover in addition to leguminous plants capable of fixing atmospheric nitrogen in order to enhance soil fertility and reduce the need for fertilization (Skousen & Zipper, 1996). Some of the advantages of using agronomic plant species for tailings revegetation include the availability of large quantities of seed of known (and reliably high) viability, the rapid establishment of ground cover, high biomass production (both shoot and root tissues), well established fertility requirements, as well as the low overall cost of seed and seeding techniques (Lea, 1982; Ripley et al., 1996). The rapid establishment of a plant cover can help to stabilize the tailings and reduce their erosive movement off-site while kick-starting ecosystem functions such as organic matter buildup and carbon sequestration, atmospheric nitrogen fixation, as well as habitat for many soil and terrestrial organisms (Ripley et al., 1996). Ideally, this pioneering vegetation will naturally succeed to the next natural seral stage community once edaphic conditions have been sufficiently ameliorated. However, given that agronomic plant species are bred under agricultural conditions, they often require continual inputs (in the form of fertilization, re-seeding, or top dressing with organics) in order to maintain a healthy cover over time (Bell & Meidinger, 1977; Johnson & Bradshaw, 1977). Additionally, introduced plant populations may not be capable of surviving infrequent local disturbance events such as pests, disease, drought or extremes of temperature as a result of their genotypic homogeneity and lack of evolved mechanism to establish under the local conditions (Brink, 1980). Established swards of agronomic plants may also hinder the establishment of other native plant species attempting to invade the reclaimed area (Groninger et al., 2007). When a dense cover is established under conditions that favor the growth of agronomic plant species (i.e. with agricultural inputs), the level of competition for light, moisture as well as mineral nutrients may be sufficiently high to exclude many native plants (Burger et al., 2010). The establishment of a self-sustaining sward of dense agronomics may in fact resist further vegetative succession, resulting in a type of arrested succession that may persist for decades (Groninger et al., 2007).

Furthermore, herbaceous vegetation represents favorable habitat for burrowing animals, such as mice and voles, which may hinder shrub and tree establishment through grazing and burrowing activity (Burger et al., 2010). A final disadvantage of the use of agronomic seed relates to the fairly common practice of over seeding in order to ensure a full plant cover is achieved (Brink, 1980). Given the abundance and the low cost of these seeds, reclamation plans have often employed excessive seeding rates to achieve the ground cover necessary for their objectives (e.g. erosion control) (Brink, 1980; Ripley et al., 1996). This practice not only ensures no vacant patches exist for native plant invasion, but it also can result in large quantities of seed being transported into adjacent intact habitat, which may lead to loss of diversity and habitat degradation (Brink, 1980; Burger et al., 2009).

A favored, and in many cases obligatory, option to seeding with competitive agronomic plant species is to seed with plant species represented in the native community (Ripley et al., 1996; CDC, 2007). The advantages to using native plant species in mine land reclamation have been recognized for decades. The most readily observable advantage is the creation of a vegetative community that is in synchrony with the surrounding landscape, with habitat features that cater to the native flora and fauna (Bell & Meidinger, 1977; Ripley et al., 1996). In addition, plant species endemic to a region have been evolutionarily bred to survive and thrive in their particular edaphic and climatic conditions (though these conditions may have been drastically altered as a result of the mining activities). As such, native populations are far more capable of establishing in a self-sustaining manor that does not require agricultural inputs or efforts (Bell & Meidinger, 1977). This characteristic is of significant importance when considering particularly harsh environments such as the arctic tundra. In such a climate, the native population is far better suited to the extremes of temperature and exposure than would be an introduced agronomic plant, bred under more favorable conditions (Bell & Meidinger, 1977). When considering native plants for use in reclamation, there exists considerable variation in terms of growth forms available for use, as opposed to agronomics where the majority of plants are herbaceous in nature (Brink, 1980). In addition, with the recognized aim of encouraging the establishment of native vegetation, low seeding rates and diverse seed mixes are favored in order to facilitate the invasion of the site by the surrounding vegetation (Polster, 1991). Natural, or human created diversity in terms of edaphic or micro-site conditions of a mine waste site can increase species diversity as a result of the increased ecological niches favoring establishment of different organisms (Winterhalder, 1995). This increased species diversity may render the community more resistant or resilient to disturbances that threaten to remove an ecologically important function from the community. In addition, during the reclamation of disturbed lands, encouraging diverse growth forms may be particularly useful given its beneficial role on soil stabilization (Pierret et al., 2011). The different root architectures and spatial distributions exhibited by various plant species that may co-exist within a community result in a greater overall soil binding capacity and a more stable soil (Pierret et al., 2011). While native plant use in reclamation is ideal, there are a number of factors that hinder their usefulness and practicality. First, native plant species seed is not always available in sufficient quantity to fulfill the large requirement of reclamation projects (Bell & Meidinger, 1977). Additionally, the quality of the seed in terms of viability is generally far lower than that of agronomic plants, further increasing the seed demand in order to ensure establishment. In comparison to any agronomic plants, little is known about the specific requirements for the optimum growth of many native plants (Bell & Meidinger, 1977). As a result, attempts to seed native plants on amended or otherwise constructed soil substrates may prove unsuccessful if the established conditions are unsuitable for the particular plant species (Nowak, 1993). Overall, the cost of using native seed sources as opposed to agronomics is considerable. That being said, in order to protect biological diversity and as a result of the many other intrinsic benefits of encouraging a native plant community this cost should be in no way prohibitive.

An alternative plant selection option for use in reclamation of mine tailings and wastes is the use of plants (species, sub-species, cultivars or ecotypes) adapted to the harsh conditions established by the substrate (Johnson et al., 1994). As a result of high selective pressures, natural vegetation colonizing mine wastes rich heavy metals, excessive salts, as well as those prone to drought often express elevated tolerance to these harsh conditions (Bradshaw et al., 1978). Various plant cultivars have been collected from harsh, metal contaminated mine sites and bred for use as metal tolerant reclamation plants (Tordoff et al., 2000). Members of the Graminae (grass) family are the most common vegetation inhabiting harsh mine wastes, commercial examples include *Festuca rubra* 'Merlin' and *Agrostis capillaris* 'Goginan' and 'Parys' (Bradshaw et al., 1978; Tordoff et al., 2000). Metal tolerant populations may express other characteristics that enhance their suitability for revegetation of mine wastes, most commonly the capacity to survive under low nutrient or moisture availability (Bradshaw et al., 1978). The major advantage of using metal, or otherwise stress tolerant plant species in revegetation of mine wastes is the low overall cost and input (Johnson et al., 1994). While traditional physical and chemical stabilization techniques may cost as much as US$ 450 per m^3, phytostabilization can bring this cost down to between US$ 0.40 – 26 per m^3 (Mendez & Maier, 2008). Unfortunately, this plant selection strategy also has numerous disadvantages that must be considered. First of all, despite the use of metal tolerant populations in mine reclamation since the 1960s, a relative few metal tolerant cultivars have been produced and of those that were, most were bred under temperate climates (Tordoff et al., 2000). Therefore, many of these plants are not well suited for use in other climates where environmental conditions are vastly different (e.g. arid or tropical environments)(Tordoff et al., 2000). Additionally, given the lack of diversity in metal tolerant populations both in terms of species as well as growth form, land reclaimed with metal tolerant plants produces a low quality grassland with little capacity for invasion by ecologically important, native community members (e.g. nitrogen fixing plants, shrubs and trees) (Tordoff et al., 2000). Secondly, mechanisms for metal tolerances are specific to the metal itself, as a result, when many metals are present at potentially toxic levels in a mine wastes, it is unlikely that even a tolerant plant population will survive (Johnson et al., 1994). Furthermore, while some slow growing metal tolerant grasses also show tolerance to low nutrient conditions, in order to maintain the high ground cover required for erosion control, continued fertilization is often required (Bradshaw et al., 1978;

Johnson et al., 1994). Finally, the use of metal tolerant plant varieties to reclaim mine lands has the disadvantage of severely restricting the potential final land use of the area. The low quality grassland characteristic of this reclamation strategy is unsuitable for agricultural and grazing purposes, and may need to be fenced in to prohibit wildlife from browsing on plant tissues with significantly elevated levels of heavy metals (Tordoff et al., 2000).

Whether planting on a constructed cover material, or directly on the mine tailings surface, careful consideration must be given to planting techniques in order to maximize plant establishment and ensure successful revegetation. Seed mixes are usually composed of several perennial grasses and legumes, in addition to a warm or cool-season annual in order to establish rapid and continuous cover as well as to initiate organic matter build-up and nutrient cycling (Skousen & Zipper, 1996). Herbaceous cover plants (grasses and forbs, generally) are most often seeded rather than transplanted, despite better success with the later, largely due to cost considerations relating to labor intensity (Mendez & Maier, 2008). Additional considerations include both the timing of seeding, as well as the rate of seeding and relative proportions of different seed in the mix (Skousen & Zipper, 1996). The ideal time to seed varies for different climates as well as plant species; however, factors that influence germination include: water availability, soil compaction, soil contact with the seed, temperature, as well as specific seed characteristics (Lyle, 1987). When planting leguminous plants, it may be necessary to provide inoculation with an appropriate strain of *Rhizobium* bacteria in order to encourage the important nitrogen fixing relationship between these organisms (Skousen & Zipper, 1996).

Thin surface applications of mulches (e.g. straw, saw dust or wood chips, shredded paper or woven materials) can facilitate seed germination by improving water infiltration and temperature extremes as well as by preventing surface crusting and removal of seed by erosion (Lyle, 1987). Mulch can be applied by hand to small or inaccessible sites, or by heavy equipment (e.g. mulch blower) to more accessible sites (Lyle, 1987). Hydroseeding is a technique that has proven effective in the revegetation of steep slopes. Using this technique, seed is applied to the surface as a slurry containing amendments (fertilizers, lime, mulch, inoculants) to promote germination and plant establishment (Skousen & Zipper, 1996). When the revegetation plan calls for the planting of shrubs and trees, species selection must consider the physical and chemical nature of the substrate as well as issues relating to altitude, aspect, climate and exposure (Johnson & Bradshaw, 1977). Additionally, competitive interactions must be factored in to the planting strategy in order to ensure that fast growing and competitively dominant ground cover does not hinder the survival and growth of the woody plants (Burger et al., 2009). To cope with potential stresses such as desiccation or frost, woody transplants should possess an adequately developed rootstock; generally two to three year old materials are ideal (Johnson & Bradshaw, 1977). Additionally, depending on the specific properties of the mine waste, it may be necessary to plant trees and shrubs in excavated pits filled with a soil substrate suitable for the growth of the plant (Johnson & Bradshaw, 1977). In some cases, when access and costs are not prohibitive, providing irrigation to the seeded or planted vegetation can significantly improve the rate of establishment and survival (Mendez & Maier, 2008). Water availability has often been shown to have the most determinant effect on the successful naturalization of mine wastes (Ripley et al, 1996). In plants, water is essential to basically all metabolic functions, including nutrient acquisition, solute transport, photosynthesis and gas exchange, as well playing a structural role while maintaining turgor and allowing for cell elongation during active growth. In addition to these base functions, water availability can affect the ability of a plant to cope with certain environmental extremes that may be common to mine waste sites, including exposure to excessive heat or sunlight (Ripley et al., 1996). Furthermore, continual irrigation, for example through drip irrigation systems, can act to leach soluble salts through the soil profile, driving them out of the active rooting zone (Lottermoser, 2003). This can prove to be of particular benefit in arid climates where evaporation at the soil surface is sufficient to drive the upward movement of soluble salts in the soil profile (Bradshaw et al., 1978). Given the importance of water to plant survival, supplementing seasonal rain through irrigation can greatly improve the success of the revegetation effort, particularly in semi-arid to arid climates, areas with hot and dry summers, as well as locations with risk of damage from exposure (Mendez & Maier, 2008).

2.3.6 Boreal forest reclamation

The boreal forest is home to eighty percent of Canada's current mining operations as well as thousands of orphaned and abandoned mine sites that are still in need of reclamation (Northwatch et al., 2008). In addition, given the expected future of mineral and oil reserve mining in the region, there is significant potential for environmental degradation if sustainable practices, including active land reclamation, are not emphasized (Cranstone, 2002; Hilson & Murck, 2000). Exploiting the Athabascan as well as the Peace River, Wabasca, and Cold Lake oil sands deposits in Alberta's boreal forest, in particular, have the potential to generate vast quantities of mine wastes requiring reclamation (Cranstone, 2002; Northwatch et al., 2008). In fact, oil sands deposits underlay approximately 37% of Alberta's boreal forest, and as of 2010, more than 715 km^2 has been disturbed by mining activities (Government of Alberta, 2011). Of that disturbed land, almost half is peatland, and only approximately 71 km^2 is under active reclamation, and to date only 104 ha (1.04 km^2) have been certifiably reclaimed (Carlson et al., 2009; Government of Alberta, 2011). As previously discussed, this region holds tremendous cultural, ecological and economic significance to the country as a whole as well as to the approximately 3.5 million inhabitants (which include over one million First Nations people represented in almost 600 communities) and its protection should be a national priority (Northwatch et al., 2008).

The boreal forest can be characterized by long, harsh winters and short summer seasons that limit the growth of native plant species and preclude the establishment of many others (Rowe, 1972). As such, the species diversity in the boreal region is quite low relative to other temperate and tropical regions; an estimated 100,000 species are native to Canada's boreal forest (with approximately 1,200 native plant species) (Northwatch et al., 2008; OFRI, 2011). Additionally, the colder temperatures limit soil microbial activity and thus retard decomposition of acidic litter deposited by the dominant coniferous tree cover and as a result, delay nutrient cycling (Rowe, 1972). The prevailing shady conditions established by dense coniferous stands, in addition to the cool temperatures, will impede evaporation of water from the soil, leading to consistently high moisture contents and even yearlong water logging in some cases. Consequently, boreal forest soils are characteristically shallow, acidic, and nutrient poor, or organically rich in the case of peatlands where productivity of the moss layer exceeds the rate of decomposition leading to a build up of organic material (Rowe, 1972).

Land reclamation in the boreal forest region of the world represents a significantly different challenge as compared to reclamation of prairie land or even temperate forest reclamation. To begin, despite its expansive area, the boreal forest is dominated by a relative few number of coniferous as well as deciduous tree species (coming from the *Pinus, Picea, Abies, Larix* and *Betula* and *Populus* genera, respectively). This relative lack of diversity of dominant overstory plant species may hinder the reclamation process by limiting the potential species to employ for particular functions, for example during the pioneering community stage. Furthermore, the remote nature of many mine sites create difficulties in terms of access as well as logistical and economic constraints on the transportation and use of various materials required for reclamation. Additionally, there are many features unique to cold climates which many influence the success of cover designs as well as seeding and planting techniques (Ryaart & Hockley, 2009). Dozens of cold weather features have been identified; however those most pertinent to mine land reclamation are ground freezing and ground ice formation, ground thawing and thaw settlement as well as freeze-thaw cycling (Ryaart & Hockley, 2009). These processes can act to displace and degrade cover materials, breakdown soil structure and create landscape features that can further affect mine waste isolation and revegetation (Ryaart & Hockley, 2009). Peatlands as well as diverse hydrological features (including many wetlands, river systems, and lakes) can be drastically affected during mine operations; these can present significant challenges to reclaim and restore. Peat growth and accumulation is a complicated process that depends on factors relating to hydrology, water chemistry as well as species composition factors (Clymo, 1991). Furthermore, while the rate of accumulation and decomposition of organic matter in peatlands varies between climates, it may take between 20-100 years for the top layer of a peat bog (acrotelm) to pass to the bottom layer (catotelm)(Clymo, 1991). The complexity of these systems in addition to their slow rate of recovery create serious impediments on the reclamation of disturbed peatlands, particularly when certain objectives must be met on a short term time scale. That being said, 93% of Canada's 1.136 million km^2 of peatlands reside in the boreal forest region and collectively they store an approximate 147 billion tons of carbon (with the majority being held in the soil layer), demonstrating the importance these ecosystems have in maintaining global climate regulation (Carlson, 2010). Additionally, mining operations in the boreal forest often result in the draining of wetlands and lakes as well as the diverting of rivers and waterways, significantly affecting local and regional drainage dynamics (Wells et al., 2010). These features are difficult to restore to their natural state; however since boreal watersheds drain into three oceans as well as Hudson Bay, proper drainage dynamics in Canada's boreal are critical to global temperature dynamics (Wells et al., 2010). In addition, rapid and extensive snowmelt can lead to erosion of surface materials including vegetative propagules as well as cover materials and amendments (Ryaart & Hockley, 2009). Finally, despite the extensive water features of the boreal forest, much of the precipitation comes in the form of snow, and during snow free days evaporation can exceed precipitation leading to possible water limitation (Rowe, 1972; Eugster et al., 2000). Given extensive exploitation of the boreal forests natural resources, as well as the tremendous importance of this biome for global climate regulation, reclamation of disturbed lands is critical. Region and site-specific remediation techniques must continue to be developed and improved in order to provide the tools to protect ecosystems from degradation and possible destruction by the hands of industry.

2.4 Natural revegetation and succession of mine tailings

2.4.1 Characteristic community development

Many industrial wastes have shown the capacity to be naturally colonized by the adjacent native plant communities, given sufficient time (Roberts et al., 1981; Shaw, 1992; Shu et al., 2005). The naturalization of mine spoils (including tailings, waste rock, and smelting slag), like that of many other anthropogenic wastes, tends to follow a common successional pattern (Roberts et al., 1981; Shaw, 1992; Smith et al., 1997; Shu et al., 2005, Young et al. 2012). This pattern is characterized by early stage pioneer vegetation (generally dominated by herbaceous vegetation (largely grasses and forbes), generally progressing to a more woody, perennial community, and ultimately to a later-stage community, often resembling the surrounding vegetation (Polster, 1991; Shu et al., 2005). Factors relating to the rate of colonization, as well as the composition and overall trajectory of communities developing on disturbed lands such as mine wastes will be discussed.

The initial colonization of mine spoils is dictated by substrate characteristics as well as the presence of plant species capable of invading and establishing under the site-specific conditions (Winterhalder, 1993). Sites with a relative few impediments to plant growth may allow for rapid plant invasion, while those with harsh characteristics such as a hostile pH, phytotoxic levels of heavy metals or other contaminants, excessive salt levels, continual erosion, or unfavorable physical properties may resist colonization for many years (Nowak, 1993; Polster, 2008). In those sites where conditions are acceptably favorable for plant growth, initial colonization will be dictated in large part by the capacity of the surrounding vegetative community to disperse their seed and propagules to the exposed substrate (Winterhalder, 1993; Alday et al., 2011). Conversely, in harsh substrates, it is the response of the plant community to the edaphic impediments that is likely to dictate initial colonization dynamics (Polster, 2008). In either case, initial vegetative cover is generally composed of herbaceous plant species capable of invading exposed areas and establishing under nutrient poor conditions (Nowak, 1993; Shu et al., 2005). Many herbaceous as well as woody pioneers are capable of colonizing exposed areas due to seed characteristics that facilitate long-range dispersion (e.g. seed appendages and seed size) as well specific characteristics that may provide them with a competitive advantage in newly disturbed environments (e.g. slow growth rates, extensive root systems, or the capacity for atmospheric nitrogen fixation)(Stark & Rendente, 1985; Winterhalder, 1993; Nowak, 1993; Shu et al., 2005; Alday et al., 2011). Alternatively, initial colonization may occur only at the margins of the waste area where favorable conditions are established by the proximity to the established vegetation, or in micro sites that present conditions favoring plant establishment (Neel et al., 2003; Shu et al., 2005, Young et al. 2012). This pioneering cover accomplishes a host of functions that will facilitate the further invasion of the area by other native plant species. Some of the important functions of pioneer vegetation include providing organic matter inputs to the soil and soil surface, the creation and protection of surface and sub-surface soil structure, as well as improving the nutrient status of the substrate (Zabowski et al., 1993; Bradshaw, 2000; Osman & Barakbah, 2011, Young et al. 2012). The improved substrate conditions permit a shift in dominant plant life forms, from a community composed primarily of annuals, to a perennial herb community, and then, potentially, to a woody stage community (Alday et al., 2011). The rate at which this shift takes place depends on many site and plant specific characteristics, that being said, some generalization can be drawn. For example, more acidic mine wastes tend to remain in the herbaceous stage longer than non-acidic wastes, and sandy, nutrient poor substrates tend to progress to a woody-stage more rapidly than do fine, nutrient rich substrates (Rebele, 1992; Winterhalder, 1993).

Following colonization, ecosystem development can proceed with further colonization by appropriate species, accumulation of nutrients in plants and soils, alteration of soil structure from plant and animal activity as well as through the reduction of any toxicity that may have existed (Bradshaw, 1983; Winterhalder, 1993). Succession of plant species will result from differential dispersal, germination, growth, survival, life span, and competitive abilities amongst community members leading to the replacement of pioneer plant species by later stage vegetation (Rebele, 1992). During plant colonization and succession, species richness tends to increase with the age of the waste site, although the maximal species richness may be found in intermediate-stage communities as a result of the co-existence of early and later staged vegetation (Wali, 1999; Alday et al., 2011, Young et al. 2012). Additionally, distinct plant communities may develop in response to variability in edaphic conditions (pH, texture, nutrient status, water availability, as well as mineralogy), or in response to climatic and micro-climatic conditions (e.g. exposure, aspect, overall topography)(Winterhalder, 1993; Rebele, 1992; Martinez-Ruiz et al., 2001; Néel et al., 2003). In fact, plant species diversity across a naturalized mine waste site can be significantly enhanced by the presence of diverse habitat features, and general resource heterogeneity (Winterhalder, 1995). Furthermore, many factors that are related to soil fertility (e.g. favorable nutrient status and textures, available moisture, and low levels of soluble salts) may decrease overall diversity in disturbed sites by favoring the establishment of a relative few competitively superior plant species (Stark & Redente, 1985; Zabowski et al., 1993). While some wastes may present features which permanently eliminate some community members from the area (Chambers et al., 1987), the progression of disturbed lands through normal successional stages provides many functions, most notably habitat for native wildlife and carbon sequestration in soils and vegetation (Shrestha & Lal, 2006; Prach & Hobbs, 2008).

2.4.2 Changes in mine tailings properties over time

While the changes, as well as the rate of change, observed through the lifetime of a mine waste dump will depend on its specific properties and climatic setting, we can expect to see changes in physical, chemical, as well as biological aspects of the substrate (Bradshaw, 2000). Generally speaking, with the exception of tailings weathering leading to increased acidity, salinity, and metal availability, most changes during tailings naturalization can be considered improvements to soil fertility. Many studies have described the natural development of plant communities on mine wastes as well as other industrially disturbed lands offering many common generalizations regarding the improvements to soil substrate characteristics and properties over successional time (Leisman, 1957; Roberts et al., 1981; Roberts et al., 1988; Smith et al., 1997; Shu et al., 2005).

To begin, as a result of a generally uniform particle size and a lack of organic matter, mine tailings are often described by a smooth surface and a compact nature (Bradshaw et al., 1978). Water infiltration and transmission characteristics are poor, and the surface is highly susceptible to wind and water erosion (Ripley et al., 1996). During the colonization of this new substrate by pioneering vegetation, plant roots and soil biota fed by the inputs of the vegetative community function to reduce compaction and overall bulk density (Wali, 1999; Bradshaw, 2000, Young et al. 2012). Not only does the infusion of organic matter decrease the overall soil density, but as root tissue dies and is decomposed, large voids are created providing important pore space for water and gas transmission as well as habitat for soil micro flora and fauna (Bradshaw, 2000; Pierret et al., 2011; Chenu & Cosentino, 2011). Furthermore, organic matter contributions in the forms of roots, fungal hyphae, as well as extracellular polysaccharides (EPS) from soil biota are the main drivers of soil structure formation and protection (Chenu & Cosentino, 2011). While sterile mine tailings are unaggregated and lack developed structural features, during the naturalization process, the plant and soil communities act to create stable aggregation and structure that will enhance soil fertility and water relations (Roberts et al., 1988; Wali, 1999). The buildup of plant litter and other organic materials will facilitate further plant invasion by contributing to improved seedbed characteristics through processes including thermal regulation, moisture retention, as well as the establishment of favorable micro sites for seed germination (Lyle, 1987). Finally, soil horizon development may occur during the naturalization process as organics accumulate at the soil surface and products are leached or mixed by biotic agents forming distinct soil horizons (Leisman, 1957; Néel et al, 2003).

Improvements to the mine tailings chemical properties through the process of naturalization include the buildup of organic matter and nutrients, reduction in availability of heavy metals, as well as changes in soluble salt content (Zabowski et al., 1993; Wali, 1999; Bradshaw, 2000). To begin, windblown organics may accumulate at the tailings surface if surface characteristics allow for it; however significant accumulation of organic matter generally beings when pioneering plants establish and return their root and shoot biomass to the substrate (Bradshaw, 2000). The initial buildup may occur relatively slowly when slow growing pioneers dominate early stages of naturalization, or when the pioneer plant species resist decomposition and nutrient turnover (related in part to the C:N of the plant tissue) (Schafer et al., 1980; Wali, 1999). That being said, the establishment of a plant cover and a litter layer will encourage further sediment, organic matter, and plant seed capture, thereby accelerating the process (Bradshaw, 2000; Osman & Barakbah 2011). In addition, some pioneer plant species exhibit characteristics that may substantially improve the fertility of their substrate. *Equisetum* species, for example, are common pioneer vegetation on naturalized tailings as well as other naturally nutrient poor habitats (Néel et al., 2003, Young et al. 2012). These plants may ameliorate substrate conditions and overall net productivity of the ecosystem by accessing and rapidly turning over nutrients present at depths out of the rooting zone of other plant species (Marsh et al., 2000). Furthermore, plant species capable of atmospheric nitrogen fixation are a significant source of nitrogen to nutrient poor soils, contributing up to 100 kg N $ha^{-1}yr^{-1}$ (Blundon & Dale, 1990; Bradshaw, 2000). Overall, the accumulation, decomposition, and mineralization of organic matter during naturalization of mine wastes is directly linked correlated with increases in organic carbon content (%), as well as total nitrogen and most other plant nutrients (Leisman, 1957; Jha & Singh, 1991; Smith et al., 1997). The rate at which this accumulation of organic matter occurs is drastically different between different climatic regions, leading to significant differences in terms of nutrient buildup (Burykin, 1985). The accumulation of organic substances as well as the formation of humus may have the potential to further improve the chemical conditions of the substrate by binding and sequestering unwanted heavy metals (Livens, 1991). In many cases, soil substrates enriched in heavy metals have been remediated through the incorporation of low cost organic materials into the rooting zone (Calace et al., 2011). The eventual decomposition of the organic compounds may release the bound metals back into solution; however, it is also possible for these metals to be incorporated into long-lasting organic-metal complexes within the structure of humus (Misra et al., 1996). Soluble salt content of mine wastes can be highly variable due to specific mineralogy as well as processing techniques and weathering conditions (Lottermoser, 2003). While soluble salts are released during the weathering of some mine wastes, the presence of salts in the active rooting zone is often attributable to the prevailing climate and location relative to the local water table (Ripley et al., 1996). In temperate and tropical regions where precipitation exceeds evaporation from the soil, there is a consistent decrease in soluble salt concentration over time due to their leaching through the soil column (Schafer et al., 1980; Jha & Singh, 1991; Wali, 1999). Conversely, in semi-arid to arid environments, not only will precipitation not leach soluble salts from the rooting zone, but the force of evaporation at the soil surface may be sufficient to bring salts up in the soil column as the water table is drawn up (Ripley et al., 1996). In such cases, the establishment of plant cover can be very slow and the establishment of woody plant species can be particularly hindered (Torbert et al., 1988; Borden & Black, 2005).

Finally, the biological properties of mine wastes change considerably during their naturalization, being both influenced by, and a driving factor in, the changes to the chemical and physical nature of the substrate (Insam & Domsch, 1988; Shrestha & Lal, 2006). Soil flora and fauna are indispensable in the breakdown and cycling of nutrients from organic matter. While barren mine wastes generally lack the necessary soil biota to accomplish these tasks, populations will develop given a suitable substrate (Bradshaw, 2000). Other important groups of organisms include nitrogen fixing bacteria as well as mycorrhizal fungi that actively contribute to plant health and soil fertility through their respective symbiotic relationships. Like the decomposer biota, these organisms are fairly ubiquitous in nature and capable of colonizing new areas rapidly, their introduction is rarely needed to ensure their presence (Barton & Northup, 2011). The dynamics of microbial succession are complicated and relative to plant succession, poorly described (Barton & Northup, 2011). However, generally speaking soil microbial populations are responding to substrate factors such as nutrient status, water and oxygen content, as well as habitat availability and grazing or predatory pressures. In fact, microbial communities are such important factors in mine waste reclamation and restoration that ratios of microbial carbon to organic carbon as well as other soil microbial community health markers (such as fatty acid methyl ester (FAME)) have been proposed as viable characteristics for use in the assessment of reclamation success (Insam & Domsch, 1988; Mummy et al., 2002).

2.4.3 Recovery of ecosystem services and functions

The dumping of mining wastes in natural settings removes many important ecosystem functions from the area including, but not exhaustively, climate and water regulation, erosion control and sediment capture, soil formation, nutrient cycling, waste treatment, refugia, food production, pollination, as well as recreation and culture (Ripley et al., 1996; Costanza et al., 1997). While some ecosystem functions and services may never be recovered or returned to their natural state, depending on the nature of the substrate and the local climate and community, other important features may recover within a few years (Chambers et al., 1987; Bradshaw, 2000). In order to make assessments regarding the natural rate of recovery of many of these services, specific attributes of the ecosystem must be quantitatively measured. Some of the attributes of particular importance include energy capture, energy flow, food web, and population dynamics, structure, niche, and community diversity, as well as successional patterns, and participation in biogeochemical cycling (Ripley et al., 1996; Mummy et al., 2002). While some above ground characteristic and features are easily observable, others are transient in nature and require considerable effort to accurately describe. Of particular importance can be the description of soil biota, given their overall absence from fresh mine wastes as well as due to their requirement for soil formation, plant establishment and transfer of soil organic matter (SOM) (Mummy et al., 2002).

The path towards the recovery of many ecosystem functions begins with the first step in soil formation, that is the formation of a biologically active surface medium capable of allowing the establishment of some pioneer vegetation (Burykin, 1985; Neel et al., 2003). This process may require little to no time, or significant amounts of time depending on the initial characteristics of the waste, and the capacity of biotic and abiotic factors to ameliorate the conditions (Wali, 1999; Martinez-Ruiz & Marrs, 2003). Subsequent steps in soil formation on mine wastes are the accumulation of humus and the leaching of weathered materials through the soil column resulting in distinct horizon development, as well as soil zonation across the waste area (Burykin, 1985). The rate of soil formation is a function of the climate, relief, parent material, organisms, as well as time since deposition and management practices under which the soil develops (Jenny, 1958; Dudal, 2004). As a result, significant variation has been observed when characterizing the development of soil substrates from industrial wastes such as mine tailings (Shaw, 1992; Bradshaw, 2000; Neel et al., 2003). The vegetative community significantly affects the rate of organic matter and soil organic carbon accumulation, with variations occurring between different land usages (e.g. agricultural vs. forest land) as well as depending on the dominant plant species present at that stage (Leisman, 1957; Shrestha & Lal, 2006). The relief of a mine waste site has been shown to have a determinant effect on plant establishment and community succession (and therefore soil formation) as a result of the exposure as well as water distribution effects (Wali, 1999; Martinez-Ruiz et al., 2001). Climate will also have a significant effect on the rate of soil development, through the effect of temperatures on organic matter decomposition and mineralization, and precipitation on leaching and weathering processes. Finally, the most important factor relating to the rate of soil formation on mine wastes is the nature of the substrate, or parent material (Shu et al., 2005). Plant colonization and the rate of succession will depend on edaphic factors such as salinity, pH, texture, and stability (Leisman, 1957; Borden & Black, 2005). Given the considerable variation both between and within mine sites, it is not surprising that at one particular site, twenty years after the wastes were dumped, vegetative cover can be non existent, or have developed into a herbaceous or woody cover of variable consistency (Gagnon & Crowder, 1984).

Initial pioneer plant cover as well as later stage communities can contribute towards numerous functions besides soil formation, including erosion control and sediment capture, water regulation, habitat, as well as nutrient cycling and waste treatment (Costanza et al., 1997). The vegetative community contributes to erosion control both through active means such as raindrop interception, fixation of particles by roots, and by increasing water infiltration, as well as passively through trapping eroding particles (Osman & Barakbah, 2011). The plant community will also have a significant role on water regulation through its effects on erosion control, soil structure development, organic matter accumulation and incorporation, as well as through transpiration and water use. Additionally, the plant community provides habitat as well as resources for animal, insects, fungi, and bacteria of all types both during its lifetime as well as during its decomposition (Barton & Northup, 2011). In fact, the progression of the vegetative community through various seral stages will foster the creation of diverse habitat features (both above and below ground) catering to the native wildlife community (Prach & Hobbs, 2008; Barton & Northup, 2011). The diversity of habitat and resources features established along the successional pathway will increase the overall resource use efficiency of the ecosystem as a result of the increased niche dimensions (Mummy et al., 2002). The overall increase in species richness observed through time since disturbance has a corresponding effect on ecosystem stability and resilience (Myers, 1995). The return of plant biomass to the soil substrate as well as the mechanisms by which it is protected are the main processes leading to microbial activity, carbon build up, as well as nutrient availability in a soil (Shrestha & Lal, 2006). Over time, accumulations of organic matter will be decomposed and mineralized, providing important nutrients to the plant and microbial community. Though total nutrient pools may not approach that of adjacent undisturbed soils, sufficient nutrient capitol for the progression of early stage communities to later, more nutrient demanding communities can often be observed within several decades (Chambers et al., 1987; Zabowski et al., 1993). Finally, the accumulation and incorporation of organic materials into the soil substrate can serve a function in waste treatment. Not only does revegetation and phytostabilization limit the movement of hazardous particles off-site, but it can also reduce the local availability of certain heavy metals of other contaminants though the formation of stable organic complexes (Mendez & Mainer, 2008). While some harmful substances (e.g. heavy metals) cannot be degraded through natural processes, they may be bound and incorporated into stable organic complexes, effectively removing them from soil solution for the foreseeable future (Calace et al., 2011).

2.5. Conclusion

The protection and conservation of our natural resources and ecosystems is paramount to the sustainability of the industries that exploit them, as well as in order to minimize the impact that these industries have on the local, regional, as well as global environments. Given the large amount of mineral exploration and mining that is, and has, occurred within Canada's boreal region, in addition to the tremendous importance that this area holds in relation to global climate regulation, water quality protection, and wildlife habitat, methods to mitigate the impacts of mining activities in this region deserves significant consideration. Furthermore, as a result of the great variation in mine tailings physical and chemical composition, a depth of technical information on the development as well as the relative success of remediation projects implemented on a range of tailings substrates would provide guidance for mine operators and reclamation technicians undertaking similarly characterized substrates. Large-scale field experiments examining the various remediation and revegetation options are an important step towards developing consistent technical approaches to reclamation processes across the great range of settings in which mining has and will continue to occur.

Chapter 3. Materials and Methods

3.1 Site description

The mine tailings left by Gunnar gold mine are situated on the west shore of Beresford Lake within Nopiming Provincial Park, in southeastern Manitoba (50°51.37', 95°29.60'). This mine was most active from 1936-1942, during which time almost 260 000 tons of mineral ore was processed, yielding 3 101 kg of gold (Bamburak,1990). The mine explored gold-bearing quartz veins of the Rice Lake Greenstone belt. The mineralogy of the tailings dump has been inferred to be primary in nature (remaining unaltered geochemically since its deposition), and composed largely of quartz, calcite, plagioclase, pyrite, Fe-oxides, galena, sphalerite, as well as micas and clays (Lambert, 2001). The Gunnar mine mill pumped tailings to the south of the mine site, in a lowland depression that covered an area of 11 hectares (Slivitzky, 1996), in the seventy-plus years since they were dumped, approximately 60% of the tailings dump has naturally succeeded to a black spruce-larch dominated boreal forest. Factors limiting the establishment as well as growth of plants throughout the site have been identified as a lack of plant available nutrients and organic matter, as well as compaction and issues relating to water availability (Renault et al, 2006). The non-vegetated tailings occupy an area of approximately 360 by 120m and show a range of elevation varying by around 1.8m (Figure 3.1) (Markham et al., 2008). There is no indication of significant differences in tailings mineralogy laterally across the tailings site; however the surface topography varies somewhat with the center of the tailings pond remaining

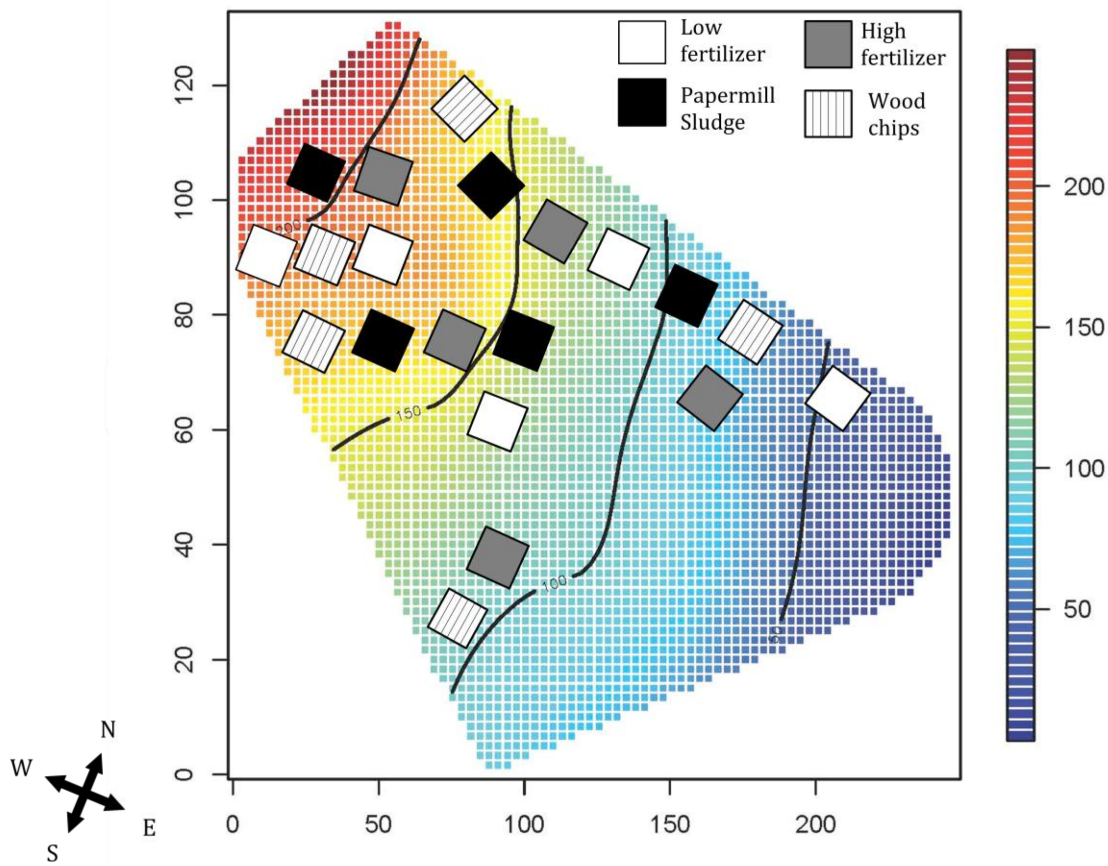

Figure 3.1 Change in relative elevation (m) across the Gunnar mine tailings site and location of experimental plots. The scale indicates cm above a reference point at the east end of the site. Elevations were measured on a 30 x 30 m grid across the site and the elevation map generated using a thin plate spline in the fields library of R (version 2.6.2).

relatively flat, while the north and north-east perimeters exhibit an undulating relief providing ideal terrain for the local ATV users. Though off the main roads and away from the view of casual tourists, this tailings deposit removes valuable land from important functions such as carbon sequestration (in soils, organic residues and standing vegetation) and habitat for a great number of organisms (Swift, 2001). The rate of naturalization, as well as the changes in tailings characteristics and vegetation has recently been described at the Gunnar mine tailings site (Young et al., 2012). The invasion of the tailings by the dominant overstorey trees (*Larix larcinia* and *Picea mariana*) are occurring at a rate of 1.45 m yr^{-1} and 1.56 m yr^{-1}, respectively, and the development of the plant community corresponds with improvements to the fertility of the substrate in terms of their chemical and physical fertility (Young et al., 2012). Based on the rate of natural succession of trees to this area, without active remediation efforts, we can expect to wait between 70 and 110 years more before the entire site becomes forested (Markham et al., 2008).

3.2 Tailings description

Many of the Gunnar gold mine tailings' properties are presented in Table 3.1. The neutral to slightly alkaline pH range will affect the bioavailability of both metals in the tailings matrix as well as any nutrients present or applied to the tailings. For example metals such as aluminum, iron, and zinc become much more bioavailable at acidic pH, while the availability of phosphorous can be significantly reduced at more basic pHs (Brady &Weil, 2008). The low levels of available nutrients present in the tailings (Table 3.1) indicate that nutritional deficiencies may be

Table 3.1 Geochemical and fertility data for Gunnar Gold mine tailing deposit (2008).

Parameter		Range of values
Conventional Parameters		
pH	dH_2O pH	7.7-8.3
	CaCl pH	7.5-7.9
Conductivity *($dS\ m^{-1}$)*		0.3 – 4.6
Texture	Sand *(%)*	82.6
	Silt *(%)*	12.9
	Clay *(%)*	4.5
Bulk Density *($g\ m^{-3}$)*		1.38 – 1.42
Geochemistry (mg kg^{-1})		
Aluminum (Al)		4,640-26,600
Arsenic (As)		9.9-229
Calcium (Ca)		23,400-57,300
Chromium (Cr)		21-91
Copper (Cu)		128-371
Iron (Fe)		22,300-53,800
Lead (Pb)		7.1-32.7
Magnesium (Mg)		5,540-18,600
Manganese (Mn)		636-1,100
Phosphorous (P)		191-507
Potassium (K)		216-492
Sodium (Na)		21-222
Zinc (Zn)		66-424
Fertility		
Available phosphorous (*mg kg^{-1}*)		<1
Inorganic nitrogen (*mg kg^{-1}*)		<0.5-18.4
Available sulfate (*mg kg^{-1}*)		69-896
Available potassium (*mg kg^{-1}*)		19-119
Organic carbon *(%)*		>0.25

observed in any inhabiting vegetation. Furthermore, as the tailings lack the necessary organic matter or clay particles to bind and exchange charged ions, any nutrients applied to the matrix will likely be prone to rapid loss through leaching. The tailings are also susceptible to compaction due to a lack of organic matter and their dense nature (Table 3.1). As a result, pore space necessary for the movement of water and gases will be limited throughout compacted tailings. This dense substrate will present mechanical stresses to plant roots attempting to grow throughout it, and soil organisms that use soil pores as habitat will be restricted from the area. The poor water infiltration and holding capacity of mine tailings may result in periods of drought as well as water logging depending on local precipitation and level of the water table. Though considerable variation in specific metal content was observable across the mine tailings site, arsenic, copper, chromium, and zinc are all present in excess of the guidelines established by the Canadian Council of Ministers of the Environment (CCME) for parkland soils in certain locations (soil contact guideline: As: 17 mg kg^{-1}, Cu 63 mg kg^{-1}:, Cr 64 mg kg^{-1}, Zn: 200 mg kg^{-1} (CCME, 2007).

3.3 Experimental design

This experiment was set up as a completely randomized design where four amendment treatments (Table 3.2) were each applied to 15m by 15m-experimental plots. Twenty experimental plots were established in early June 2009, covering much of the unvegetated mine tailings (Figure 3.1). This number of plots maximized the area being actively revegetated, and allowed each amendment treatment to be

Table 3.2. Amendment treatments for revegetation experiment at the Gunnar mine tailings.

Treatment	Organic amendment rate (kg m^{-2}):	Fertilizer rate (kg N ha^{-1})*:
Low Fertilizer	none	100
High fertilizer	none	250
Wood chips, low fertilizer	3.50	100
Papermill sludge, low fertilizer	3.70	100

*Applied as a 25-5-10 (N-P-K) granular fertilizer, 13-week release.

replicated five times. In order to minimize the variation between plots, we avoided any areas that had pre-existing vegetation or organic matter build-up, as well as areas with highly varying topography and regions disturbed by ephemeral surface streams. The experimental plots were subdivided into four quadrants, each of which received a unique seeding strategy (split-plot design) (Figure 3.2); *Picea mariana* seedlings were planted across all plots.

3.4 Amendments

Most amendment and seeding treatments chosen for this large-scale field experiment (Table 3.2, Figure 3.2) were selected based on a prior pilot experiment performed at the Gunnar tailings deposit throughout 2006 - 2008. Renault et al. (2006) found that the incorporation of papermill sludge and inorganic fertilizer to the tailings surface through rotor tilling could be of benefit to the growth and survival of certain test species, including *Festuca rubra* (red fescue) and *Medicago sativa* (alfalfa). Papermill sludge is one of the main waste by-products produced during the processing of timber for the pulp and paper industry, it is generated through effluent clarifying treatments. The physical and chemical properties of the sludge produced by any given mill depends on the treatments through which it is generated; however two general types of sludge are common to modern paper mill facilities, these are primary and secondary (or activated) sludge (Watson & Hoitink, 1985). Primary papermill sludge is composed largely of cellulose fibers, tannins, lignins, as well as additives (such as clays), and is generated through primary sedimentary clarification. Secondary (or activated) sludge is produced through

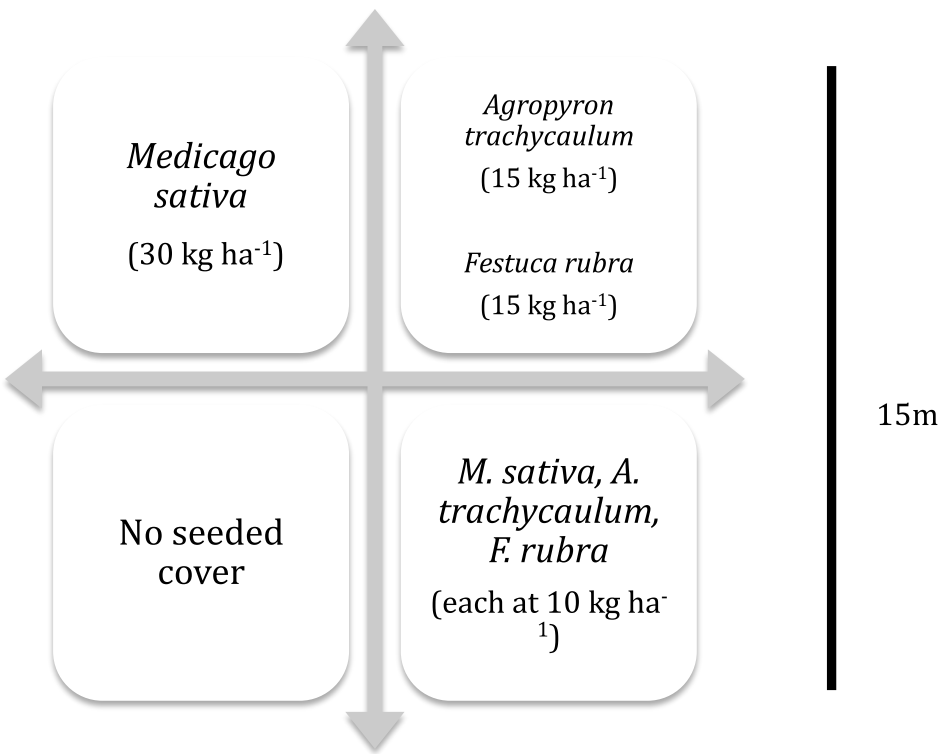

Figure 3.2. Ground cover plant species planting strategy for Gunnar revegetation project, total area represents one full experimental plot.

biological treatments where nutrients as well as desired microbial populations are added to the mill effluent in order to reduce its high biological oxidation demand (BOD)(Watson & Hoitink, 1985). Both types of sludge are characterized as having some slowly degradable organic matter (largely cellulose) as well as large amount of calcium and lesser amounts of other nutrients such as nitrogen, phosphorous, potassium, magnesium and sodium (Cabral et al., 1998). Secondary sludge contains a significantly greater proportion of nutrients, especially nitrogen and phosphorous, and generally has a carbon to nitrogen ratio (C: N) in the range of 6 to 10 (as compared to primary sludge, which generally has a C:N> 100)(Watson & Hoitink, 1985; Cabral et al., 1998). The papermill sludge used in the field experiment at the Gunnar mine tailings site was obtained from the Tembec papermill situated in Powerview-Pine Falls, MB. This facility combined three separate waste streams to produce their final sludge, which they would then use to fuel their boiler systems. The three streams were primary sludge (settled out from primary clarification), secondary sludge (largely dead bacteria and protozoa from conventional aerobic activated processes), as well as old new paper and magazines, which were pulped and added to the sludge to aid in the dewatering process. Sludge was dewatered with the use of a gravity table, belt-press, and screw-press yielding a dry, dense, organic rich, granular product with the following nutrient contents: 35.3 mg kg^{-1} nitrate-N; 45.0 mg kg^{-1} phosphate-P; 484 mg kg^{-1} potassium-K; 49.0 mg kg^{-1} sulphate-S; 5,580 mg kg^{-1} calcium; 1,090 mg kg^{-1} magnesium; the C: N of this sludge was approximately 25:1. This organic amendment incorporation ameliorates water and nutrient availability and has been shown to improve soil physical properties including macro-aggregate content, bulk density, field capacity and organic content (Camberto et al., 2006).

The second organic amendment, a mixture of coarse and fine wood chips, has been used as a soil additive to improve water infiltration and retention and as a carbon source (Rensburg & Morgenthal, 2004). This material was an attractive option for our project as it was readily available to us in large quantities from the same source as the papermill sludge, thereby reducing the costs associated with the transport of the material. This material has a low nutrient content and a very high carbon to nitrogen ratio, exceeding 150:1. As a result, upon incorporation, it was expected that some level of nutrient (particularly nitrogen) immobilization would occur, as microbial growth will consume any available nutrients during decomposition of the carbon supply.

Previous studies have indicated that little plant establishment can be expect in these mine tailings without providing an initial inorganic fertilization (Renault et al., 2006), as such all of our amendment treatments will include some fertilization (Table 3.2). Two treatments were established that did not receive an organic matter addition; these plots were amended with one of two fertilization rates (100 kg N ha^{-1} and 250 kg N ha^{-1}) (Table 3.2). The fertilizer chosen for this experiment was obtained from the turf and reclamation branch of Brett Young Seed, MB. This fertilizer was a 13-week slow-release, granular product with a NPK mix of 25-5-10. Organic materials as well as fertilizers were spread on the surface of the tailings at which point they were incorporated into the top 15 cm of the tailings by way of rotor tillage. Papermill sludge and wood chips were moved throughout the site from the area where they were dumped both by hand, using wheelbarrows, as well as with the use of a cab-less John Deere 4720 compact tractor fitted with a 400x loader (1850 mm wide bucket). The amendment incorporation and tillage was accomplished by this same compact tractor, which was fitted with a John Deere three point hitch tiller set to till to the depth of approximately 15 cm (Figures 3.3. – 3.6.). Papermill sludge was applied at a rate of 3.50 kg m^{-3}, and the wood chips at a rate of 3.70 kg m^{-3}. The rate of papermill sludge was set based on the successful use of this organic amendment in small-scale field, as well as greenhouse experiments (Renault, 2006; Green & Renault, 2008). Additionally, the rate of wood chips was based on successful use of this organic material in prior reclamation projects (Vasconcelos & Cabral, 1993; Rensburg & Morgenthal, 2004). The intention of this project was to focus on a rate that could be optimally feasible on a large scale and therefore we kept our rate of application relatively low (Table 3.2).

3.5 Seeding and planting treatments

The plant species introduced to the Gunnar mine tailings were chosen based on desirable characteristics relating to their tolerance to the tailings environment as well as their potential to ameliorate the substrate. The plants range considerably in growth form and include two grass species, a legume, a shrub, and a dominant over story tree species. All plants are naturalized within Manitoba, and the shrub and tree species were chosen in the hopes of mimicking the surrounding vegetation. Some important characteristics and considerations for value of the chosen plant species are presented as follows:

Figure 3.3. Pile of wood chips amendment and equipment used to move amendments around site.

Figure 3.4. Organic amendments spread on surface of experimental plots; background: wood chips, foreground: papermill sludge.

Figure 3.5. Broadcast application of inorganic fertilizers.

Figure 3.6. Incorporation of amendment treatments into surface (15-20 cm) of the Gunnar mine tailings.

- *Festuca rubra* (creeping red fescue)
    - Common reclamation grass species with a great ability to provide erosion control
    - Exhibits high levels of drought, salt and acid tolerance (Hardy BBT, 1989)

- *Agropyron trachycaulum* (slender wheatgrass)
    - Common reclamation grass species with great soil building potential due to its significant biomass production
    - Exhibits medium drought tolerance and high salt tolerance (Hardy BBT, 1989)

- *Medicago sativa* (alfalfa)
    - Capable of fixing atmospheric nitrogen, this plant species is common to disturbed habitats and can serve to ameliorate the soil substrate,
    - Abundant flower production provides for insect pollinators

- *Picea mariana* (black spruce)
    - Native to the area, *P. mariana* is the most common over story community member in this area of Nopiming provincial park

- *Salix* spp. (willows)
    - Common species and growth forms surrounding the mine tailings site
    - Fast growing and easy to propagate
    - Provides food source for wildlife including moose and deer.

All grass and *M. sativa* seed was obtained from Brett Young Seed, MB. Dormant one-year-old *Picea mariana* seedlings (from Manitoba seed zone 2) were obtained from Pine Land Nursery, Hadashville, MB in the spring of 2009 and were refrigerated for approximately one week until two days prior to their planting.

Seedlings were an average of 21.6 cm in height with an average root collar diameter of 2.00 mm and were planted in Can-Am Multipot #1 (57 cm3) cells. *P. mariana* seedlings were planted across all experimental plots in rows spaced ~ 1.5 m from each other at a rate of 4,400 stems ha^{-1} shortly after treatment applications. *Salix* spp*.* (*Salix discolor, S. lucida, and S. planifolia*) cuttings were collected during early May in 2009 (one month pretreatment) and 2010 (almost one year post treatment) from the shrub dominated region to the east of the Gunnar tailings pond. Cuttings were rooted in a greenhouse with natural light supplemented with high pressure sodium lamps set to 16 hour photoperiod and with a temperature of 25 ± 5°C over a period of several weeks. In 2009, the *Salix* cuttings were propagated in undivided planting trays and out-planted as bare rootstock in early June of that year. The young bare root cuttings did not survive the 2009 field season, likely as a result of the high exposure to sun and wind leading to an inadequate water supply. For the 2010 planting of *Salix* spp., cuttings were rooted in separated soil plugs (43 cm^3), and out planted with the small soil plug, which provided sufficient substrate to allow for the initial establishment of the young shrub. In both seasons, *Salix* spp. were planted at a rate of ~ 2,500 stems ha^{-1}.

3.6 Tailings Fertility

3.6.1 Sampling

Composite tailings samples were collected throughout the experimental period (2009 – 2011) at various points during the growing season (spring, summer, and fall). For the first growing season (2009), samples were taken on a plot basis, as it was not expected that the seeding treatments would have had sufficient time to exert a significant effect on the tailings fertility measurements. Following the first growing season, as differences in vegetation cover were observed between sub-plots, samples were taken on a sub-plot basis. Samples were collected on June 4th, July 7th, and September 10th for 2009; May 27th and October 4th for 2010; and June 11th for 2011. Sampling was done with a 2 cm diameter Oakfield soil corer (Oakfield, Wisconsin) soil sampler. When sampling was done on a plot wide basis, a minimum of 15 cores per plot (10 – 15 cm deep) were collected, while when sampling was done on a sub-plot basis, a minimum of 8 cores were collected. Tailings samples were air dried in a well-ventilated greenhouse. Bulk density was sampled in the early summer of 2009, 2010, as well as 2011.

3.6.2 Analyses

Particle size distribution was assessed by the hydrometer method (Green, 1978) for each plot in order to ensure that there were no major differences in texture across the experimental area. Bulk density was assessed using prior to treatment, and one and two years following treatment application (Green, 1978). Tailings sampling for bulk density was done in the field using a coring device, which drew a 590-cm^3 sample (15 cm x 7.5 cm core). Tailings electrical conductivity was measured prior to and following treatment application using a conductivity meter (Traceable, Control Company, Texas, USA). For this analysis, 20 g of air dry mine tailings were saturated with 20 mL of deionized water, well mixed, and filtered through Whatman 42 filter paper. This filtrate was then used for the determination of electrical conductivity (Schofield & Taylor, 1955). Organic carbon was quantified throughout the experiment (by Walkley-Black wet digestion (Walkley & Black, 1934; Grewling & Peech, 1960). For this determination, 2.5 g of mine tailings (passed through a 0.5 mm sieve) were digested with 20 mL of concentrated sulfuric acid, in the presence of 10 mL of 1N potassium dichromate; a blank was included where no tailings were present. After being swirled for one minute and allowed to incubate for one hour, 100 mL of water and a few drops of *o*-phenanthroline indicator were added to the suspension. The suspension was then titrated with 0.5 N ferrous sulfate heptahydrate to an endpoint that is characterized by a change in color from blue-green cast to red. Organic carbon was then determined using the following formula:

$$\text{Carbon in soil (\%)} = M \times ((V_1 - V_2) / \text{Mass of soil (g)}) \times 0.39$$

Where,

M = Molarity of ferrous sulphate solution

0.39 = Correction factor

V_1 = Volume of ferrous sulphate required for blank

V_2 = Volume of ferrous sulphate required for sample

Total inorganic nitrogen was measured throughout the experiment by the 2M KCl micro diffusion method (Mulvaney, 1996). Inorganic nitrogen was extracted from 25 g of air-dried tailings (passed through a 0.5 mm sieve) by adding 100 mL 2M potassium chloride and shaking this suspension for an hour on a Gyrotory bench shaker (New Brunswick Scientific Company). After allowing the suspension to settle overnight, the supernatant was poured off and collected in micro diffusion jars. Magnesium oxide and Devarda's alloy were each added to the extract (in excess of 0.2 g), swirled, and closed for a five-day incubation period in the airtight jar in the presence of 4 mL of 4% (w/v) aqueous boric acid indicating solution containing Bromocresol Green – Methyl Red Indicator (0.1% bromocresol green and 0.02% methyl-red in methanol)(Ricca Chemical Company). Following the incubation period, the boric acid indicator solution was diluted with 5 mL of distilled water before being titrated to a pink end point with 2.5 mM sulfuric acid. A standard curve was established using a stock N solution (equal concentrations ammonium sulphate and potassium nitrate). Olsen's available phosphate was measured according to the method described by Karla and Maynard (1991). Available phosphate was extracted from 2.5 g of air-dried tailings (passed through a 0.5 mm sieve) by adding 50 mL of 0.5M sodium bicarbonate (pH = 8.5) to the tailings, and shaking the suspension for 30 minutes on a Burrell wrist-action shaker. The suspension was then filtered through Whatman 42 filter paper prior to phosphate determination. The determination procedure required 10 mL of the extract, to which the following additions were made: 1.0 mL 2.5 M sulfuric acid, followed by 15.5 mL distilled water, 8 mL ammonium molybdonate solution (4.8 x 10^{-2} M) and antimony potassium tartarate (2.9 x 10^{-3} M) solution diluted by equal volume of sulfuric acid (2.5 M), and finally an additional 15.5 mL distilled water. After allowing the solution to stand for ten minutes, the absorbance was read at 882 nm with a spectrophotometer (Pharmacia Biotech, Ultrospec 2000). Phosphate content was determined using a standard curve prepared with a stock standard P solution of monopotassium phosphate. Changes in total metal content of the tailings were examined during the 2009 field season by way of ICP-MS analysis at the ALS Environmental Laboratories in Winnipeg, MB. The protocol employed an Aqua Regia (mixture of hydrochloric and nitric acids) digest to liberate metals from the sediment matrix before being quantifies through conventional ICP-MS methods. Cation exchange capacity was quantified two years following treatment application and compared to reference, unamended tailings. Tailings samples were sent to ALS Environmental Laboratories in Winnipeg, MB for analysis, where the sodium acetate saturation method was used for cation exchange capacity determination. In this determination, the soil/sediment sample was treated with 1M ammonium acetate to saturate the exchange sites with ammonium. After the excess ammonium was removed by applying isopropanol, the bound ammonium was leached from the exchange sites with 1M sodium chloride, and quantified with use of an autoanalyzer (Chapman, 1965). This value represents an approximation of cation exchange capacity. Aggregate size distribution was characterized two years following treatment application according to the methods described by Kemper & Rosenau (1986). Forty grams of air dried tailings were placed atop four stacked sieves (2, 1, 0.5 and 0.25 mm), and lowered into place until the water level of the upstroke was just breaching the surface of the top sieve. The sample was wetted by capillary action before undergoing five minutes of vertical oscillations (1.5 cm) at a rate of 30 cycles per minute. The aggregates remaining in each sieve was washed into tarred beakers and dried at 105 °C for 24 hours before being weighed. Moisture content was determined in sub-samples by oven drying the tailings at 105 °C for 24 hours.

3.7 Plant growth and stress

3.7.1 Sampling

Plant cover (%) as well as *Picea mariana* seedling survival rate (%) was assessed throughout the entire experimental period, beginning late in the first growing year (September, 2009). For the following field season, spring and fall *P. mariana* survival was documented, as well as during one additional sample date two years following the initial treatment application (June 2011). *P. mariana* seedlings were considered to be alive when > 5% of their foliage was still green; observations on plant injury and stress were also made during these survivorship assessments. In some cases, particularly when considering the papermill sludge amended plots, live seedlings were hidden from observation by the established ground cover vegetation. Plant cover was again assessed in September 2010 and July 2011 during which time un-seeded plant species were identified. During plant cover assessment, five randomly placed 1m x 1m quadrates were placed in each quarter plot and all plant species present were assessed for their contribution to the total ground cover. Above and below ground biomass harvests were made following the second year of the experiment in order to provide productivity estimates (August 26th, 2010). Harvests were not done during the first year in order to minimize the disturbance and encourage plant establishment and growth. Above ground tissues within a 0.25m x 0.25m quadrate were harvested, washed and separated based on species prior to being dried down and weighed. Below ground tissues were collected in a 393 cm^3 metal cylinder inserted 20 cm into the surface of the harvested area, separated from the tailings substrate before being washed, dried for 24 hours at 65°C, and weighed. *Salix* spp. survival and growth was not documented given the variability of the plant material and the obvious effect of the planting technique on establishment.

Tissues for physiological measurements were collected in the field on a sub-plot basis and stored on ice until the point at which they could be appropriately frozen or freeze dried with a freeze dryer (Labconco, Kansas City, MO, USA). Entire shoots of the grasses and alfalfa were collected on June 16th, 2010 from the Gunnar experimental plots, and kept refrigerated prior to their processing for pigment content analysis in the two days that followed their collection. Grass and alfalfa shoots were collected July 29th, 2010 frozen with liquid nitrogen, and stored at -80°C until they were analyzed for proline content. Shoot tissues were again collected in July 29th of 2011; however in this case they were freeze-dried for storage and preservation purposes prior to pigment content determination at a later date. Shoot tissues of *Medicago sativa* as well as *Agropyron trachycaulum* were collected in July 27th, 2011, and freeze-dried prior to being sent for elemental composition analysis to ALS Laboratories, Mississauga, ONT. During the July 27th, 2011 field trip *Medicago sativa* plants established on the experimental plots were selected to measure gas exchange. Additionally, the nitrogen fixation capacity of *M. sativa* plants was assessed on August 24th, 2011 during the final field trip of the season.

3.7.2 Analyses

For the pigment analysis, 0.10 g of fresh tissue from each species (*A. trachycaulum, F. rubra, M. sativa*) was finely chopped in 20 mL scintillation vials prior to having 2 mL aliquots of 80% acetone added and left incubate for at least three hours in the dark. Following the incubation, the extract was collected and stored in the dark. This process was repeated several (4-5) times before the collected acetone extracts, which has leached the pigments from the plant tissue, were quantified by reading the absorbance at 480, 645, and 663 nm in a spectrophotometer (Pharmacia Biotech, Ultraspec 2000). Pigment contents of the plant tissues were then determined with the following formulae (MacKinney, 1941; Sestak et al., 1971):

$$\text{Chl a} = 12.72 * A663 - 2.58 * A645$$

$$\text{Chl b} = 22.87 * A645 - 4.67 * A663$$

$$\text{Carotenoids} = A480 + 0.114 * A663 - 0.638 * A645.$$

Pigment analysis was again conducted during the 2011 field season, however in this case only *M. sativa* was collected for analysis, as the other species appeared to have begun dying back. For this sampling, dry tissue was used in order to see if there was an effect of increased water content on pigment concentrations. In this case, methanol was used as the extracting solvent introduced to the plant tissue; this protocol does not allow for the quantification of carotenoid contents. The tissue used for this analysis had been freeze-dried immediately upon its return from the field. Tissues were prepared and treated in the same manor as for the acetone extraction with the exception of the solvent used. The quantity of pigments is then derived using the following formulae (MacKinney, 1941; Sestak et al., 1971):

$$\text{Chl a} = 16.5 * A665 - 8.3 * A650$$

$$\text{Chl b} = 33.8 * A650 - 12.5 * A665.$$

Proline content determination was done using the protocol described by Sofo et al. (2004) on tissues that had been collected, frozen with liquid nitrogen upon their return from the field, and stored at -80°C. During this analysis, 0.5 g of fresh plant tissue was homogenized by grinding with a mortar and pestle in the presence of 1.75 g of sand as well as 5 mL 3% sulfosalicylic acid. In the case of the grass tissues, the uppermost 5 cm of the blades of grass was removed prior to collecting sufficient tissue from the remaining plant samples. Alternatively, for *M. sativa*, whole leaves were removed from stems for use in the analysis. This solution was then centrifuged for 15 minutes at 4900 g. The supernatant (1 mL) was then mixed with 2 mL of the reaction mixture (1% acetic ninhydrin in 60% acetic acid) and heated to 100°C for one hour using a dry bath incubator (Fisher Scientific). The reaction vessel was then cooled for five minutes in an ice bath, prior to being vortexed with 3 mL of toluene added to it. The absorbance of the solution containing the extracted proline (top phase) was then read at 520 nm, and quantified with the use of a standard curve made with proline.

Dried shoot tissues of *A. trachycaulum* as well as *M. sativa* were collected and freeze dried prior to being sent to Actlabs in Mississauga, ONT for elemental composition determination. Tissue samples (bulk samples collected from across experimental plot) were not ground prior to being sent for analysis; sample preparation was conducted by ALS laboratory. The protocol used by this lab employed an Aqua Regia digest followed by conventional ICP-MS methods to quantify many elements (15 g macerated aliquot analyzed).

Light response curves were created using a portable infrared gas analyzer (LICOR LI-6400 Portable Photosynthesis System) to measure photosynthetic rate (μmol CO_2 m^{-2} s^{-1}) at a range of light intensities (0, 20, 50, 100, 200, 500, 1000, and 1500 μmol photons m^{-2} s^{-1}) of *M. sativa* plants established on the various experimental plots. In addition, transpiration rate and stomatal conductance (use to estimate the degree of opening of the stomata,) were measured in response to the same light intensities. All measurements were taken between 11:00 – 13:00 on July 11th, 2011, the ambient temperature was between 21 – 24 °C, carbon dioxide reference levels were 400.0 ± 0.5 μL L^{-1}, and the flow rate was maintained at 400 mL min^{-1}. Total area of the leaves enclosed in the chamber was determined with a leaf area meter (LICOR LI-3100). Maximal photosynthetic rate was estimated using the following simple exponential function described by Chalker (1981):

$$P = P_m I/(I + I_k)$$

where P = gross photosynthesis

I = irradiance

P_m = photosynthetic capacity

I_k = horizontal asymtote.

Symbiotic nitrogen fixation was assessed using the acetylene reduction method described by Myrold et al., (1999). All assays were conducted between the hours of 11:30 – 14:30. During this assay, a portion of the *M. sativa* root tissue containing nodules was harvested and sealed in a 60 mL glass jar with a rubber seal cap. This vessel had 10% of its volume removed and replaced with acetylene prior to incubation within the root zone for approximately 1h. Sub-samples of the gas within the reaction vessel were collected and stored in vials that had their gas evacated, and returned to the University of Manitoba to be analyzed for gas content within the following days. Gas samples were analyzed with a Varian 3400 gas chromatograph with a 0.25 mL sampling valve and a Haysep T column (Varian Canada Inc., Edmonton, AB). Root tissue and nodules were washed and freeze-dried prior to having the number of nodules and their weight determined.

3.8 Statistical analysis:

The complete statistical model for this study included the amendment effect, the seeding effect, sample date and their interaction effects (full factorial), as well as the random effect of the sampling plot nested within treatment. As tailings sampling was not done on a sub-plot basis during the 2009 season, the analysis of this data did not include seeding as a model effect. Furthermore, since seeding treatment did not significantly affect any tailings property (except bulk density), the seeding treatment effect was removed from the statistical model so that the entire study period could be taken into account for these measures. Variables only measured one time during the study (water stable aggregate distribution, biomass harvest and all plant stress analyses) did not include the sample date as model effect. The analysis of nitrogen fixation and gas exchange in *Medicago sativa*, as well as elemental composition in *M. sativa* and *Agropyron trachycaulum* did not include the seeding model effect as plants were sampled across multiple sub-plots. For ease of interpretation, the effect of the four amendment treatment crossed with the four seeding treatments on plant cover was analyzed and presented on a year-by-year basis. To avoid problems of many zeros in a data set when *Picea mariana* survival reached zero across an entire amendment treatment, the treatment was excluded from further statistical analysis. Homogeneity of variance was verified for all data sets and only plant biomass yield data did not exhibit homogenous residuals. For this data set a log transformation was performed. Tukey's HSD test was employed as a measure of comparison of means ($\alpha = 0.05$) when significant model effects were detected in an ANOVA model. As a result of several 'below detection limit' values within the cation exchange data set, a Wilcoxon ranked sum test was used with the 'each pair method' used to compare means. All statistical analysis was done on JMP 8.0 statistical software package (SAS Institute Inc, Carey, North Carolina).

Chapter 4. Results

4.1 Tailings Fertility

4.1.1 Electrical conductivity

The amendment treatments did not significantly affect the electrical conductivity of the Gunnar mine tailings during the current study, and there was no noticeable effect of the seeding treatment on electrical conductivity (in 2010). The initial, pre-treatment electrical conductivity of the Gunnar mine tailings varied between approximately 2 – 4 dS m^{-1} across the experimental area (Table 4.1). The experimental plots that were randomly selected for the high fertilizer rate treatment had on average almost twice the conductivity levels as compared to all other experimental plots (3.99 ± 0.55 and 2.10 ± 0.27 dS m^{-1}, respectively). Conductivity levels fluctuated somewhat over the course of the study however after two field seasons the levels very nearly matched their pre-treatment values (Table 4.1). The greatest fluctuation was witnessed in the wood chips amendment treatment while the low fertilizer rate treatment saw the least fluctuation.

4.1.2 pH

The pH of the differently amended tailings was not affected by amendment application, nor did it change in any significant manor over the 2009 growing season (Table 4.2). The tailings pH remained consistently within a narrow range, between 7.14 to 7.34.

Table 4.1. Electrical conductivity (dS m^{-1}) of differently amended tailings over the 2009-10 field seasons; The ANOVA model showed no differences between treatments. ; Replicates are bulked composite subsamples (4) from across experimental plots (Mean ± SE; n = 5).

Treatment:	Electrical Conductivity (dS m^{-1})				
	June 2009*	July 2009	October 2009	May 2010	October 2010
Low Fertilizer	2.10 ± 0.64	1.98 ± 0.37	2.75 ± 0.51	2.08 ± 0.21	2.18 ± 0.48
High Fertilizer	3.99 ± 0.55	4.55 ± 0.79	4.78 ± 1.17	3.62 ± 0.37	4.82 ± 0.81
Wood Chips, low fertilizer	2.03 ± 0.39	2.16 ± 0.24	3.12 ± 0.71	2.07 ± 0.16	1.91 ± 0.28
Papermill Sludge, low fertilizer	2.17 ± 0.20	2.76 ± 0.33	3.08 ± 0.33	3.19 ± 0.72	2.26 ± 0.32

*Pre-treatment value

Table 4.2. pH of differently amended tailings over the 2009 field season; The ANOVA model showed no differences between treatments; Replicates are bulked composite subsamples (4) from across experimental plots (Mean ± SE; n = 5).

	pH		
Treatment:	June 2009*	July 2009	October 2009
Low Fertilizer	7.31±0.06	7.23±0.05	7.14±0.06
High Fertilizer	7.34±0.14	7.25±0.13	7.38±0.17
Wood Chips, low fertilizer	7.29±0.07	7.31±0.08	7.30±0.10
Papermill Sludge, low fertilizer	7.25±0.06	7.26±0.07	7.15±0.08

*Pre-treatment value

4.1.3 Organic carbon content

Over the course of the study, a significant interaction effect was observed between the amendment treatment and sample date, while the seeding treatment did not affect soil organic carbon at any point during the study. The unvegetated Gunnar mine tailings showed a low level of soil organic carbon (<0.29%) prior to amendment application (Figure 4.1). Regardless of the treatment, organic carbon content was not increased by amendment application as of one month following application. However, by the end of the 2009-field season, a significant increase of 0.17% in soil organic carbon was observed in the papermill sludge amended plots. No other treatment resulted in a significant increase in organic carbon after the first field season (Figure 4.1). A further increase in organic carbon was observed in sludge-amended plots over the 2010 field season, resulting in the highest levels observed throughout the experiment. Following the 2010 winter, we observed a statistically significant increase of 0.18% in soil organic carbon for the wood chips amended plots (Figure 4.1). This increase compared to all other changes in carbon content between sample dates was quite great, with a mean increase across seeding treatments of 0.19%. At the end of our two year experimental period, the two amendment treatments receiving organic additions (wood chips or papermill sludge) showed equivalent levels of soil organic carbon, levels which were significantly greater than that of the treatments that received no organic addition (Figure 4.1).

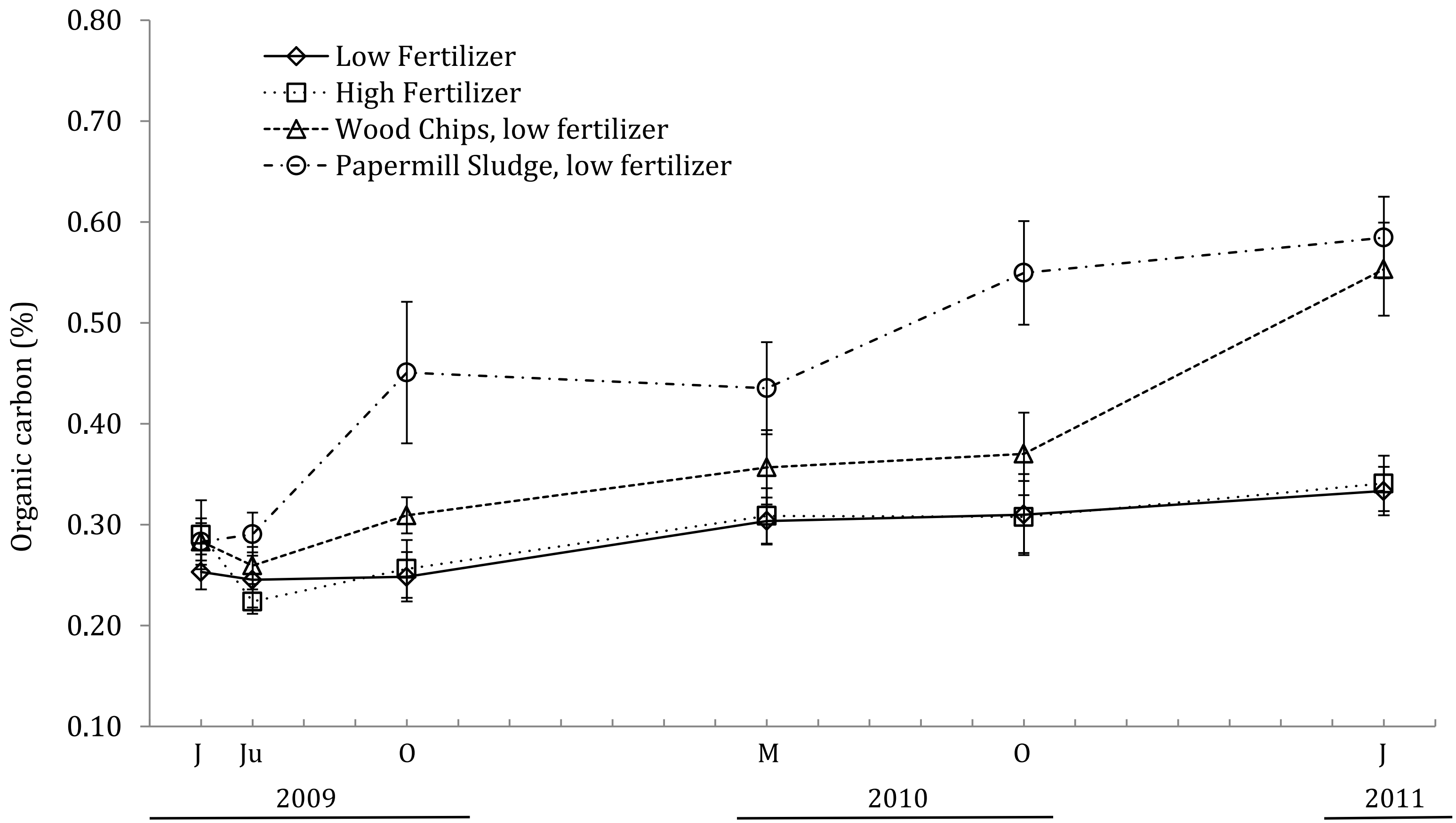

Figure 4.1. Organic carbon (%) in differently amended Gunnar mine tailings throughout 2009-2011 field seasons; sub-plot values averaged across amendment treatments, J: June, Ju: July, O: October, M: May (Mean ± SE; n = 5). Due to the number of treatment and date combinations, post hoc test differences are presented in a separate table.

Table 4.3. Statistically significant groupings of sample date x amendment treatment effects as found by Tukeys HSD test for organic carbon (α = 0.05). Treatment combinations with the same letters are not significantly different from one another.

Sample date	Amendment Treatment	Statistical Grouping
June 2009*	Low fertilizer	de
	High fertilizer	cde
	Wood chips + low fertilizer	cde
	Papermill sludge + low fertilizer	de
July 2009	Low fertilizer	e
	High fertilizer	e
	Wood chips + low fertilizer	de
	Papermill sludge + low fertilizer	cde
October 2009	Low fertilizer	de
	High fertilizer	de
	Wood chips + low fertilizer	cde
	Papermill sludge + low fertilizer	abcd
May 2010	Low fertilizer	cde
	High fertilizer	cde
	Wood chips + low fertilizer	cde
	Papermill sludge + low fertilizer	bc
October 2010	Low fertilizer	cde
	High fertilizer	cde
	Wood chips + low fertilizer	cde
	Papermill sludge + low fertilizer	a
June 2011	Low fertilizer	cde
	High fertilizer	cde
	Wood chips + low fertilizer	ab
	Papermill sludge + low fertilizer	a

*Pre-treatment values

4.1.4 Inorganic nitrogen content

A significant interaction effect was observed between the amendment treatment and sample date while the seeding treatment did not affect inorganic nitrogen at any point during the study. The unvegetated Gunnar tailings showed very low initial inorganic nitrogen content (~ 1 – 6 mg kg^{-1}). The input of inorganic fertilizer associated with all amendment treatments resulted in a significant increase of nitrogen one month following application, in July2009 (Figure 4.2). The high fertilizer amended plots showed the highest nitrogen levels, almost twice that of the low fertilizer plots (79.34 ± 11.95 and 34.38 ± 2.16 mg kg^{-1}. respectively) (Figure 4.2). At the end of the 2009 field season, the wood chips amended plots had returned to the initially low levels of inorganic nitrogen. The low fertilizer and papermill sludge amended plots had similar nitrogen levels at the end of the field season, which was still slightly elevated in comparison to the pre-treatment content. At the end of the first field season, the high fertilizer amended plots showed twice the inorganic nitrogen content than the low fertilizer and papermill sludge amended plots (42.36 ± 6.07 as compared to 20.27 ± 2.12 and 21.35 ± 2.71 mg kg^{-1}, respectively). Additionally, at this point in the 2009 field season, the high fertilizer amended plots contained more than four times more nitrogen than the plots amended with wood chips (42.36 ± 6.07 and 8.89 ± 4.38 mg kg^{-1}, respectively) (Figure 4.2).

In spring of 2010, the papermill sludge and high fertilizer amended plots had lost a significant amount of nitrogen, leading to levels similar to the pre-treatment tailings (Figure 4.2). High fertilizer amended plots had lost almost half of their

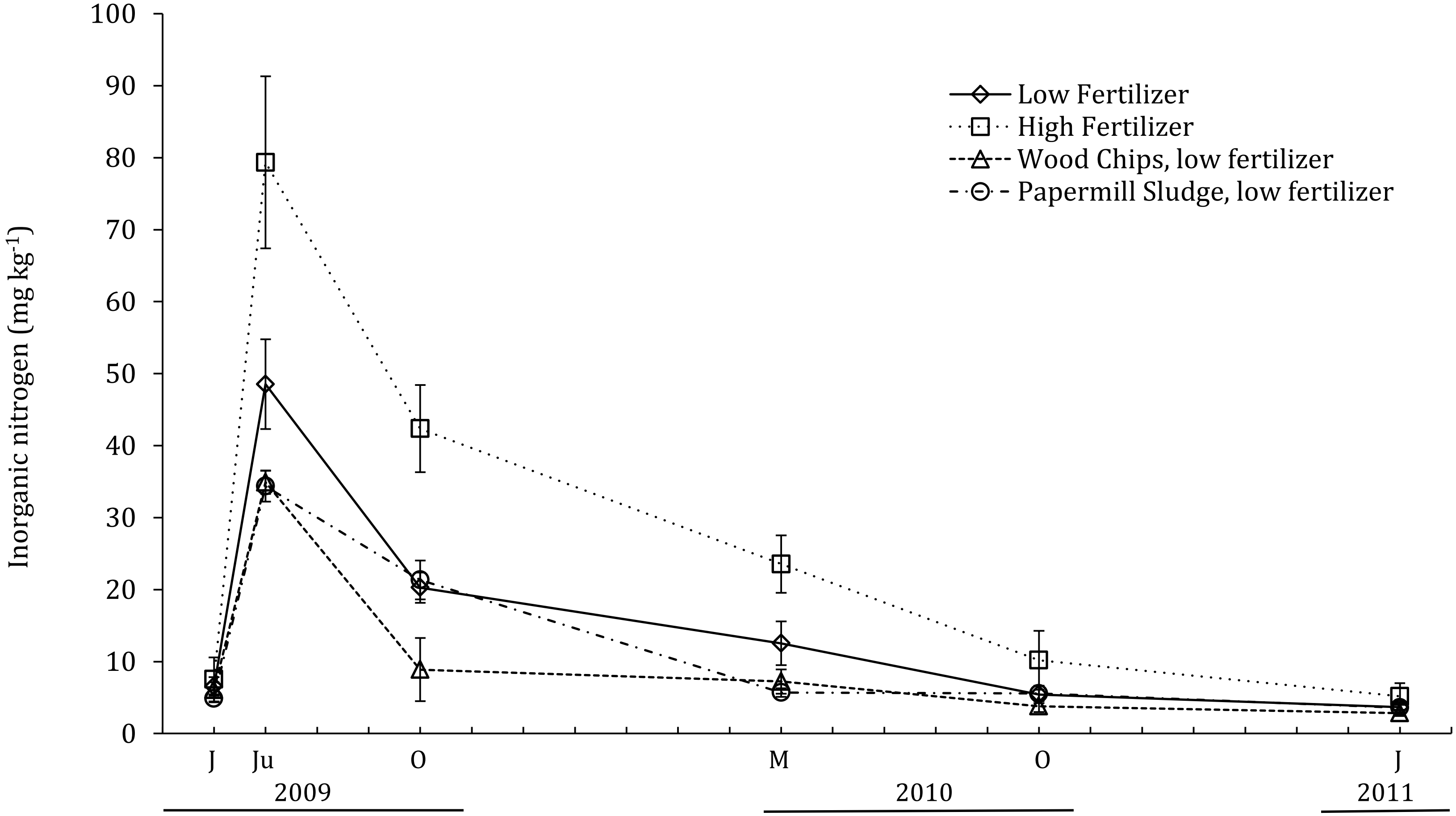

Figure 4.2. Inorganic nitrogen content (mg kg^{-1}) of differently amended Gunnar mine tailings throughout the 2009-2011 field seasons; sub-plot values averaged across amendment treatments J: June, Ju: July, O: October, M: May (Mean ± SE; n = 5).

Table 4.4. Statistically significant groupings of sample date x amendment treatment effects as found by Tukeys HSD test for inorganic nitrogen (α = 0.05).

Sample date	Amendment Treatment	Statistical Grouping
June 2009*	Low fertilizer	ef
	High fertilizer	def
	Wood chips + low fertilizer	def
	Papermill sludge + low fertilizer	ef
July 2009	Low fertilizer	b
	High fertilizer	a
	Wood chips + low fertilizer	bc
	Papermill sludge + low fertilizer	bc
October 2009	Low fertilizer	cd
	High fertilizer	b
	Wood chips + low fertilizer	def
	Papermill sludge + low fertilizer	cd
May 2010	Low fertilizer	de
	High fertilizer	c
	Wood chips + low fertilizer	ef
	Papermill sludge + low fertilizer	ef
October 2010	Low fertilizer	f
	High fertilizer	def
	Wood chips + low fertilizer	ef
	Papermill sludge + low fertilizer	ef
June 2011	Low fertilizer	f
	High fertilizer	ef
	Wood chips + low fertilizer	ef
	Papermill sludge + low fertilizer	ef

*Pre-treatment values

remaining nitrogen from the previous fall (from 42.36 ± 6.07 to 23.56± 2.00 mg kg^{-1}), but still showed an elevated level of inorganic nitrogen as compared to initial values. After two full experimental years (June 2011), all plots had returned to their initial, unamended level of inorganic nitrogen ranging between 2.5 and 6.0 mg kg^{-1}.

4.1.5 Available phosphate content

Over the course of the study, a significant interaction effect was observed between the amendment treatment and sample date while the seeding treatment did not affect soil available phosphate at any point during the study. The initially low available phosphate levels of the Gunnar mine tailings (<0.35 mg kg^{-1}) were quickly increased following the application of the high fertilizer rate and the papermill sludge (July 2009, Figure 4.3). The low fertilizer and wood chips amended plots showed no increase in available phosphate over the 2009 field season. By spring of 2010, there was no significant difference in phosphate between amendments. Looking at the data over the entirety of the two-year experimental period, there was an overall trend of gradually decreasing soil phosphate, with the papermill sludge and high fertilizer plots showing an initial pulse following amendment application (Figure 4.3).

4.1.6 Available potassium content

There was a significant interaction between sample dates and amendment treatments during the 2009 field season. Initial levels of available potassium in the Gunnar mine tailings ranged from approximately 80 – 120 mg kg^{-1} (Table 4.6).

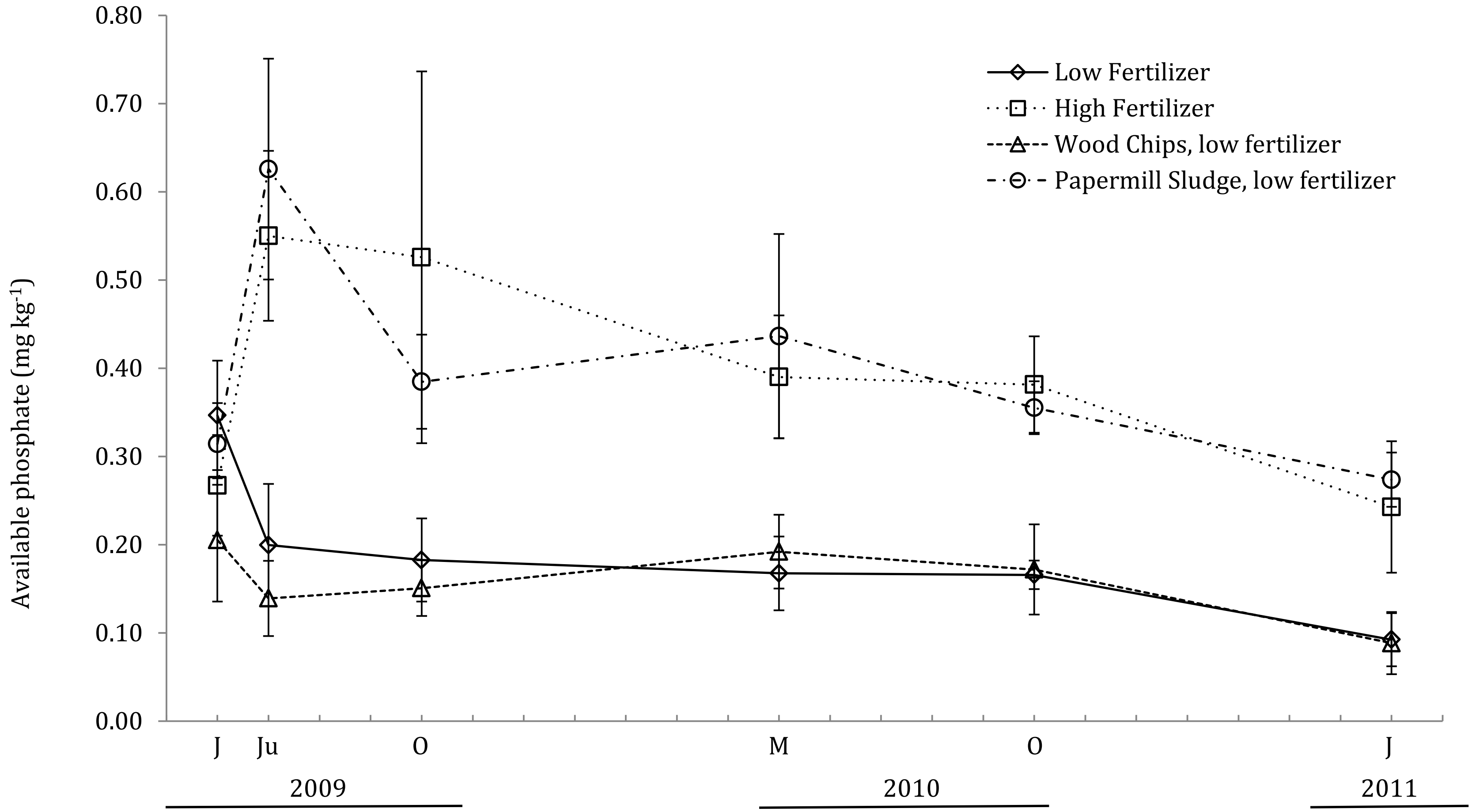

Figure 4.3. Available phosphate content (mg kg^{-1}) of differently amended Gunnar mine tailings throughout the 2009-2011 field seasons; sub-plot values averaged across amendment treatments; J: June, Ju: July, O: October, M: May (Mean ± SE, n = 5).

Table 4.5. Statistically significant groupings of sample date x amendment treatment effects as found by Tukeys HSD test for available phosphate (α = 0.05).

Sample date	Amendment Treatment	Statistical Grouping
June 2009*	Low fertilizer	abcdefgh
	High fertilizer	bcdefgi
	Wood chips + low fertilizer	defghi
	Papermill sludge + low fertilizer	bcdefghi
July 2009	Low fertilizer	defghi
	High fertilizer	ab
	Wood chips + low fertilizer	fghi
	Papermill sludge + low fertilizer	a
October 2009	Low fertilizer	defghi
	High fertilizer	abc
	Wood chips + low fertilizer	fghi
	Papermill sludge + low fertilizer	abcdefg
May 2010	Low fertilizer	ghi
	High fertilizer	abce
	Wood chips + low fertilizer	fghi
	Papermill sludge + low fertilizer	abde
October 2010	Low fertilizer	ghi
	High fertilizer	abcdef
	Wood chips + low fertilizer	ghi
	Papermill sludge + low fertilizer	bcdefg
June 2011	Low fertilizer	i
	High fertilizer	dfghi
	Wood chips + low fertilizer	hi
	Papermill sludge + low fertilizer	cfghi

*Pre-treatment values

Table 4.6. Available potassium content (mg kg^{-1}) of differently amended Gunnar mine tailings over the 2009 field season; interaction of sample date and amendment found is represented by levels not connected by the sample letter; replicates are bulked subsamples from across experimental plots (Mean ± SE; n = 5).

	Available potassium (mg kg^{-1})		
Treatment:	June 2009*	July 2009	October 2009
Low Fertilizer	90.4 ± 19.4 ^{b}	123.0 ± 21.1 ^{b}	97.0 ± 19.3 ^{b}
High Fertilizer	120.8 ± 17.6 ^{b}	248.4 ± 35.6 ^{a}	186.4 ± 37.2 ^{ab}
Wood Chips, low fertilizer	90.0 ± 15.6 ^{b}	124.6 ± 7.0 ^{ab}	146.0 ± 17.6 ^{b}
Papermill Sludge, low fertilizer	78.8 ± 18.6 ^{b}	137.4 ± 10.9 ^{b}	116.8 ± 28.3 ^{ab}

*Pre-treatment values

Following treatment application (June 2009), the high fertilizer amended plots showed a significant increase in available potassium (Table 4.6). By the end of the first field season, the available potassium of all amended plots had returned to levels similar to unamended tailings.

4.1.7 Total metals

The sample date as well as the interaction of amendment and sample date affected the content of several elements within the Gunnar mine tailings during the 2009 field season. Al, Cr, Mn, Mg, Ni, Ti, and Zn showed an overall decrease in content over the 2009 field season. Conversely, the levels of Cu, Na and Mo increased by the end on the 2009 field season (Table 4.7). Additionally, the Fe and K content of the amended tailings showed change over the 2009 field season; however the initial values and the values at the end of the field season were similar (Table 4.7). An interaction between sample dates and amendment treatments was observed for magnesium, molybdenum, and sodium. In the case of magnesium, the low fertilizer amended plots showed a decrease in total content from the pre-treatment to the October 2009 sample date (Table 4.7), while the molybdenum content of the low fertilizer amended plots significantly increased between the pre-treatment and October 2009 sample dates (Table 4.7). When considering the sodium content of the differently amended tailings over the 2009 field season, we found that the high fertilizer amended plots exhibited significantly elevated levels following amendment application (July 2009), and this elevated level persisted through the October 2009 sample date (Table 4.7).

Table 4.7. Changes in elemental content of differently amended tailings over 2009 growing season. Letters connecting amendment treatment values are used when only the sampling date is significant; letters beside individual amendment values when there is an interation between amendment and sample date (Mean ± SE; n = 5).

Element	Sample Date					
Amendment	Pre-treatment		July 2009		October 2009	
Aluminum (Al)						
Low Fertilizer	27,240 ± 1,150	a	27,000 ± 1,400	a	21,260 ± 2,002	b
High Fertilizer	26,040 ± 3,026		26,260 ± 2,111		22,280 ± 1,642	
Wood Chips, low fertilizer	27,920 ± 394		25,500 ± 1,440		23,500 ± 1,220	
Papermill Sludge, low fertilizer	22,790 ± 1,008		25,780 ± 1,410		24,640 ± 1,562	
Arsenic (As)						
Low Fertilizer	24.0 ± 3.8		23.7 ± 5.1		43.2 ± 15.3	
High Fertilizer	36.5 ± 14.4		29.9 ± 13.7		35.5 ± 13.1	
Wood Chips, low fertilizer	20.6 ± 3.3		22.2 ± 5.5		21.2 ± 2.6	
Papermill Sludge, low fertilizer	26.3 ± 9.0		27.2 ± 7.6		19.5 ± 3.5	
Cadmium (Cd)						
Low Fertilizer	1.23 ± 0.13		1.25 ± 0.16		1.23 ± 0.02	
High Fertilizer	1.11 ± 0.12		1.04 ± 0.11		1.05 ± 0.15	
Wood Chips, low fertilizer	1.33 ± 0.22		1.31 ± 0.22		1.09 ± 0.12	
Papermill Sludge, low fertilizer	0.98 ± 0.12		1.05 ± 0.09		1.08 ± 0.12	
Calcium (Ca)						
Low Fertilizer	60,500 ± 1,500		62,100 ± 800		56,800 ± 2,700	
High Fertilizer	59,400 ± 3,500		60,200 ± 2,200		59,300 ± 2,100	
Wood Chips, low fertilizer	61,500 ± 2,200		63,200 ± 2,600		62,200 ± 1,500	
Papermill Sludge, low fertilizer	52,600 ± 1,600		61,200 ± 900		61,000 ± 1,400	

* Pre-treatment values

Table 4.7. Continued...

Element Amendment	Sample Date June 2009*		July 2009		October 2009	
Chromium (Cr) (mg kg^{-1})						
Low Fertilizer	85 ± 3		92 ± 4		74 ± 6	
High Fertilizer	84 ± 9		92 ± 7		81 ± 7	
Wood Chips, low fertilizer	88 ± 1	ab	90 ± 4	a	82 ± 4	b
Papermill Sludge, low fertilizer	73 ± 3		89 ± 4		88 ± 5	
Cobalt (Co) (mg kg^{-1})						
Low Fertilizer	37.4 ± 1.6		35.9 ± 1.6		40.1 ± 2.1	
High Fertilizer	39.4 ± 4.1		36.3 ± 2.7		38.6 ± 4.9	
Wood Chips, low fertilizer	40.0 ± 3.2		39.7 ± 3.8		37.1 ± 1.8	
Papermill Sludge, low fertilizer	31.3 ± 0.7		36.3 ± 1.7		36.3 ± 0.7	
Copper (Cu) (mg kg^{-1})						
Low Fertilizer	194 ± 20		223 ± 25		253 ± 21	
High Fertilizer	177 ± 23		198 ± 18		222 ± 32	
Wood Chips, low fertilizer	206 ± 28	b	231 ± 28	ab	215 ± 10	a
Papermill Sludge, low fertilizer	165 ± 10		195 ± 9		235 ± 35	
Iron (Fe) (mg kg^{-1})						
Low Fertilizer	54,840 ± 1,940		59,040 ± 2,200		51,460 ± 2,480	
High Fertilizer	53,820 ± 3,800		56,940 ± 2,270		51,720 ± 2,760	
Wood Chips, low fertilizer	56,140 ± 1,460		57,280 ± 1,910		52,120 ± 1,830	
Papermill Sludge, low fertilizer	45,630 ± 2,030		54,400 ± 5,340		53,200 ± 2,380	
Lead (Pb) (mg kg^{-1})						
Low Fertilizer	10.7 ± 1.2		11.9 ± 1.1		12.6 ± 2.4	
High Fertilizer	13.6 ± 1.7		14.3 ± 2.1		14.2 ± 3.2	
Wood Chips, low fertilizer	10.1 ± 0.7		12.6 ± 0.9		11.3 ± 1.4	
Papermill Sludge, low fertilizer	10.5 ± 1.1		13.3 ± 1.5		13.5 ± 1.5	

* Pre-treatment values

Table 4.7. Continued...

Element Amendment	Sample Date June 2009*	 July 2009	 October 2009
Magnesium (Mg) (mg kg^{-1})			
Low Fertilizer	20,380 ± 580[a]	20,960 ± 810[a]	16,640 ± 1,700[b]
High Fertilizer	20,200 ± 2,110[ab]	21,220 ± 1,460[ab]	19,040 ± 1,390[ab]
Wood Chips, low fertilizer	20,620 ± 380[ab]	19,800 ± 1,000[ab]	19,320 ± 990[ab]
Papermill Sludge, low fertilizer	17,270 ± 830[ab]	21,200 ± 1,450[ab]	20,620 ± 1,070[ab]
Manganese (Mn)(mg kg^{-1})			
Low Fertilizer	1,057 ± 29	1,170 ± 82	962 ± 65
High Fertilizer	1,047 ± 74	1,081 ± 60	971 ± 44
Wood Chips, low fertilizer	1,066 ± 24 b	1,112 ± 36 a	1,041 ± 31 c
Papermill Sludge, low fertilizer	882 ± 43	1,096 ± 36	1,031 ± 36
Molybdenum (Mo) (mg kg^{-1})			
Low Fertilizer	0.95 ± 0.30[b]	1.21 ± 0.39[ab]	1.80 ± 0.75[a]
High Fertilizer	1.57 ± 0.48[ab]	1.73 ± 0.47[ab]	1.50 ± 0.40[ab]
Wood Chips, low fertilizer	0.68 ± 0.08[ab]	0.88 ± 0.11[ab]	0.74 ± 0.11[ab]
Papermill Sludge, low fertilizer	0.78 ± 0.12[ab]	1.07 ± 0.21[ab]	0.91 ± 0.14[ab]
Nickel (Ni) (mg kg^{-1})			
Low Fertilizer	70.4 ± 2.9	67.6 ± 2.6	65.6 ± 2.7
High Fertilizer	69.2 ± 4.0	66.8 ± 2.6	66.4 ± 4.6
Wood Chips, low fertilizer	74.9 ± 3.1	72.5 ± 3.3	66.6 ± 2.2
Papermill Sludge, low fertilizer	58.7 ± 1.9	67.1 ± 2.3	67.9 ± 1.7
Phosphorous (P) (mg kg^{-1})			
Low Fertilizer	200 ± 0	200 ± 0	200 ± 0
High Fertilizer	200 ± 0	200 ± 0	200 ± 0
Wood Chips, low fertilizer	200 ± 0	200 ± 0	200 ± 0
Papermill Sludge, low fertilizer	200 ± 0	200 ± 20	300 ± 20

* Pre-treatment values

Table 4.7. Continued...

Element Amendment	June 2009*		July 2009		October 2009	
	Sample Date					
Potassium (K) ($mg\ kg^{-1}$)						
Low Fertilizer	430 ± 40		500 ± 40		400 ± 40	
High Fertilizer	450 ± 40		640 ± 50		480 ± 40	
Wood Chips, low fertilizer	430 ± 20		490 ± 20		440 ± 40	
Papermill Sludge, low fertilizer	360 ± 30		530 ± 50		460 ± 60	
Sodium (Na) ($mg\ kg^{-1}$)						
Low Fertilizer	50 ± 40^{abc}		160 ± 32^{abc}		160 ± 59^{abc}	
High Fertilizer	150 ± 75^{c}		550 ± 220^{a}		490 ± 220^{ab}	
Wood Chips, low fertilizer	40 ± 18^{bc}		130 ± 20^{abc}		260 ± 86^{abc}	
Papermill Sludge, low fertilizer	70 ± 34^{abc}		140 ± 15^{abc}		170 ± 35^{abc}	
Titanium (Ti) ($mg\ kg^{-1}$)						
Low Fertilizer	1011 ± 90		1027 ± 83		824 ± 89	
High Fertilizer	923 ± 130		1097 ± 126		838 ± 89	
Wood Chips, low fertilizer	1102 ± 30	a	1042 ± 62	a	926 ± 61	b
Papermill Sludge, low fertilizer	860 ± 35		1010 ± 85		950 ± 71	
Vanadium (V) ($mg\ kg^{-1}$)						
Low Fertilizer	120 ± 8		132 ± 8		103 ± 14	
High Fertilizer	112 ± 14		124 ± 11		106 ± 12	
Wood Chips, low fertilizer	122 ± 3	b	125 ± 8	a	115 ± 8	b
Papermill Sludge, low fertilizer	98 ± 7		122 ± 9		119 ± 10	
Zinc (Zn) ($mg\ kg^{-1}$)						
Low Fertilizer	190 ± 20		200 ± 20		180 ± 5	
High Fertilizer	180 ± 10		170 ± 10		160 ± 10	
Wood Chips, low fertilizer	220 ± 40	a	200 ± 20	ab	170 ± 10	b
Papermill Sludge, low fertilizer	150 ± 20		170 ± 10		180 ± 10	

* Pre-treatment values

4.1.8 Cation exchange capacity

Both sample date as well as amendment treatments significantly affected cation exchange capacity, though no interaction effect was found between these model effects. In 2010, the papermill sludge amended tailings exhibited a cation exchange capacity greater than all other amendment treatments as well as the unamended tailings (Figure 4.4). All other amendment treatments showed cation exchange capacities equal to that of the unamended tailings. One year later, at the 2011 sample date, the cation exchange capacity of both organically amended treatments (papermill sludge and wood chips) was greater than that of the other amendment treatments and the unamended mine tailings (Figure 4.4).

4.1.9 Water stable aggregate distribution

The amendment treatment model effect significantly influenced total macro-aggregation at the time of sampling (2010) while seeding treatments did not affect tailings aggregation. The total amount of macro-aggregates in the organically amended mine tailings was significantly greater than both the unamended mine tailings as well as tailings that did not receive organic addition as part of their amendment (Table 4.8). Total macro-aggregation witnessed in the papermill sludge amended tailings and the wood chips amended tailings were twice that of the other amendment treatments (19.6 ± 2.4 and 10.7 ± 1.5, respectively). The total macro-

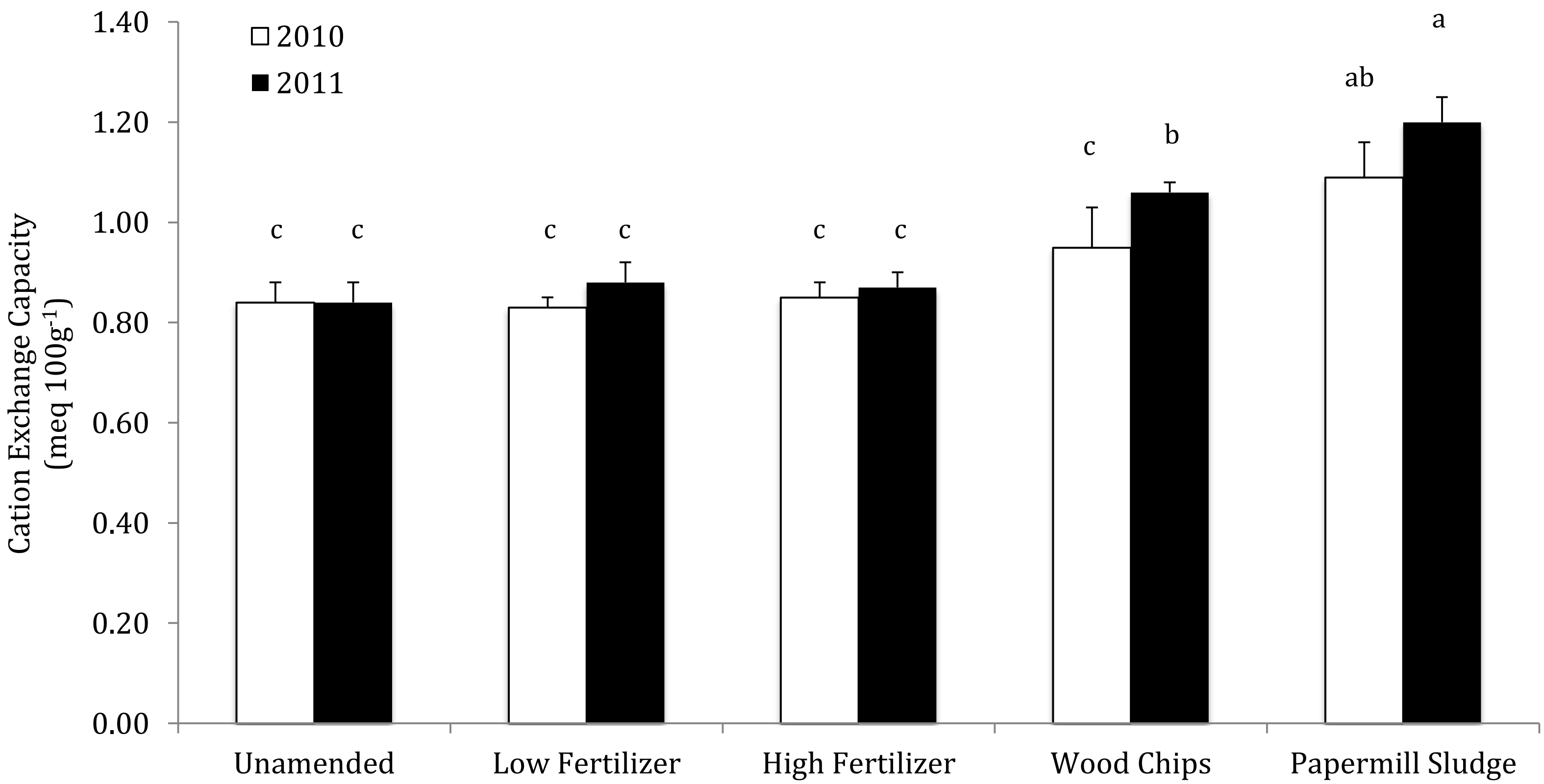

Figure 4.4. Cation exchange capacity (meq $100g^{-1}$) of unamended and amended Gunnar mine tailings over the 2010-2011 field seasons; values not connected by the same letter are significantly different by Wilcoxon 'each pair' method (Mean ± SE; n = 5).

Table 4.8. Water-stable macro-aggregate (>0.25mm) distribution (%) of amended Gunnar mine tailings in the fall of 2010; within a size class levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 5).

	Aggregate size classes				
Amendment	0.25-0.5 mm	0.5-1.0 mm	2.0-1.0 mm	>2.0 mm	Total Aggregates
Low Fertilizer	2.3 ± 0.3 ^{b}	1.4 ± 0.5 ^{a}	1.5 ± 0.3 ^{c}	5.9 ± 1.9 ^{bc}	11.1 ± 2.5 ^{b}
High Fertilizer	3.0 ± 0.7 ^{ab}	1.4 ± 0.4 ^{a}	1.8 ± 0.3 ^{bc}	4.0 ± 1.0 ^{c}	10.2 ± 1.9 ^{b}
Wood Chips, low fertilizer	3.4 ± 0.6 ^{ab}	2.1 ± 0.6 ^{a}	2.5 ± 0.5 ^{ab}	9.6 ± 2.4 ^{ab}	17.5 ± 3.5 ^{a}
Papermill Sludge, low fertilizer	4.1 ± 0.5 ^{a}	2.4 ± 0.5 ^{a}	3.1 ± 0.5 ^{a}	12.0 ± 2.9 ^{a}	21.6 ± 3.3 ^{a}
Unamended tailings	4.7 ± 1.3 ^{a}	1.8 ± 0.3 ^{a}	2.7 ± 0.7 ^{bc}	0.6 ± 0.2 ^{d}	9.7 ± 1.2 ^{b}

aggregation observed in the amendment treatments that did not receive organic amendment was equal to that of the unamended Gunnar mine tailings (Table 4.8). All amendment treatments saw an increase in the amount of aggregates in the > 2.0 mm size class as compared to the unamended mine tailings. The > 2.0 mm aggregate size class was characterized by the most variability in terms of proportion of total macro-aggregates, ranging from 0.6 ± 0.2 in unamended tailings to 12.0 ± 2.9 in papermill sludge amended tailings (Table 4.8).

4.1.10 Bulk Density

Both sample date as well as seeding treatment model effects affected tailings bulk density while amendment treatment did not significantly influence bulk density over the course of the current study. Because amendment treatment did not influence the experimental model, data presented in Figure 4.5 was bulked across amendment treatments. Over the course of the experiment, the bulk density of the amended mine tailings was significantly decreased relative to the unamended Gunnar mine tailings (Figure 4.5). Bulk density had decreased from an average of 1.40 ± 0.01 g cm^{-3} to 1.35 ± 0.01 g cm^{-3} one year after amendment application and to 1.31 ± 0.01 g cm^{-3} two years after amendment application. In addition, sub-plots that were planted with *Medicago sativa* and grasses exhibited lower bulk densities than those sub-plots that received no seeding at all across the 2010 and 2011 seasons (Figure 4.5). The seeding treatments that included either *M. sativa* or the grasses (but not both) had bulk density levels that were intermediate between the other two seeding treatments (Figure 4.5).

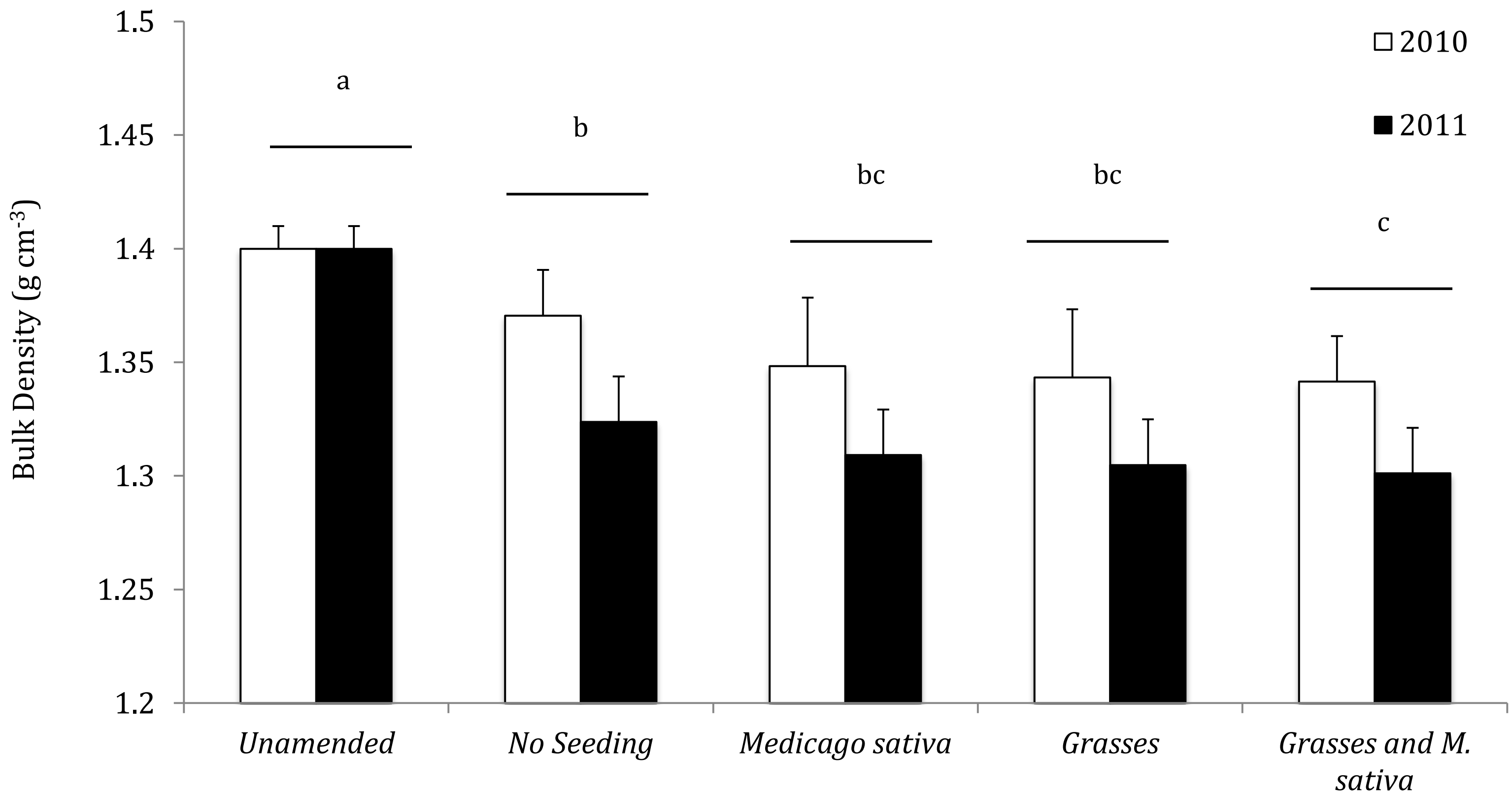

Figure 4.5. Bulk density (g cm^{-3}) of Gunnar mine tailings over the 2010-2011 field seasons; a significant decrease in bulk density was observed across amendments over time and bulk density in subplots seeded with grasses and alfalfa decreased more than un-seeded sub-plots; different letters represent the effect of the seeding treatment (Mean ± SE; n = 5).

4.2 Plant Growth

4.2.1 Plant cover

The plant cover established over the course of the two-year field experiment was significantly affected by the amendment and seeding treatments and an interaction between these treatments was observed.

In 2009, different cover levels were established depending on seed mix as well as the amendment treatment applied (Table 4.9; Figures 4.6 - 4.9). *Medicago sativa* was not observed in sub-plots where it was not seeded. *M. sativa* established its greatest ground cover in papermill sludge amended plots, when it was seeded alone or in the mix with grasses (14.5 ± 6.3 and 13.1 ± 3.2%, respectively). For the three other amendment treatments, there was no difference in *M. sativa* ground cover between sub-plots, regardless of being seeded or not. As for *Festuca rubra*, when this grass was seeded, papermill sludge amended plots allowed the greatest ground cover (Table 4.9). The ground cover established in the papermill sludge amended plots seeded with a grass mix was approximately twice that found in the wood chips amended plots (average of 24% vs. 13%, respectively). *F. rubra* ground cover was lowest in the low fertilizer and high fertilizer plots (below 10%) across all seeding treatments, as well as the sub-plots of the papermill sludge and wood chips amended plots not seeded with *F. rubra* (in all cases, 0% cover) (Table 4.9). As for *Agropyron trachycaulum*, plots amended by papermill sludge allowed approximately twice the ground cover as compared to wood chips and low fertilizer amended plots and four times greater cover values as compared to the high fertilizer plots.

Table 4.9. Ground cover (%) of seeded (*M. sativa, F. rubra* and *A. trachycaulum*) and unseeded (other) plants present on amended Gunnar mine tailings, 4 months after planting; significant interaction effect of amendment and seeding treatments found by Tukeys HSD test (α = 0.05) is represented by levels not connected by the same letter; analyses performed on each plant species individually (Mean ± SE; n = 5).

Amendment	Ground Cover (%)				
Seeded plants	*Medicago sativa*	*Festuca rubra*	*Agropyron trachycaulum*	Other	Total
Low fertilizer					
None	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	0.2 ± 0.3 ^{bc}	0.2 ± 0.3 ^{d}
M. sativa	1.2 ± 0.6 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{c}	1.2 ± 0.6 ^{d}
Grasses	0.0 ± 0.0 ^{c}	9.1 ± 2.2 ^{cde}	16.0 ± 2.1 ^{bcd}	0.0 ± 0.0 ^{c}	25.1 ± 3.9 ^{bc}
Grasses and M. sativa	1.4 ± 1.0 ^{c}	4.6 ± 1.2 ^{de}	15.3 ± 2.3 ^{bcd}	0.0 ± 0.0 ^{c}	21.3 ± 3.2 ^{bcd}
High fertilizer					
None	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{d}
M. sativa	0.1 ± 0.1 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{c}	0.1 ± 0.7 ^{d}
Grasses	0.0 ± 0.0 ^{c}	1.8 ± 0.8 ^{e}	8.1 ± 1.9 ^{de}	0.0 ± 0.0 ^{c}	9.9 ± 2.5 ^{cd}
Grasses and M. sativa	0.1 ± 0.1 ^{c}	1.1 ± 0.7 ^{e}	9.2 ± 2.8 ^{cde}	0.0 ± 0.0 ^{c}	10.3 ± 3.2 ^{cd}
Wood chips, low fertilizer					
None	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	1.2 ± 0.3 ^{bc}	1.2 ± 0.7 ^{d}
M. sativa	6.2 ± 3.0 ^{abc}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	1.7 ± 0.7 ^{b}	7.9 ± 3.7 ^{cd}
Grasses	0.0 ± 0.0 ^{c}	14.0 ± 2.7 ^{bc}	20.3 ± 1.2 ^{bc}	0.0 ± 0.0 ^{c}	34.3 ± 3.8 ^{b}
Grasses and M. sativa	3.3 ± 1.0 ^{bc}	12.3 ± 4.0 ^{bcd}	22.5 ± 5.1 ^{b}	0.0 ± 0.0 ^{c}	38.0 ± 10.0 ^{b}
Papermill sludge, low fertilizer					
None	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	4.8 ± 0.9 ^{a}	4.8 ± 0.9 ^{cd}
M. sativa	14.5 ± 6.3 ^{a}	0.0 ± 0.0 ^{e}	0.0 ± 0.0 ^{e}	5.2 ± 1.7 ^{a}	19.7 ± 8.1 ^{bcd}
Grasses	0.0 ± 0.0 ^{c}	28.5 ± 3.3 ^{a}	44.5 ± 1.4 ^{a}	1.8 ± 0.8 ^{b}	74.8 ± 3.9 ^{a}
Grasses and M. sativa	13.1 ± 3.2 ^{ab}	20.0 ± 3.2 ^{ab}	40.8 ± 5.2 ^{a}	1.0 ± 0.5 ^{b}	74.9 ± 10.8 ^{a}

Figure 4.6. Plant growth on a papermill sludge amended plot late in the first growing season (2009).

Figure 4.7. Plant growth on a wood chips amended plot late in the first growing season.

Figure 4.8. Plant growth on a low fertilizer rate amended plot late in the first growing season.

Figure 4.9. Plant growth on a high fertilizer rate amended plot late in the first growing season.

The wood chips amended plots allowed for greater establishment of *A. trachycaulum* as compared to the high fertilizer plots. Within the first growing season, only papermill sludge amended sub-plots where grasses were not seeded allowed significant establishment of volunteer plants (Table 4.9). Table 4.10 presents the unseeded plant species that were able to establish within the amended experimental plots over the course of the study. These unseeded plant species are largely herbaceous pioneers in nature, and include members of the Compositae, Poaceae, Fabaceae, Equisetaceae, and Rosaceae families (Table 4.10). Across all amendment and seeding treatments it is clear that papermill sludge amended sub-plots seeded with the grass mix promoted the greatest total plant ground cover within the first growing season. When grasses were included in the seeding treatment, significantly higher plant cover was observed.

In the 2010 field season, fairly similar trends in plant cover establishment were observed as compared to the 2009 season (Tables 4.9 and 4.11). Papermill sludge amended sub-plots seeded with *M. sativa* promoted cover establishment, while no other amendment treatment allowed for the establishment of statistically significant levels of *M. sativa* cover. Similarly, papermill sludge amended sub-plots seeded with grasses showed greater *F. rubra* establishment as compared to all other amended tailings. Establishment of *F. rubra* was not affected by seeding treatment in the high fertilizer amended plots where little establishment occurred. *Agropyron trachycaulum* cover was greatest in papermill sludge amended sub-plots where it was seeded; all other amendment sub-plots where it was seeded saw equivalent cover establishment (Table 4.11). Establishment of non-seeded plants rose

Table 4.10. Plant species growing in experimental plots during the 2009-2011-field season at the Gunnar gold mine tailings.

Family	*Plant species*	Common name
Amaranthaceae	*Chenopodium rubrum L.*	Red goosefoot
	Chenopodium glaucum L.	Oak-leaved goosefoot
Celastraceae	*Parnassia palustris L.*	Bog star
Compositae	*Aster brachyactis L.*	Rayless aster
	Aster ericoides L.	Heath aster
	Aster simplex Willd.	Panicled aster
	Chrysanthemum leucanthemum Lam.	Ox-eye daisy
	Senecio vulgares L.	Common groundsel
	Solidago canadensis L.	Canada goldenrod
	Solidago graminifolia L.	Flat-topped goldenrod
	Solidago nemoralis Aiton.	Gray goldenrod
	Sonchus arvensis L.	Perennial sow thistle
	Taraxacum officinale F.H. Wigg	Common dandelion
Cyperaceae	*Carex viridula Michx.*	Green sedge
	Carex aurea Nutt.	Golden sedge
Equisetaceae	*Equisetum arvense L.*	Common horsetail
Fabaceae	*Trifolium pratense L.*	Red clover
	Trifolium repens L.	White clover
	Trifolium hybridum L.	Alsike clover
	Melilotus alba Medik.	White sweet clover
	Melilotus officinalis L.	Yellow sweet clover
	Medicago lupulina L.	Yellow trefoil
Juncaceae	*Juncus alpina Vill.*	Alpine rush
Plantaginaceae	*Plantago major L.*	Common plantain
Poaceae	*Hordeum jubatum L.*	Fox-tail barley
	Agrostis stolonifera L.	Red top
	Poa palustris L.	Fowl bluegrass
Rosaceae	*Potentilla norvegica L.*	Rough cinquefoil

Table 4.11. Ground cover (%) of seeded (*M. sativa, F. rubra* and *A. trachycaulum*) and unseeded (other) plants present on amended Gunnar mine tailings,13 months after planting; significant interaction effect of amendment and seeding treatments found by Tukeys HSD test (α = 0.05) is represented by levels not connected by the same letter; analyses performed on each plant species individually (Mean ± SE; n = 5).

Amendment	Ground Cover (%)				
Seeded plants	*Medicago sativa*	*Festuca rubra*	*Agropyron trachycaulum*	Other	Total
Low fertilizer					
None	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	17.3 ± 7.3 ^{bc}	17.3 ± 7.3 ^{c}
M. sativa	1.5 ± 1.2 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	16.6 ± 3.0 ^{bc}	18.1 ± 3.7 ^{c}
Grasses	0.0 ± 0.0 ^{b}	11.0 ± 1.9 ^{cd}	20.5 ± 3.1 ^{b}	3.6 ± 1.2 ^{c}	35.1 ± 3.0 ^{bc}
Grasses and M. sativa	1.8 ± 0.4 ^{b}	11.4 ± 4.0 ^{cd}	21.5 ± 2.3 ^{b}	3.3 ± 1.7 ^{c}	38.0 ± 2.7 ^{bc}
High fertilizer					
None	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	29.4 ± 3.3 ^{ab}	29.4 ± 3.3 ^{c}
M. sativa	3.3 ± 2.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	22.2 ± 5.0 ^{bc}	25.5 ± 6.5 ^{c}
Grasses	0.0 ± 0.0 ^{b}	3.0 ± 1.7 ^{def}	12.0 ± 5.3 ^{bc}	10.9 ± 4.1 ^{bc}	25.9 ± 6.7 ^{c}
Grasses and M. sativa	0.0 ± 0.0 ^{b}	2.0 ± 0.9 ^{ef}	19.8 ± 5.8 ^{b}	13.3 ± 4.0 ^{bc}	35.1 ± 7.1 ^{bc}
Wood chips, low fertilizer					
None	9.7 ± 3.9 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	9.8 ± 3.4 ^{bc}	19.5 ± 3.7 ^{c}
M. sativa	0.0 ± 0.0 ^{b}	10.5 ± 2.9 ^{cd}	24.8 ± 5.2 ^{b}	2.9 ± 1.3 ^{c}	38.2 ± 6.6 ^{bc}
Grasses	11.0 ± 4.3 ^{b}	12.3 ± 0.6 ^{c}	22.3 ± 16.3 ^{b}	2.0 ± 1.1 ^{c}	47.6 ± 3.0 ^{bc}
Grasses and M. sativa	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	50.1 ± 10.9 ^{a}	50.1 ± 10.9 ^{b}
Papermill sludge, low fertilizer					
None	0.0 ± 0.0 ^{b}	47.0 ± 5.0 ^{a}	62.8 ± 9.7 ^{a}	7.8 ± 2.6 ^{bc}	117.6 ± 12.3 ^{a}
M. sativa	36.5 ± 10.7 ^{a}	23.8 ± 3.0 ^{b}	49.3 ± 3.7 ^{a}	4.7 ± 3.9 ^{c}	114.3 ± 12.0 ^{a}
Grasses	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	29.4 ± 3.3 ^{ab}	29.4 ± 3.3 ^{c}
Grasses and M. sativa	3.3 ± 2.0 ^{b}	0.0 ± 0.0 ^{f}	0.0 ± 0.0 ^{c}	22.2 ± 5.0 ^{bc}	25.5 ± 6.5 ^{c}

considerably across all amendment treatments during the 2010 field season, while seeded plant species remained largely contained within sub-plots where they were directly seeded (Table 4.11). The greatest volunteer establishment was observed in papermill sludge amended sub-plots where nothing was seeded (Figure 4.10). In these particular sub-plots, there was an increase from a maximum of ~ 5% in 2009, up to a maximum of ~ 50% in 2010 (Tables 4.9 & 4.11). All other unseeded sub-plots had significantly less volunteer establishment, with levels of ground cover being equal between amendments (Figure 4.11). As we compare total ground cover between amendments and seedings, it is clear that papermill sludge amended sub-plots, whose seeding treatment included the grass mix, promoted the greatest plant cover. Furthermore, for all amendment treatments other than the sludge, the sub-plot seeding treatment did not influence total ground cover values.

When cover estimates were made in July 2011, seeded species showed significant establishment only when they were seeded in tailings amended with papermill sludge (Table 4.12). As for the volunteer plant cover, un-seeded sub-plots amended with papermill sludge allowed the greatest cover establishment. (~ 55%). The next highest volunteer plant cover was seen on un-seeded sub-plots amended with the high fertilizer treatment; these sub-plots allowed statistically equal ground cover as all other high fertilizer amended sub-plots.

Figure 4.12 presents the average total cover found in each amendment treatment over the three sample years by seeding treatment. Unseeded sub-plots remained largely unvegetated over the first field season in all amendments after which volunteer invasion was witnessed, to a varying extent, in all treatments.

Figure 4.10. Unseeded subplot amended with papermill sludge encouraged high levels of weedy establishment.

Figure 4.11. Unseeded subplots that were otherwise amended did not encourage similarly high volunteer establishment (plot shown here amended with wood chips treatment).

Table 4.12. Ground cover (%) of seeded (*M. sativa, F. rubra* and *A. trachycaulum*) and unseeded (other) plants present on amended Gunnar mine tailings, 2 years after planting; significant interaction effect of amendment and seeding treatments found by Tukeys HSD test ($\alpha = 0.05$) is represented by levels not connected by the same letter; analyses performed on each plant species individually (Mean ± SE; n = 5).

Amendment *Seeded plants*	Ground Cover (%) *Medicago sativa*	 *Festuca rubra*	 *Agropyron trachycaulum*	 Other	 Total
Low fertilizer					
None	0.0 ± 0.0 [c]	0.0 ± 0.0 [b]	1.6 ± 1.0 [c]	11.3 ± 3.3 [bcde]	13.8 ± 3.1 [d]
M. sativa	0.5 ± 0.5 [c]	0.0 ± 0.0 [b]	1.5 ± 1.0 [c]	11.6 ± 2.5 [bcde]	14.9 ± 2.5 [d]
Grasses	0.0 ± 0.0 [c]	8.1 ± 3.3 [b]	9.1 ± 2.6 [bc]	5.2 ± 2.8 [de]	28.3 ± 4.9 [d]
Grasses and M. sativa	3.4 ± 1.4 [c]	4.3 ± 0.8 [b]	14.3 ± 2.9 [bc]	3.0 ± 2.4 [e]	29.9 ± 2.7 [d]
High fertilizer					
None	0.3 ± 0.3 [c]	0.0 ± 0.0 [b]	0.5 ± 0.3 [c]	32.2 ± 5.4 [b]	33.5 ± 5.7 [cd]
M. sativa	3.6 ± 2.3 [c]	0.0 ± 0.0 [b]	0.6 ± 0.5 [c]	28.1 ± 3.4 [bc]	35.0 ± 4.2 [cd]
Grasses	5.3 ± 5.3 [c]	0.0 ± 0.0 [b]	3.5 ± 2.4 [bc]	22.6 ± 8.2 [bcde]	39.0 ± 6.1 [cd]
Grasses and M. sativa	1.8 ± 1.1 [c]	2.8 ± 2.8 [b]	15.2 ± 6.5 [bc]	16.6 ± 3.6 [bcde]	46.7 ± 8.2 [cd]
Wood chips, low fertilizer					
None	0.0 ± 0.0 [c]	0.0 ± 0.0 [b]	2.0 ± 1.2 [bc]	8.2 ± 2.3 [cde]	2.0 ± 2.4 [d]
M. sativa	11.0 ± 4.4 [bc]	0.0 ± 0.0 [b]	1.3 ± 1.0 [c]	6.7 ± 3.3 [cde]	16.7 ± 3.4 [d]
Grasses	1.8 ± 1.8 [c]	5.8 ± 2.4 [b]	17.0 ± 3.9 [b]	1.5 ± 0.7 [e]	28.6 ± 3.0 [d]
Grasses and M. sativa	9.5 ± 4.7 [bc]	8.5 ± 3.1 [b]	7.3 ± 2.6 [bc]	2.8 ± 2.2 [e]	33.1 ± 4.7 [cd]
Papermill sludge, low fertilizer					
None	1.0 ± 0.6 [c]	0.0 ± 0.0 [b]	1.0 ± 0.5 [c]	55.0 ± 9.7 [a]	58.0 ± 9.6 [bc]
M. sativa	39.1 ± 11.2 [a]	0.0 ± 0.0 [b]	2.5 ± 1.0 [bc]	27.4 ± 3.9 [bcd]	81.2 ± 12.2 [b]
Grasses	0.5 ± 0.3 [c]	38.0 ± 2.4 [a]	53.0 ± 4.8 [a]	11.9 ± 3.9 [bcde]	110.9 ± 4.8 [a]
Grasses and M. sativa	30.4 ± 11.0 [ab]	31.3 ± 2.6 [a]	46.3 ± 5.4 [a]	12.4 ± 4.1 [bcde]	139.2 ± 5.9 [a]

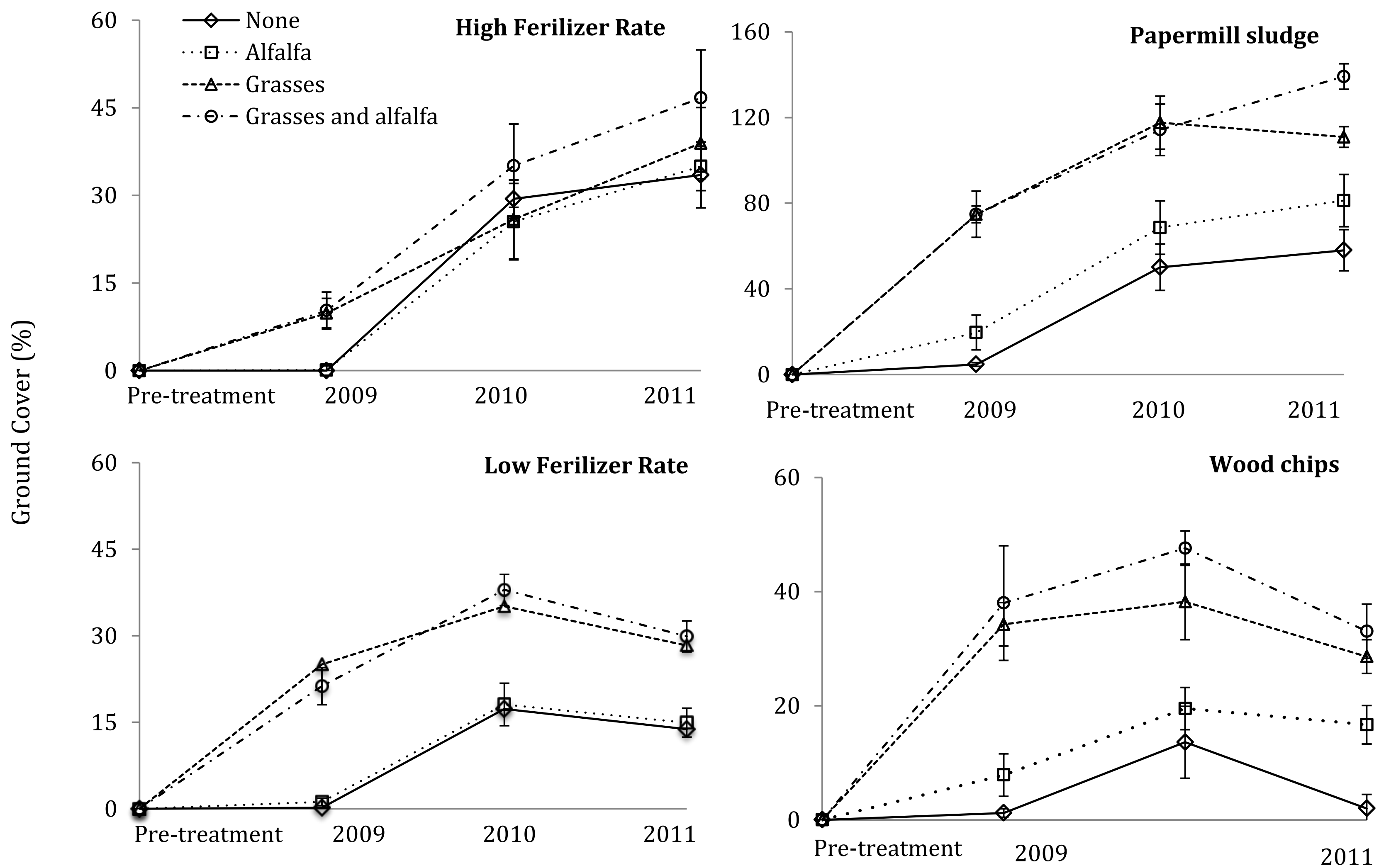

Figure 4.12. Ground cover (%) for differently amended and seeded Gunnar mine tailings (Mean ± SE; n = 5).

By 2010, the high fertilizer rate amended plots exhibited equivalent cover establishment regardless of seeding technique (Figure 4.12). After the 2010 field season, a decrease in cover was witnessed in certain sub-plots of the low fertilizer and the wood chips amendment treatments, while the high fertilizer rate and the papermill sludge treatments were able to maintain the established cover (Figure 4.12).

4.2.2 Shoot biomass

Shoot biomass production was significantly affected by both amendment and seeding treatment and an interaction was observed between these model effects. As such, the individual effects are not presented. The shoot biomass yields follow the trends witnessed in plant cover establishment over the 2010-11 sample dates (Tables 4.9, 4.11-13). For all three seeded plant species, only papermill sludge amended sub-plots in which they were seeded allowed increased shoot production (Table 4.13). Furthermore, sub-plots amended with papermill sludge and left un-seeded yielded by far the greatest shoot growth as compared to other un-seeded subplots. Overall, for total shoot biomass production, sub-plots amended with papermill sludge and seeded with the grass mix exceeded all other sub-plots (Table 4.13).

Table 4.13. Shoot biomass yield (g m^{-2}) of amended Gunnar mine tailings during 2010 field season; interaction of amendment and seeding treatments was found by Tukeys HSD test ($\alpha = 0.05$) and is represented by levels not connected by the same letter, analyses performed on each plant species individually (Mean ± SE, n = 5).

Amendment	Shoot Biomass Production (g m^{-2})				
Seeded plants	*Medicago sativa*	*Festuca rubra*	*Agropyron trachycaulum*	*Others*	*Total*
Low Fertilizer					
No seeding	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	5.7 ± 1.0 ^{bc}	5.7 ± 1.0 ^{c}
M. sativa	0.7 ± 0.7 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	3.4 ± 1.3 ^{bc}	4.1 ± 2.0 ^{c}
Grasses	0.0 ± 0.0 ^{b}	3.1 ± 1.3 ^{bc}	11.8 ± 2.8 ^{b}	0.7 ± 0.5 ^{c}	15.6 ± 4.6 ^{bc}
Grasses and M. sativa	1.7 ± 0.7 ^{b}	3.8 ± 1.3 ^{bc}	20.1 ± 3.5 ^{b}	0.3 ± 0.3 ^{c}	25.9 ± 5.8 ^{bc}
High Fertilizer					
No seeding	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	11.8 ± 4.4 ^{bc}	11.8 ± 4.4 ^{c}
M. sativa	1.6 ± 1.0 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	11.8 ± 4.0 ^{bc}	13.3 ± 5.0 ^{bc}
Grasses	0.0 ± 0.0 ^{b}	0.8 ± 0.6 ^{c}	17.8 ± 14.0 ^{b}	2.0 ± 1.1 ^{c}	20.6 ± 15.8 ^{bc}
Grasses and M. sativa	0.0 ± 0.0 ^{b}	2.2 ± 2.0 ^{bc}	9.6 ± 2.1 ^{b}	3.8 ± 2.2 ^{bc}	15.7 ± 6.2 ^{bc}
Wood Chips, low fertilizer					
No seeding	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	8.7 ± 2.0 ^{bc}	8.7 ± 2.0 ^{c}
M. sativa	3.4 ± 1.9 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	2.4 ± 1.3 ^{c}	5.8 ± 3.3 ^{c}
Grasses	0.0 ± 0.0 ^{b}	4.6 ± 1.6 ^{bc}	16.0 ± 3.3 ^{b}	3.5 ± 3.3 ^{bc}	24.0 ± 8.2 ^{bc}
Grasses and M. sativa	4.0 ± 2.5 ^{b}	6.0 ± 1.6 ^{bc}	13.0 ± 2.8 ^{b}	0.2 ± 0.2 ^{c}	23.2 ± 7.1 ^{bc}
Papermill Sludge, low fertilizer					
No seeding	0.0 ± 0.0 ^{b}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	28.9 ± 7.8 ^{a}	28.9 ± 7.8 ^{bc}
M. sativa	24.9 ± 6.6 ^{a}	0.0 ± 0.0 ^{c}	0.0 ± 0.0 ^{b}	19.0 ± 5.6 ^{ab}	43.9 ± 12.2 ^{b}
Grasses	0.0 ± 0.0 ^{b}	19.7 ± 6.2 ^{a}	61.1 ± 1.3 ^{a}	1.1 ± 1.0 ^{c}	81.9 ± 8.4 ^{a}
Grasses and M. sativa	35.6 ± 15.9 ^{a}	12.1 ± 5.4 ^{ab}	44.8 ± 20.0 ^{a}	1.3 ± 0.6 ^{d}	93.9 ± 42.0 ^{a}

4.2.3 Root biomass

Both amendment and seeding model effects affected the root biomass yields from the differently amended and seeded Gunnar mine tailings, but no interaction was found between effects (Table 4.14). The amendment treatments that did not include an organic addition (the low and high fertilizer amendments) resulted in significantly lower root growth than the sludge amended plots, and the wood chips amendment treatment was intermediate between these extremes (Table 4.14). Furthermore, sub-plots seeded with either the grass treatment or the grass and *M. sativa* treatment yielded significantly more root biomass as compared to sub-plots left un-seeded or seeded with *M. sativa* alone (Table 4.14).

4.2.4 *Picea mariana* seedling survival

The amendment and seeding treatments influenced *Picea mariana* survival and there was a significant interaction between these model effects and the sample date. After the first field season (2009), similar *P. mariana* seedling survival rates (~50%) were observed for the low fertilizer, papermill sludge and wood chips amended plots, regardless of the seeding treatment (Table 4.15). The high fertilizer amended plots, alternatively, showed a significantly reduced survival rate in the fall of 2009 compared to the other amendment treatments (~10%). After the first field season no seedlings planted in the high fertilizer plots had survived; from this point on the high fertilizer amendment treatment was removed from data analysis to maintain the assumptions of the ANOVA analysis. The other three amendment treatments saw considerably reduced survival following their first winter in the

Table 4.14. Root biomass yield (g m^{-3}) of amended Gunnar mine tailings, summer of 2010; no statistically significant interaction was found between amendment and seeding treatments, however both individual treatments produced significant effects by Tukeys HSD test ($\alpha = 0.05$) (Mean ± SE, n = 5).

Treatment: *Seeded plants*	Root Biomass (g m^3)		
Low Fertilizer			
None	4.1±2.3	b	b
M. sativa	2.6±1.2		
Grasses	24.3±6.9		a
Grasses and M. sativa	28.6±8.1		
High Fertilizer			
None	5.5±2.0	b	b
M. sativa	5.6±3.4		
Grasses	10.5±4.4		a
Grasses and M. sativa	18.9±11.4		
Wood Chips, low fertilizer			
None	5.3±2.3	ab	b
M. sativa	8.0±3.4		
Grasses	38.4±5.9		a
Grasses and M. sativa	38.7±6.4		
Papermill Sludge, low fertilizer			
None	14.3±5.1	a	b
M. sativa	35.7±10.6		
Grasses	70.0±19.9		a
Grasses and M. sativa	78.7±17.9		

Table 4.15. Survival rates (%) of *Picea mariana* seedlings on amended Gunnar mine tailings throughout the study; levels not connected by the same letter indicate a significant effect of either amendment (Fall 2009) or seeding (all other dates), high fertilizer treatment was omitted from analysis after 2009; (Mean ± SE; n = 5).

Amendment / *Seeding*	Survival Rates (%) Fall 2009		Spring 2010		Fall 2010		Spring 2011	
Low Fertilizer								
None	49.2 ± 4.6	a	4.2 ± 3.2	b	2.5 ± 2.5	b	6.7 ± 3.9	b
M. sativa	50.8 ± 6.6	a	1.7 ± 0.7	b	1.7 ± 1.7	b	1.7 ± 1.7	b
Grasses	49.2 ± 6.6	a	22.5 ± 6.8	a	21.7 ± 6.2	a	19.2 ± 6.3	a
Grasses and M. sativa	64.2 ± 4.3	a	22.2 ± 4.9	a	21.7 ± 4.8	a	20.0 ± 4.4	a
High Fertilizer								
None	17.5 ± 6.8	b	0.0 ± 0.0		0.0 ± 0.0		0.0 ± 0.0	
M. sativa	11.7 ± 5.2	b	0.0 ± 0.0		0.0 ± 0.0		0.0 ± 0.0	
Grasses	0.0 ± 0.0	b	0.0 ± 0.0		0.0 ± 0.0		0.0 ± 0.0	
Grasses and M. sativa	9.2 ± 5.5	b	0.0 ± 0.0		0.0 ± 0.0		0.0 ± 0.0	
Papermill Sludge, low fertilizer								
None	51.7 ± 8.8	a	15.0 ± 3.3	b	15.0 ± 5.7	b	14.2 ± 8.5	b
M. sativa	44.2 ± 9.1	a	5.0 ± 3.1	b	1.7 ± 1.0	b	7.5 ± 5.0	b
Grasses	52.5 ± 6.3	a	32.5 ± 6.6	a	30.8 ± 8.2	a	26.7 ± 5.4	a
Grasses and M. sativa	43.3 ± 8.8	a	20.8 ± 5.7	a	25.0 ± 9.7	a	19.2 ± 8.5	a
Wood Chips, low fertilizer								
None	59.2 ± 8.7	a	3.3 ± 2.0	b	2.5 ± 1.7	b	0.8 ± 0.8	b
M. sativa	62.5 ± 9.2	a	8.0 ± 3.3	b	7.5 ± 4.6	b	4.2 ± 1.9	b
Grasses	55.0 ± 8.1	a	25.2 ± 8.7	a	23.3 ± 10.7	a	20.8 ± 10.3	a
Grasses and M. sativa	72.9 ± 7.7	a	38.2 ± 9.3	a	35.8 ± 10.4	a	34.7 ± 14.1	a

field (average loss of 40.5 ± 3.2%) (Table 4.15). Seedling survival remained similar between amendment treatments for the remainder of the study during which time the seeding treatment influenced *P. mariana* survival rate. Of the three amendment treatments that had seedlings remaining alive, a greater proportion of seedlings survived past the first winter when planted where grasses had been seeded (Table 4.15). In these amendments, survival rates remained relatively constant over 2010-2011 sample dates and by the end of the study period seedlings planted with a grass cover averaged 23.5 ± 8.2% survival while those planted without a grass cover averaged 5.9 ± 3.6% survival.

4.3 Plant Stress

4.3.1 Pigment content

When pigment extraction was performed on fresh *M. sativa* tissues, during the 2010 field season, amendment treatments did not influence the levels of any pigments (Table 4.16). The seeding treatment did not influence chlorophyll contents in *M. sativa*, while the carotenoid content was significantly lower when *M. sativa* was seeded with the grass mix as compared to when seeded alone (Figure 4.13). When pigment extraction was performed in the 2011 field season on dry tissues, the amendment treatment did affect the amount of total chlorophyll present (Table 4.17). The total chlorophyll content of *M. sativa* seeded in papermill sludge amended or high fertilizer amended plots was significantly greater than that of the low fertilizer amended plots.

Pigment contents in *Agropyron trachycaulum* established on the Gunnar mine tailings for almost two full growing seasons were affected both by the amendment and the seeding treatments and no interaction was observed. Chlorophyll (a, b and total) as well as carotenoid content of *A. trachycaulum* was higher when seeded in high fertilizer amended tailings, as compared to papermill sludge or wood chips amended tailings (Table 4.18). Additionally, the carotenoid content of *A. trachycaulum* established on differently amended Gunnar mine tailings was significantly higher when seeded without *M. sativa* (Figure 4.14).

Table 4.16. Chlorophyll a, b and carotenoid content (mg g^{-1} FW) of *Medicago sativa* on amended tailings, summer 2010; no interactive effect of amendment and seeding treatment was found (α = 0.05), however carotenoid content was significantly affected by seeding (see Figure 4.13) (Mean ± SE; n = 4-5).

	Pigment Content (mg g^{-1} FW)				
Amendment: *Seeding*	Chl a	Chl b	Total Chl	Chl a / Chl b	Carotenoids
Low Fertilizer					
M. sativa alone	1.31 ±0.03	0.35±0.03	1.66 ±0.05	3.71±0.39	0.07 ±0.001
With grasses	1.40 ±0.07	0.39±0.03	1.79±0.09	3.61±0.17	0.06±0.006
High Fertilizer					
M. sativa alone	1.29 ±0.04	0.39±0.04	1.69 ±0.01	3.28±0.45	0.07 ±0.002
With grasses	1.12 ± 0.16	0.26±0.05	1.38± 0.21	4.29±0.18	0.05 ±0.005
Wood Chips, low fertilizer					
M. sativa alone	1.43 ± 0.12	0.46± 0.11	1.89± 0.22	3.10±0.38	0.07 ±0.008
With grasses	1.31 ±0.04	0.35±0.03	1.66±0.06	3.70±0.21	0.06±0.003
Papermill Sludge, low fertilizer					
M. sativa alone	1.50 ±0.08	0.44 ±0.03	1.93±0.08	3.42±0.29	0.08 ±0.004
With grasses	1.29 ±0.06	0.33 ±0.02	1.62 ±0.07	3.93±0.20	0.06±0.002

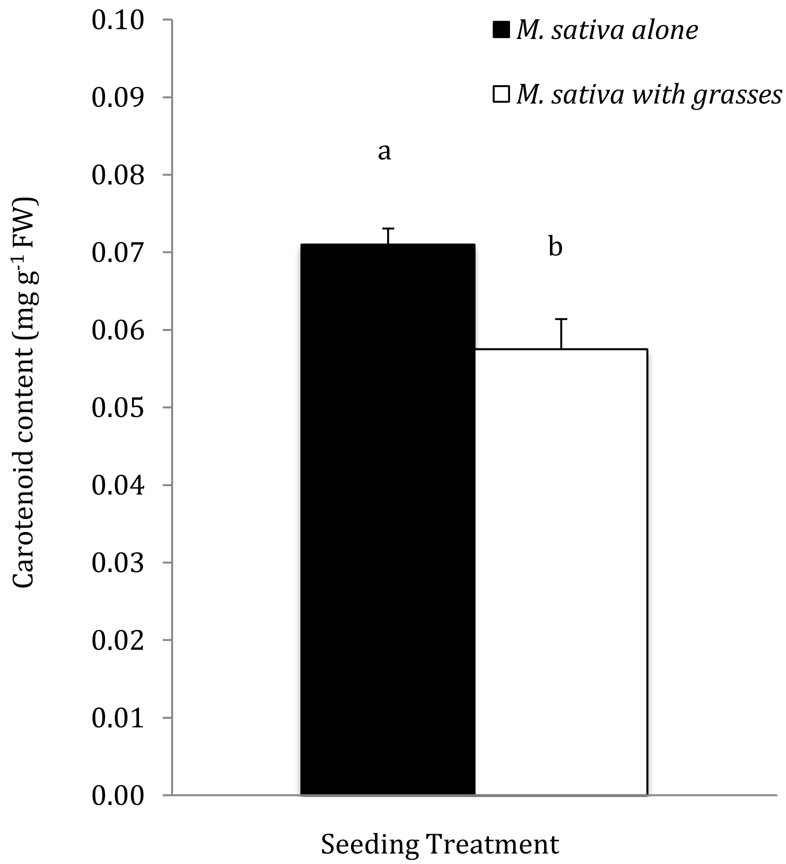

Figure 4.13. Carotenoid content (mg g^{-1} FW) of *Medicago sativa* seeded alone or with a grass mix on tailings; values are grouped across amendment treatments, summer 2010; levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 5).

Table 4.17. Pigment content (mg g^{-1} DW) of *Medicago sativa* growing on amended Gunnar mine tailings, July 2011; tissue was collected from across experimental plots; levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 3-5).

Treatment	Chl a	Chl b	Total Chl	Chl a / Chl b
Low Fertilizer	0.35±0.04 ^{b}	0.10 ±0.01^{b}	0.45 ±0.05 ^{b}	3.74±0.26
High Fertilizer	0.55 ±0.05^{a}	0.15±0.01^{a}	0.70±0.06^{a}	3.62±0.34
Wood Chips, low fertilizer	0.45 ±0.04^{ab}	0.12 ±0.01^{ab}	0.57±0.05^{ab}	3.69±0.29
Papermill Sludge, low fertilizer	0.53 ±0.04 ^{a}	0.14 ±0.01^{ab}	0.66±0.05 ^{a}	3.87±0.26

Table 4.18. Chlorophyll a,b and carotenoid content (mg g^{-1} FW) of *Agropyron trachycaulum* on differently amended Gunnar mine tailings, summer 2010; no interactive effect of amendment and seeding treatment was found by Tukeys HSD test (α = 0.05), however amendment treatment affected all response levels; levels not connected by the same letter are significantly different (α = 0.05)(Mean ± SE; n = 5).

Amendment:	Pigment Content (mg g^{-1} FW)									
Seeding	Chl a		Chl b		Total Chl		Chl a / Chl b		Carotenoids	
Low Fertilizer										
Grasses alone	1.12 ± 0.10	ab	0.59 ± 0.05	ab	1.71 ± 0.15	ab	1.89 ± 0.02	ab	0.123 ± 0.009	ab
Grasses and M. sativa	1.40 ± 0.07		0.43 ± 0.20		1.83 ± 0.09		3.28 ± 0.07		0.085 ± 0.004	
High Fertilizer										
Grasses alone	1.30 ± 0.10	a	0.68 ± 0.05	a	1.97 ± 0.15	a	1.91 ± 0.03	a	0.141 ± 0.010	a
Grasses and M. sativa	1.62 ± 0.09		0.49 ± 0.03		2.12 ± 0.12		3.30 ± 0.05		0.099 ± 0.005	
Wood Chips, low fertilizer										
Grasses alone	1.10 ± 0.06	b	0.55 ± 0.03	b	1.62 ± 0.09	b	1.93 ± 0.04	b	0.118 ± 0.006	b
Grasses and M. sativa	1.32 ± 0.07		0.40 ± 0.02		1.72 ± 0.09		3.27 ± 0.09		0.080 ± 0.004	
Papermill Sludge, low fertilizer										
Grasses alone	0.92 ± 0.09	b	0.46 ± 0.05	b	1.38 ± 0.14	b	1.97 ± 0.05	b	0.102 ± 0.010	b
Grasses and M. sativa	1.12 ± 0.06		0.31 ± 0.02		1.43 ± 0.08		3.66 ± 0.06		0.079 ± 0.004	

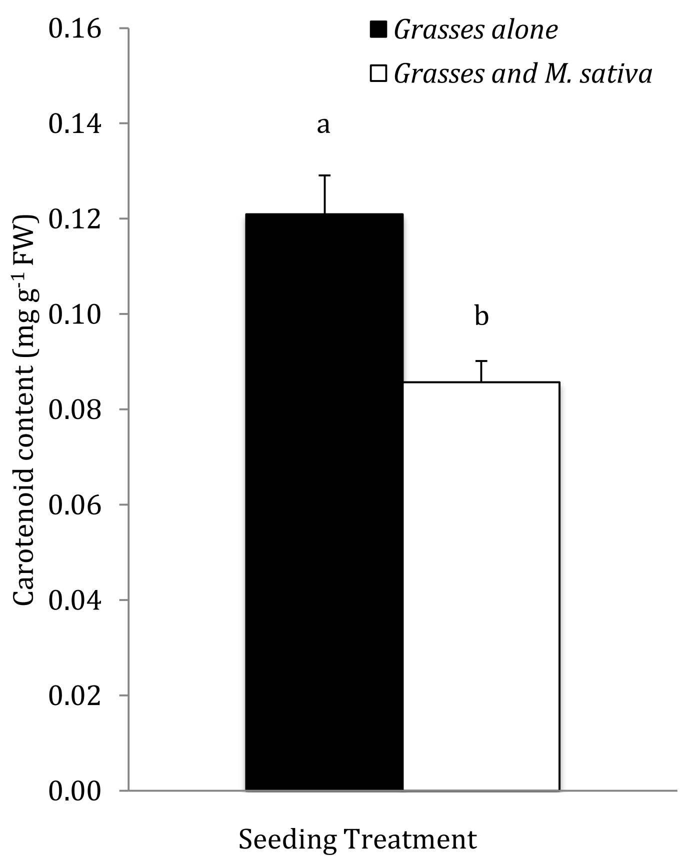

Figure 4.14. Carotenoid content (g mg^{-1} FW) of *Agropyron trachycaulum* seeded as a grass mix or as a grass mix supplemented with *M. sativa* in tailings, summer 2010; values grouped across amendment treatments; levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 5).

Pigment content (both total chlorophyll and carotenoid) of *Festuca rubra* sampled near the end of the 2010 field season was affected by the seeding treatment it was exposed to, but not affected by the amendment treatment (Figures 4.15 and 4.16). *Festuca rubra* that was seeded without *M sativa* yielded higher levels of both total chlorophyll and carotenoids as compared to *F. rubra* that was seeded along with *M. sativa* (Figures 4.15 and 4.16). *Festuca rubra* seeded alone yielded total chlorophyll and carotenoid contents approximatley 44% and 36% higher, respectively, than when seeded with *M. sativa*, (Table 4.19).

4.3.2 Proline content

The proline levels of the different plants species showed considerable variation on the differently amended and seeded tailings (with values ranging from 0.78 ± 0.04 to 7.29 ± 2.52 μmol g^{-1}FW)(Table 4.20). *Medicago sativa* contained the highest proline levels ranging from 2.58 ± 0.17 to 7.29 ± 2.52 μmol g^{-1} FW, while *A. trachycaulum* and *F. rubra* showed lower levels ranging from 0.78 ± 0.06– 1.45 ± 0.18 μmol g^{-1} FW (Table 4.20). No statistically significant differences were detected based on the effect of amendment treatments.

4.3.3 Symbiotic nitrogen fixation

Symbiotic nitrogen fixation per mass of nodules in *M. sativa* (as measured by acetylene reduction) was significantly affected by the mine tailings amendment treatment (Table 4.21). Plants present in tailings treated with the high fertilizer rate

Table 4.19. Chlorophyll a, b and carotenoid content (mg g^{-1} FW) of *Festuca rubra* planted on amended Gunnar mine tailings, summer 2010; no interactive effect of amendment and seeding treatment was found by Tukeys HSD test (α = 0.05), however seeding treatment affected all response levels (α = 0.05) (see Figures 4.15 and 4.16) (Mean ± SE; n = 4 - 5).

Amendment: *Seeding*	Pigment Content (mg g^{-1} FW) Chl a	Chl b	Total Chl	Chl a/ Chl b	Carotenoids
Low Fertilizer					
Grasses alone	0.32 ±0.06	0.11±0.02	0.43 ±0.08	2.89±0.10	0.022±0.004
Grasses and M. sativa	0.23 ±0.02	0.04±0.01	0.27±0.03	5.23±1.75	0.012±0.001
High Fertilizer					
Grasses alone	0.44 ±0.01	0.14±0.03	0.57± 0.12	3.07±0.13	0.024±0.005
Grasses and M. sativa	0.31±0.09	0.08±0.03	0.39± 0.12	3.93±0.45	0.018±0.005
Wood Chips, low fertilizer					
Grasses alone	0.26 ±0.04	0.09 ±0.01	0.34±0.06	2.97±0.12	0.018±0.002
Grasses and M. sativa	0.15±0.02	0.03 ±0.01	0.18±0.02	5.54±1.59	0.009±0.001
Papermill Sludge, low fertilizer					
Grasses alone	0.29±0.05	0.10 ±0.02	0.39 ±0.07	3.01±0.24	0.021±0.003
Grasses and M. sativa	0.25 ±0.04	0.06 ±0.01	0.31±0.05	4.32±1.10	0.016±0.003

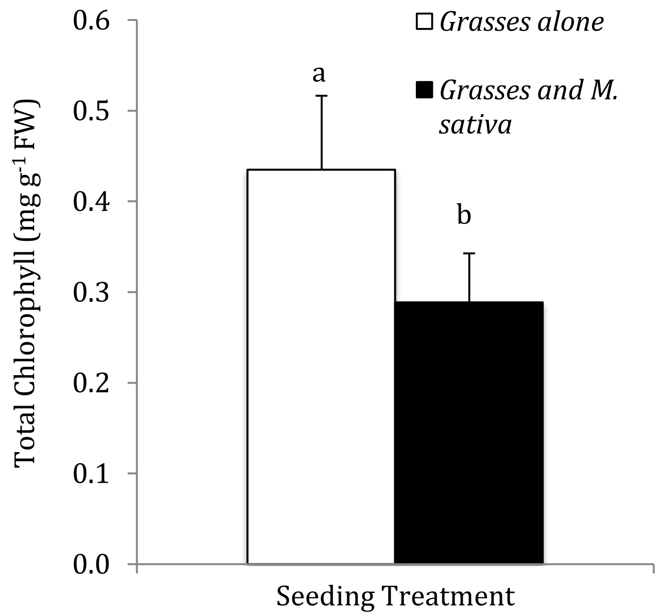

Figure 4.15. Total chlorophyll content (mg g^{-1} FW) of *Festuca rubra* seeded either with or without alfalfa in tailings, summer 2010; values grouped across amendment treatments; levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 5).

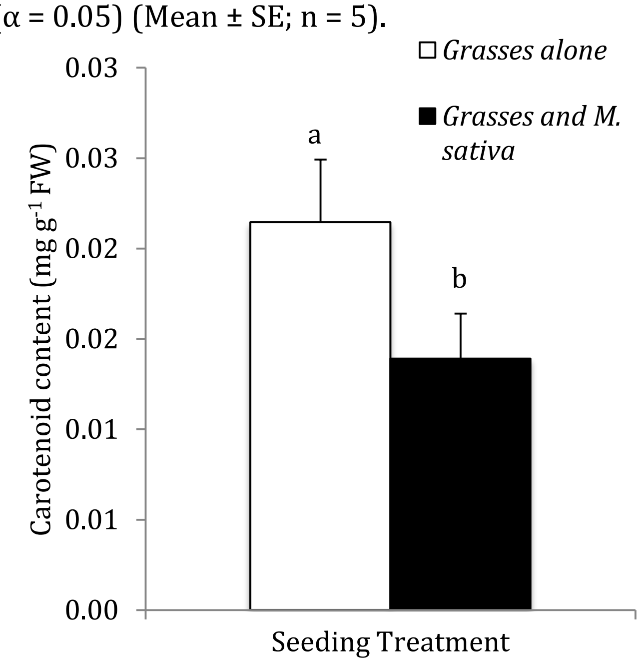

Figure 4.16. Carotenoid content (mg g^{-1} FW) of *Festuca rubra* seeded either with or without alfalfa in tailings, summer 2010; values grouped across amendment treatments; levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 5).

Table 4.20. Proline content (μmol g^{-1} FW) of three seeded species grown on differently amended Gunnar mine tailings, summer 2010; no significant differences were found by Tukeys HSD test (α = 0.05) (Mean ± SE; n = 4-5).

Treatment	Proline Content (μmol g^{-1} FW)		
Seeding strategy	*Medicago sativa*	*Festuca rubra*	*Agropyron trachycaulum*
Low Fertilizer			
M. sativa or grasses alone	5.87±0.90	1.45±0.18	1.41±0.35
M. sativa with grasses	3.41±0.45	1.19±0.12	1.18±0.18
High Fertilizer			
M. sativa or grasses alone	2.79±0.16	0.93±0.23	1.22±0.25
M. sativa with grasses	2.58±0.17	1.32±0.25	1.24±0.21
Wood Chips, low fertilizer			
M. sativa or grasses alone	7.29±2.52	1.11±0.17	0.78±0.04
M. sativa with grasses	2.77±0.37	0.92±0.09	1.25±0.43
Papermill Sludge, low fertilizer			
M. sativa or grasses alone	5.02±0.92	0.97±0.17	0.78±0.06
M. sativa with grasses	3.31±0.23	0.84±0.14	1.11±0.12

Table 4.21. Nitrogen fixation (μmol C_2H_2 $mL^{-1}g^{-1}$ of nodule h^{-1}) of *Medicago sativa* nodules present on differently amended Gunnar mine tailings, July 2011; levels not connected by the same letter are significantly different by Tukeys HSD test ($\alpha = 0.05$); (Mean ± SE; n = 3-5).

Amendment Treatment	Nitrogen fixation (μmol C_2H_2 $mL^{-1}g^{-1}$ of nodule h^{-1})
Low Fertilizer	843 ± 516 ^{b}
High Fertilizer	5167 ± 595 ^{a}
Papermill Sludge, low fertilizer	597 ± 461 ^{b}
Wood Chips, low fertilizer	1125 ± 595 ^{b}

amendment were able to fix significantly more atmospheric nitrogen as compared to the other amendment treatments during the 2011 sampling period (Table 4.21). *Medicago sativa* established in tailings treated with either the low fertilizer rate, papermill sludge or wood chips amendments did not show any significant differences in nitrogen fixation. The nitrogen fixation rate was more than four times higher for *M. sativa* established in the high fertilizer rate amended tailings than that of *M. sativa* present in differently amended Gunnar mine tailings (Table 4.21).

4.3.3 Gas exchange

There was no significant effect of amendment treatment on the measured gas exchange parameters (photosynthetic rate, stomatal conductance and transpiration)(Figures 4.17 – 4.19). All plants reached their light compensation point between ~ 20 – 50 μmol photons m^{-2} s^{-1}, where the photosyntheic rate exceeds their respiration rate. The light saturated phase appeared to begin around the 1000 μmol photons m^{-2} s^{-1} level, at which point the *M. sativa* plants were photosynthesizing at a rate ranging between ~ 10 – 15 μmol $C0_2$ $m^{-2}s^{-1}$. Maximal photosynthetic rates were approximated using the exponential function detailed by Chalker (1981). Based on this function, the maximal photosynthetic rates for the various amendments are 17.64, 11.33, 16.16, and 11.62 μmol $C0_2$ $m^{-2}s^{-1}$ for papermill sludge, high fertilizer, low fertilizer, and wood chips amendments, respectively. The transpiration and conductivity rates increased markedly in response to increasing light intensity and plateaued approximatley 350 and 150 μmol photons $m^{-2}s^{-1}$, repectively (Figures 4.18 and 4.19).

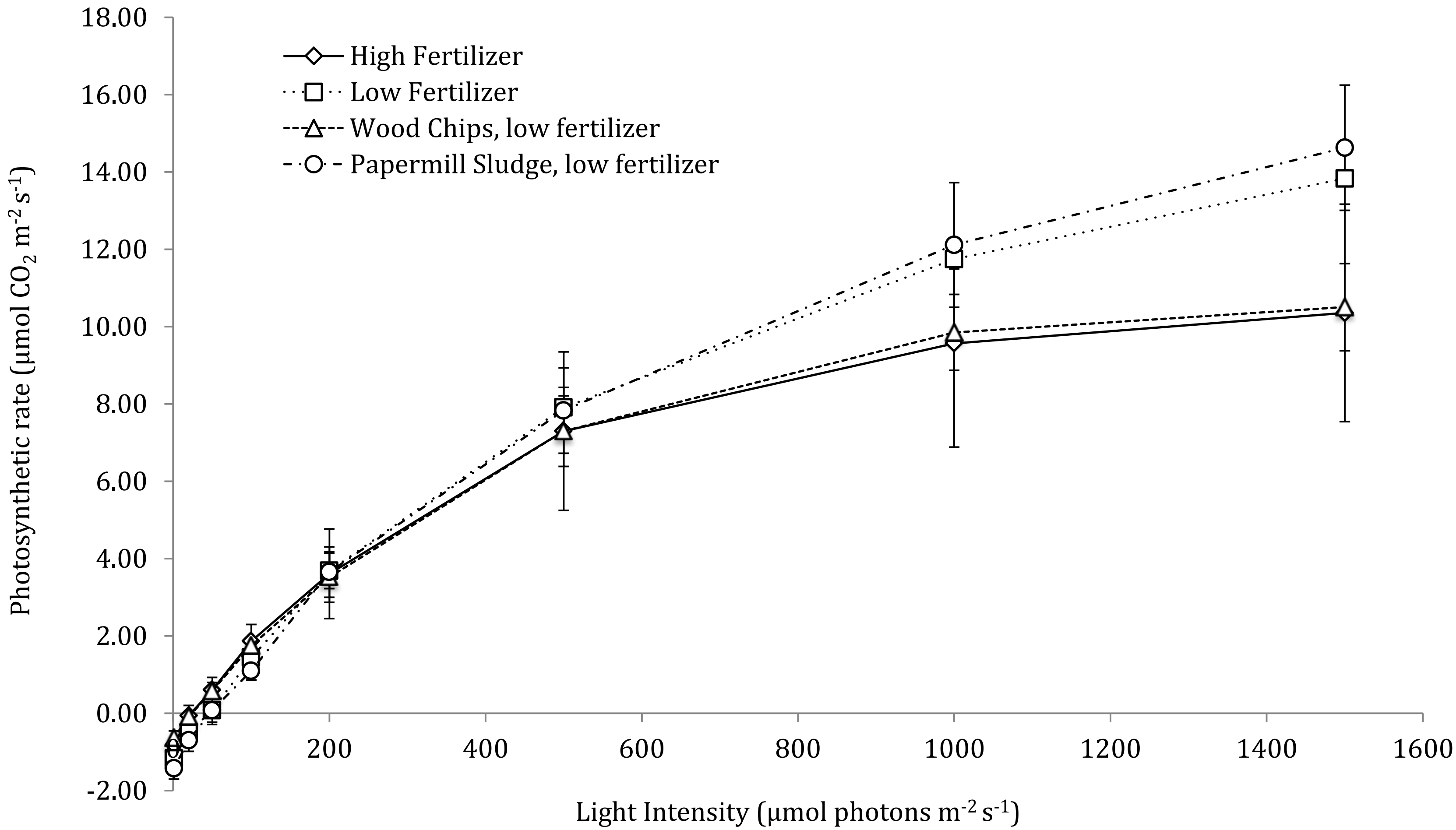

Figure 4.17. Photosynthetic rate (μmol CO_2 m^{-2} s^{-1}) of *Medicago sativa* established on differently amended Gunnar mine tailings under various light intensities, August 2011 (μmol photons m^{-2} s^{-1}), July 2011 (Mean ± SE, n = 2 - 3). No statistically significant differences were found by Tukeys HSD test (α = 0.05).

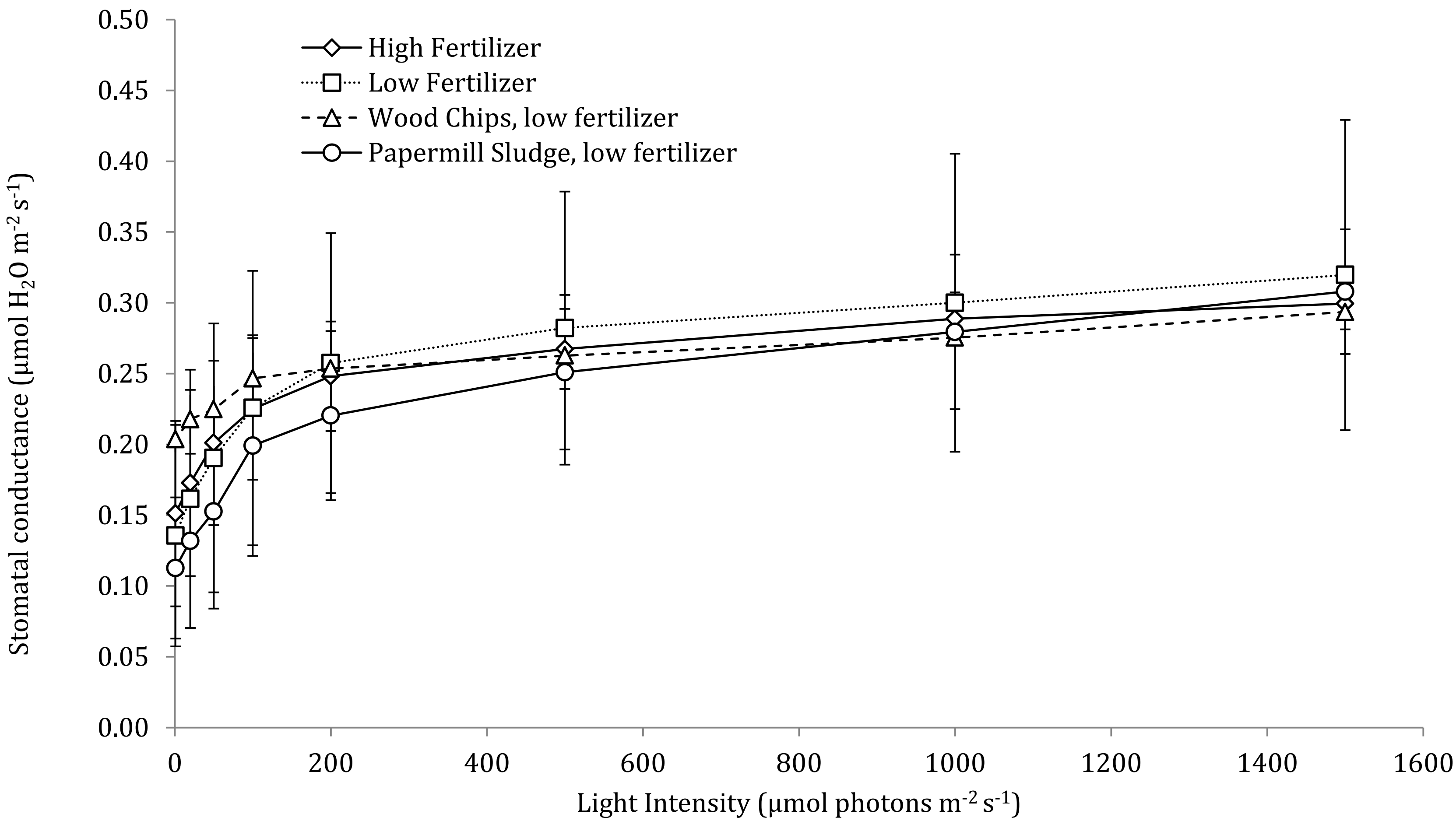

Figure 4.18. Stomatal conductivity (mmol H_2O m^{-2} s^{-1}) of *Medicago sativa* established on differently amended Gunnar mine tailings under various light intensities, August 2011 (µmol photons m^{-2} s^{-1}), July 2011 (Mean ± SE, n = 2 - 3). No statistically significant differences were found by Tukeys HSD test ($\alpha = 0.05$).

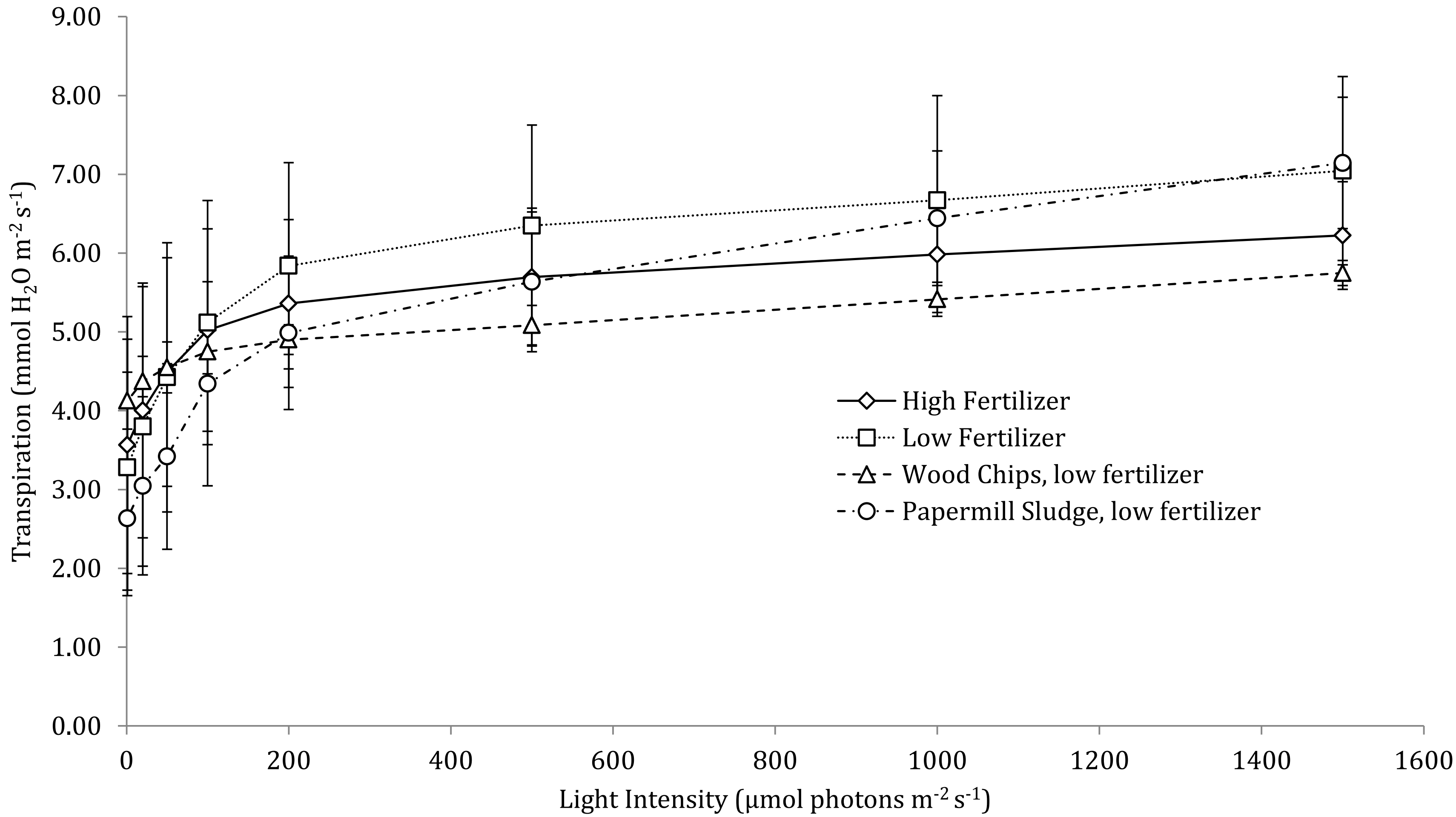

Figure 4.19. Transpiration rate (mmol H_2O m^{-2} s^{-1}) under various light intensities of *Medicago sativa* growing on differently amended Gunnar mine tailings (Mean ± SE, n = 2 - 3). No statistically significant differences were found by Tukeys HSD test (α = 0.05).

4.3.5 Elemental composition

The elemental composition of *M. sativa* and *A. trachycaulum* plants established on differently amended Gunnar mine tailings are presented in Table 4.22. While there were inter-specific differences between the contents of certain elements in these plants, relatively few differences were found based on the amendment treatment that they were subjected to. In fact, the only statistically significant difference was found for *A. trachycaulum* while considering manganese content (Table 4.22); plants grown in low fertilizer amended tailings exhibited a greater manganese content than those plants established on papermill sludge amended mine tailings (30.40 ± 0.13 vs. 20.18 ± 0.11 mg kg^{-1}, respectively). *M. sativa* shoot tissues had greater content of many elements relative to *A. trachycaulum*, including calcium, magnesium, molybdenum, copper, boron, arsenic, cadmium, cobalt, chromium and nickel (Table 4.22). In the case of *M. sativa*, although the level of many plant nutrients are sufficient for adequate plant growth, potassium, calcium and manganese may be deficient (Fageria et al., 2011). Conversely, in *A. trachycaulum* tissues, the levels of potassium, calcium, magnesium, manganese, copper, and boron are all low relative to average content in crop plant tissues (Fageria et al., 2011). Molybdenum levels in both species are high relative to the average range of this elelment in crop plants, however the levels within *M. sativa* shoot tissue are very high (Fageria et al., 2011).

Table 4.22. Elemental composition of *Medicago sativa* and *Agropyron trachycalum* shoot tissues from plants established on amended Gunnar mine tailings harvested in July, 2011 (Mean ± SE); levels not connected by the same letter are significantly different by Tukeys HSD test (α = 0.05).

Element (unit) *Amendment*	*Medicago sativa*	*Agropyron trachycaulum*
Potassium (mg kg^{-1})		
Low Fertilizer	87±18	40±2.0
High Fertilizer	77±7.6	41±2.3
Wood Chips, low fertilizer	75±6.8	35±2.9
Papermill sludge, low fertilizer	88±5.5	37±2.0
Calcium (mg kg^{-1})		
Low Fertilizer	11,670±228	1,710±76
High Fertilizer	8,330±150	1,990±114
Wood Chips, low fertilizer	7,670±146	1,460±165
Papermill sludge, low fertilizer	11,400±428	1,585±110
Magnesium (mg kg^{-1})		
Low Fertilizer	2,470±400	700±40
High Fertilizer	2,035±135	670±40
Wood Chips, low fertilizer	2,340±385	900±95
Papermill sludge, low fertilizer	2,500±260	745±45
Manganese (mg kg^{-1})		
Low Fertilizer	24.05±0.03	30.40±0.13^{a}
High Fertilizer	18.36 ±0.28	27.61±0.19^{ab}
Wood Chips, low fertilizer	21.91±0.16	26.83±0.16^{ab}
Papermill sludge, low fertilizer	16.45±0.21	20.18±0.11^{b}
Molybdenum (mg kg^{-1})		
Low Fertilizer	8.37±0.03	0.44±0.00
High Fertilizer	6.39±0.17	0.40±0.00
Wood Chips, low fertilizer	7.38±0.18	0.65±0.01
Papermill sludge, low fertilizer	8.34 ±0.05	0.49±0.00
Iron (mg kg^{-1})		
Low Fertilizer	355±12	265±30
High Fertilizer	290±10	295±55
Wood Chips, low fertilizer	250±10	290±50
Papermill sludge, low fertilizer	285±10	280±45

Table 22. Continued…

Element (unit) Amendment	*Medicago sativa*	*Agropyron trachycaulum*
Copper (mg kg^{-1})		
Low Fertilizer	5.67±0.01	2.40±0.01
High Fertilizer	5.69±0.07	2.91±0.04
Wood Chips, low fertilizer	4.49±0.06	2.86±0.03
Papermill sludge, low fertilizer	6.29±0.03	2.79±0.02
Zinc (mg kg^{-1})		
Low Fertilizer	31.0±0.0	28.0±0.1
High Fertilizer	42.9±1.5	30.6±0.3
Wood Chips, low fertilizer	29.6±0.1	34.4±0.2
Papermill sludge, low fertilizer	28.4 ±0.2	36.1±0.3
Boron (mg kg^{-1})		
Low Fertilizer	37.5±0.0	2.8±0.0
High Fertilizer	23.4±0.5	3.4±0.0
Wood Chips, low fertilizer	24.6±0.5	2.9±0.0
Papermill sludge, low fertilizer	38.3±0.2	2.8±0.0
Aluminum (mg kg^{-1})		
Low Fertilizer	130.6±0.5	128.1±1.2
High Fertilizer	105.3±4.2	128.5±2.4
Wood Chips, low fertilizer	97.1±3.3	130.4±2.2
Papermill sludge, low fertilizer	95.0±3.6	123.3±2.1
Silver (mg kg^{-1})		
Low Fertilizer	<0.02	<0.01
High Fertilizer	<0.02	<0.01
Wood Chips, low fertilizer	<0.02	<0.01
Papermill sludge, low fertilizer	<0.02	<0.01
Arsenic (mg kg^{-1})		
Low Fertilizer	0.9±0.0	<0.12
High Fertilizer	0.8±0.0	<0.12
Wood Chips, low fertilizer	0.8±0.0	<0.11
Papermill sludge, low fertilizer	1.2±0.0	<0.11
Cadmium (mg kg^{-1})		
Low Fertilizer	0.13±0.00	0.03±0.00
High Fertilizer	0.26±0.00	0.04±0.00
Wood Chips, low fertilizer	0.17±0.00	0.03±0.00
Papermill sludge, low fertilizer	0.13±0.00	0.04±0.00

Table 22. Continued…

Element (unit) Amendment	*Medicago sativa*	*Agropyron trachycaulum*
Cobalt (mg kg^{-1})		
Low Fertilizer	0.66±0.00	0.27±0.00
High Fertilizer	0.53±0.01	0.35±0.01
Wood Chips, low fertilizer	0.44±0.01	0.27±0.00
Papermill sludge, low fertilizer	0.49±0.01	0.26±0.00
Chromium (mg kg^{-1})		
Low Fertilizer	<0.88	<0.40
High Fertilizer	<0.77	<0.41
Wood Chips, low fertilizer	<0.75	<0.35
Papermill sludge, low fertilizer	<0.88	<0.37
Sodium (mg kg^{-1})		
Low Fertilizer	68±3.0	< 30
High Fertilizer	38±5.0	< 30
Wood Chips, low fertilizer	75±3.0	< 30
Papermill sludge, low fertilizer	65±1.5	< 30
Nickel (mg kg^{-1})		
Low Fertilizer	1.8±0.0	0.9±0.0
High Fertilizer	1.6±0.0	1.0±0.0
Wood Chips, low fertilizer	1.5±0.0	1.2±0.0
Papermill sludge, low fertilizer	1.4±0.0	0.9±0.0
Titanium (mg kg^{-1})		
Low Fertilizer	6.4±0.0	4.5±0.0
High Fertilizer	6.0±0.1	5.1±0.1
Wood Chips, low fertilizer	6.0±0.1	4.5±0.1
Papermill sludge, low fertilizer	5.8±0.1	4.6±0.0

Chapter 5. Discussion

5.1 Tailings fertility

Tailings conductivity and pH of the tailings were not considered to be problematic in terms of plant response, and they were not significantly affected by any amendment treatment. There was a pulse of available nutrients (N, P and K) following fertilization in all treatments, however this effect was short lived with all treatments returning to background levels within two years. The levels of certain elements in the tailings changed slightly over the first year, though not in a predictable manor. Physical properties of the Gunnar mine tailings were generally improved or unaffected by the amendment treatments, with by far the greatest improvements being found in the papermill sludge amended tailings. These improvements included an increase in fine organic carbon, cation exchange capacity, water stable aggregation, as well as a decrease in bulk density. The wood chips amended tailings, likewise, improved certain characteristics however the rate of improvement as well as the magnitude was generally significantly lower than for papermill sludge amended tailings.

5.1.1 Electrical conductivity and pH

While some mine wastes are characterized as having levels of soluble salts sufficient to negatively affect plant growth, those left by the Gunnar mine are below the salinity threshold of 4 dS m^{-1} (Belden et al., 1990; Borden & Black, 2005; Lottermoser, 2003). The primary sources of soluble salts in mine wastes are the weathering of carbonate or sulfide minerals as well as through mineral processing techniques that require the addition of salts (Lottermoser, 2003). Furthermore, the presence of salts in the active rooting zone is directly related to prevailing climatic factors that encourage either the leaching of salts away from, or the draw of salts up into the active rooting zone (Ripley et al., 1996). During the natural colonization of the Gunnar mine tailings, an overall decrease in conductivity was observed with increasing time since plant establishment (Young et al., 2012). The naturalized tailings were described by a rapid decrease in conductivity over the first five meters (most recently colonized), followed by a more gradual reduction over the remaining 40 meters (older vegetation). Additionally, the levels of salinity near the fringe of the encroaching vegetation (~1.2 dS m^{-1}) were lower than those observed across the exposed, unvegetated experimental area (~ 2 – 4 dS m^{-1}). These observations suggest that the establishment of a native vegetative community fosters conditions conducive to the leaching and removal of soluble salts from the active rooting zone (Young et al., 2012). The Gunnar mine tailings are described by a sandy texture with a very low cation exchange capacity as well as minimal organic matter content. In similarly composed substrate, Feagly et al (1994) attributed the low overall soluble salt content to a physical and chemical composition that did not promote salt accumulation, namely a well-drained structure with little capacity to bind available ions. During other field applications of wood chips and papermill biosolids, the substrate electrical conductivity remained relatively unchanged relative to unamended substrates regardless of application rate (Price & Voroney, 2007; Tahboub et al., 2008). In contrast, a study on amend saline bentonite mine spoil reported a beneficial effect (a decrease) of wood waste incorporation on conductivity (Belden et al., 1990). The authors suggested that the decrease in soluble salts in those wastes amended with wood residues was an indirect response to the improved water infiltration and holding capacity of the substrate leading to improved leaching and loss of soluble salts from the active rooting zone (Belden et al., 1990). In a similar fashion, when aiming to reclaim saline soils in arid to semi-arid climates, the use of continual irrigation (e.g. drip irrigation systems) has been successful in leaching salts from the upper, active rooting volume (Johnson et al., 1994). Given the non-saline nature of the Gunnar mine tailings, in addition to low sensitivity to salts exhibited by the seeded cover vegetation, it is unlikely that salinity negatively affected plant establishment during the experimental period. Unfortunately, due to a high degree of intrinsic variation in the tailings salt content, subtle changes in relation to amendment treatments may have been missed.

Several types of reactions generally govern the pH of mine tailings; these can be described as acid producing, acid consuming, and non-acid producing/consuming reactions (Lottermoser, 2003). The mineralogy of the tailings dump has been inferred to be primary in nature (remaining unaltered geochemically since its deposition), and composed largely of quartz, calcite, plagioclase, pyrite, Fe-oxides, galena, sphalerite, as well as micas and clays (Lambert, 2001). Given the neutral to slightly alkaline pH of the mine tailings, it must be assumed that the acid consuming reactions are sufficient to neutralize the acidity produced through the oxidation of iron containing minerals, most importantly pyrite (Lottermoser, 2003). While application of organic amendments can lead to changes in substrate pH (Walker et al., 2004; Fageria et al., 2011), at the low application rates (< 3.70 kg m^{-2}) used at the

Gunnar mine site it is not surprising that none affected substrate pH. Over time, as the tailings are colonized by the surrounding plant communities (largely *Picea mariana-Larix larcinia* forest stands) a gradual acidification of the substrate may be witnessed as a result of biological factors including the deposition of low pH plant resides (Crocker & Major, 1955; Raven et al., 2005).

5.1.2 Organic Carbon

Over the two-year experimental period, the organic carbon levels in the fine soil fraction (< 0.5 mm) of amended tailings where organic additions were made (wood chips and papermill sludge treatments) doubled. In fact, the levels of soil organic carbon witnessed in the organically amended tailings exceeded that of the adjacent naturally revegetated tailings (Young et al., 2012). Based on the rate of increase in the naturalized tailings during their gradual colonization (over a 70 year period), it would likely take in excess of a hundred years for native vegetation to achieve equivalent carbon levels to these tailings. The rate of organic materials applied to the Gunnar tailings was low relative to most similar studies and the corresponding increase in soil organic carbon was expectedly low (Tester, 1990; Bulmer, 2000; Price & Varoney, 2007). Following the application of our organic materials (3.70 kg m^{-2} for papermill sludge and 3.50 kg m^{-2} for wood chips, equivalent to 2.15 kg C m^{-2} and 2.03 kg C m^{-2}, respectively) and their incorporation into the upper 0.15-0.2 m (bulk density of 1.40 kg m^{-3}), we would have expected an increase in soil carbon between 0.76 – 1.02% for the papermill sludge amended tailings and 0.73 - 0.97% wood chips amended tailings (see Appendix C). At a rate of

5.6 kg m^{-3}, Green and Renault (2008) found in a greenhouse experiment that the incorporation of papermill sludge to tailings from Central Manitoba, a similar mine site in Nopiming Park, resulted in an increase in soil carbon of 0.8%. Similarly, the organic carbon of mine tailings has been increased in other studies through amendment with modified humic substances as well as other organic materials both in controlled and field settings (Ibrahim & Goh, 2004; Reid & Naeth, 2004a; Szczerski, et al., 2013. The discrepancy between the expected increase in carbon and the observed increase in carbon suggests that a considerable fraction of the organic matter addition is present in the greater than 0.5 mm diameter soil fraction.

While the papermill sludge amended plots exhibited an increase in soil organic carbon relatively rapidly (after the first growing season), the carbon enrichment from the wood chip incorporation was not detectable in the fine tailings fraction until two full years following amendment application. Since the wood chips applied to the tailings were, at least initially, greater than 0.5 mm in size, the carbon enrichment made by this amendment was not detected before some decay occurred, bringing it into the fine soil fraction. The wood chips may have resisted decomposition for the two-year period as a result of their physical structure, in addition to their high carbon and low nitrogen content. The wood chips amendment ranged from sawdust sized particles up to approximately 36 cm^3; this large difference in surface area contributes to the more rapid decay witnessed in the saw dust fraction of this amendment. In a similar fashion, following application of either wood chips or saw dust to degraded forest soils, Bulmer (2000) witnessed a greater incorporation of the organic matter into the fine soil fraction in sawdust amended soils as compared to the soils amended with wood chips. The extremely high C to N ratio of this material would suggest that it is rich in carbon-rich structural type molecules that will degrade much more slowly than other more nitrogen rich substances (Brady & Weil, 2008). The decay rate of wood will depend on environmental factors such as temperature and moisture, as well as the presence of important guilds of decomposer organisms (Gonzalez et al., 2008; Barton & Northup, 2011). In dry boreal environments, Gonzalez et al (2008) found little loss of mass from *Populus tremuloides* stakes occurring over the first four years (< 10%), while in moist boreal environments the loss was approximately 30% over the four year study. Similarly, when applied to bentonite mine spoils in a Northern Great Plains situated study, the average decomposition rate of *Pinus ponderosa* wood residues was 10.4% after two years and 25.3% after five years, regardless of application rates (Shuman & Belden, 1991). The lag time of two years before this amendment is incorporated into the fine fraction of the Gunnar tailings organic carbon has important implications. If this first noticeable increase represents a fine fraction of the wood chips amendment then it can be expected that as larger fractions decompose there will be further increases in soil organic carbon for this treatment over time. In fact, based on the expected increase in soil organic carbon as a direct result of the wood chips addition (0.97 – 0.73%) and the increase witnessed over the two year study (0.27%) we expect to see a further increase in soil organic carbon of between 0.46 – 0.70% following the decomposition of the remaining organic matter.

Given that mine tailings are characterized by extremely low organic carbon content as compared to natural soils, initiating its build-up can represent a significant step towards restoring the disturbed land (Bradshaw, 1983; Wali, 1999; Néel et al., 2003; Larney & Angers, 2012). The organic addition will directly ameliorate substrate fertility through an increase in exchange capacity, reduction in bulk density, as well as better water infiltration and retention characteristics (Foley & Cooperband, 2002; Camberto et al., 2006; Larney & Angers, 2012). Furthermore, providing this initial carbon source to mine tailings could dramatically increase their biological activity and have important feedback relations on the overall chemical and physical fertility of the tailings (Ibrahim & Goh, 2004; Green & Renault, 2008; Larney & Angers, 2012; Fageria et al., 2011).

5.1.3 Inorganic nitrogen content

The relative increases in inorganic nitrogen following amendment applications were as expected with the high fertilizer treatment showing elevated levels as compared to all other treatments. However, one month following fertilization, the plots that had 2.5 times the N applied (high fertilizer treatments) had only 1.6 times the N content of the other treatments. This relative discrepancy indicates that the fertilizer was leached rapidly from the tailings, and that the high fertilizer rate amendment treatment was excessive given the lack of established vegetation at the time of fertilization and the inability of this substrate to retain nutrients (Fageria et al., 2011). No visual symptoms of nutrient deficiencies were witnessed during the 2009 field season, indicating that these nitrogen levels were indeed sufficient for the establishment and growth of the seeded plant species. By the end of the first growing season, the wood chips amended tailings had returned to nitrogen content equal to their initial unamended content, while the other treatments remained N-enriched relative to their initial levels. Given the high C to N ratio of the wood chips amendment, it was expected that a significant proportion of the applied nitrogen would be bound in microbial biomass during the immobilization phase of the decomposition of this carbon-rich organic material (Brady & Weil, 2008). The rate at which wood wastes are decomposed can be influenced by the nitrogen availability in the soil substrate, with higher nitrogen contents being associated with a more rapid decomposition (Shuman & Belden, 1991). That being the case, it can be predicted that at the end of the first growing season, when soil nitrogen had returned to their initially low levels in the wood chip amended tailings, the rate of their decomposition may be retarded by nitrogen limitation.

While the nitrogen levels of the tailings amended with papermill sludge and low fertilizer as well as the ones amended with only low fertilizer remained similar throughout the course of the experiment, it is important to consider the amount of nitrogen retained in plant biomass. Papermill sludge amended tailings allowed for the establishment of significantly greater ground cover as well as biomass yields relative to the low fertilizer amended tailings, indicating a greater access or ability to make use of nitrogenous resources during plant development. The highest biomass yield was observed on papermill sludge amended plots seeded with grasses and *Medicago sativa* (93.9 g of shoots and 78.7 g of roots per m^2), while the lowest one was observed on low fertilizer amended plots left unseeded (5.7 g of shoots and 4.1 g of roots per m^2). Assuming average dry tissue nitrogen content of 1.2% (Fageria et al., 2011), the vegetative biomass accounted for between 0.12 and 2.07 g N m^{-2} (equivalent to 1.2 and 20.7 kg N ha^{-1}, respectively); the most successfully vegetated plots removed approximately 20% of the 100 kg N ha^{-1}applied through plant uptake (see Appendix c). In addition, plants established on papermill sludge amended tailings have access to an additional nutrient resource, the sludge itself (35.3 mg kg^{-1} nitrate-N content). The papermill sludge obtained from the Tembec papermill was described in detail in Section 3.4, and contains secondary sludge as one third of its composition. Secondary papermill sludge is enriched with nitrogen, phosphorous, and other mineral nutrients as a result of their addition during active biological wastewater treatment. These nutrients are immobilized by microbial populations necessary during treatment process and remain bound in organic form as a component of microbial biomass (Watson & Hoitnik, 1985; Rashid et al., 2006). Because of the relatively high fertility of secondary papermill sludge, it is very often combined with other, lower quality waste sludge streams in order to produce a material capable of being applied to soils without adverse affects to plant growth (as was the case in the current study) (Rashid et al., 2006). While some plants and many soil organisms have some capacity for the direct uptake and use of organic forms of nitrogen (amino acids and low molecular weight polypeptides), most of the nitrogen supplied for plant growth is in inorganic forms, most commonly nitrate or ammonium (Hopkins, 1999; Persson et al., 2003). As such, in order for plants to have access to the applied organic nitrogen present in the papermill sludge, the organic molecules must first be mineralized to an inorganic form. At the application rate of papermill sludge used in the experiment (3.70 kg m^{-2} sludge contains 123.50 mg nitrate-N m^{-2}, equivalent to 1.20 kg ha^{-1}), the nitrogen enrichment was quite insignificant relative to the initial 100 kg N ha^{-1} fertilizer application. That being said, the slow release nature of this nitrogen source (requiring mineralization) may provide an effective method for providing vegetation with low levels of soil N while avoiding the rapid leaching of nitrogen from the tailings.

Throughout the course of the experiment at the Gunnar mine tailings, there was no observable effect of the sub-plot level seeding treatments on soil nitrogen levels. Sub-plot level differences in available nitrogen were expected for a number of reasons. First, those sub-plots that were seeded could have exhibited a greater loss in available nitrogen relative to those sub-plots that were left un-seeded given that normal plant growth requires access to soil N (Fageria et al., 2011). Furthermore, since the papermill sludge amended tailings exhibited by far the greatest overall plant growth, it was expected that the N removed from the soil pool as a result of plant uptake and growth would be greatest for this amendment treatment. If we combine the tailings N levels of the October 2010 sample date with the N present in the vegetation (assuming 1.2% N dry weight) at the time of biomass harvest (late August, 2010) we can get a more complete picture of the N dynamics of the amended tailings. As previously outlined, the amount of N bound in plant tissues varied greatly between amendment and seeding combinations, accounting for between 0.12 and 2.07 g N m^{-2} (for the papermill sludge amended tailings planted with grasses and *M. sativa* and the 100 kg N ha^{-1} amended tailings left unseeded, respectively). The N bound in plant tissue will return to the soil pool during plant litter decomposition where it can again be used within the plant-soil system (Barton & Northup, 2011). As such, the incorporation of soil N into the cover vegetation represents a valuable method through which available nitrogen is retained within a plant-soil system, thereby reducing negative leaching effects.

The inclusion of *M. sativa* (inoculated with a *Rhizobium* spp.) in the seed mix for two of the four seeding strategies could have also influenced soil nitrogen levels, though no effect was detectable during this study. *M. sativa* is a leguminous plant which forms a symbiotic association with nitrogen fixing bacterium (*Rhizobium meliloti*). Nodule structures are formed (often in clusters) on roots of the plant during this mutually beneficial relationship; this is the site of nitrogen fixation (Fageria et al., 2011). Through the action of this symbiosis, the nitrogen status of the substrate may be enriched, providing not only a competitive advantage to N-fixing plant species under nutrient poor conditions, but also contributing to an overall pool of soil nitrogen feeding the plant and soil communities (Bradshaw, 2000; Barton & Northup, 2011). Considerable variation in nitrogen fixation rates exist for *M. sativa* growing the field, ranging broadly from 78 to 350 kg N ha^{-1} yr^{-1} (Carlsson & Huss-Danell, 2003; Fageria et al., 2011). During the 2011 field season, all *M. sativa* nodule clusters used in the acetylene reduction assay were actively fixing N, and differences were found based on amendment treatment. Very little fixed nitrogen will escape live plant tissue and any enrichment to the soil N pool may not be observable until plant roots and nodules senesce and are turned over in the soil (Russelle et al., 1994). During the establishment year, fine root turnover in the upper 30 cm in a loamy sand soil for four alfalfa germplasms averaged 48%, and it was estimated that as much as 60 kg N ha^{-1} had been released through fine root turnover during one year (Goins & Russelle, 1996). The overall low establishment witnessed for *M. sativa* seeded sub-plots in many cases may have contributed to our inability to observe a seeding treatment effect on available nitrogen. Sparse establishment would leave relatively large spaces of tailings outside of the zone of influence created by the nodulated plant roots. Finally, the low background levels of available nitrogen present in the Gunnar tailings may have diluted the effect of nitrogen fixation over the large sample area of the plot. If for example, only 3 of the 8 soil sample cores taken for fertility analyses were within an area directly influenced by the action of *M. sativa* roots, it would be difficult to observe an overall effect of the roots that happened to be captured by the sampling design.

5.1.4 Available phosphate content

Only the Gunnar mine tailings amended with either papermill sludge or high fertilizer rate responded with an increase in available phosphate relative to their pre-treatment levels. Given the fact that inorganic phosphate was included in the fertilizer mix chosen as part of all amendment treatments (25-5-10 NPK, at a minimum of 20 kg P ha^{-1}), it was expected that the available levels of phosphate would be increased following amendment application. Additionally, the papermill sludge amendment application represented an input of 1.6 kg phosphate-P ha^{-1}. Therefore, further increases in available phosphate were expected for this amendment treatment. The lack of noticeable increase in available phosphate in the tailings where wood chips and low fertilizer, or just low fertilizer were applied may suggest mineral and/or biological fixation and immobilization. During decomposition of an organic material, the saprophytic community growth will require both N and P (Brady & Weil, 2008). During microbial immobilization, P becomes unavailable for plant uptake; however the eventual turnover of decomposer biota will release the bound nutrients and lead to an enrichment of the resource. In contrast to this process, mineral fixation events involving both primary mineral surfaces and dissolved mineral constituents can lead to the removal of available P that does not readily become available over the short term (Brady & Weil, 2008). As a result of fixation processes occurring under both acidic and alkaline conditions, the maximal phosphate availability is expected to occur under a neutral pH range (pH = 6-7) (Conyers & Moody, 2009). In addition to substrate pH, the amount and type of clay mineralogy and humic substances will play a role in determining soil P-fixing capacity and in turn the amount of phosphorous available for plant uptake (Brady & Weil, 2008). Nevertheless, symptoms of plant nutrient deficiencies attributable to a lack of phosphorous were not observed in the cover vegetation, and given that the background levels of phosphate (~0.35 mg kg^{-1}) would be adequate for plant growth in most cases (Fageria et al., 2011), it is unlikely that phosphorous was limiting establishment or growth of the plant community.

Organic soil amendments as well as inorganic fertilizers have been used under many conditions to improve the plant available nutrients, including P, in many naturally and anthropongenically degraded soils and mine wastes (Tester, 1990; Feagley et al., 1994; Carpenter & Fernandez, 2000; Reid & Naeth, 2005b;

Gallardo et al., 2010). It should not be a surprise that returning organic matter to soils would improve their P content since the removal of plant biomass from lands is the primary source of loss of P from natural soils (Brady & Weil, 2008). In addition, given the rapid rate and strong capacity for weathered soils of various types to bind inorganic forms of P, the use of organic substances to provide a slowly released form of plant available-P is a valuable option for improving nutrient dynamics in many situations (Conyers & Moody, 2009; Larney & Angers, 2012). Sewage sludge, peat moss, animal refuse, as well as various types of papermill sludge have been shown to significantly increase soil P content in natural and constructed soils as well as mineral wastes from mining activities (Tester, 1990; Feagley et al., 1994; Carpenter & Fernandez, 2000; Reid & Naeth, 2005b; Szczerski et al., 2012; Gallardo et al., 2010). The solubility of the organic P present in the organic matter will influence its movement into the soil substrate, and soil characteristics relating to the adsorption and fixation of mineralized organic P will influence how much of this becomes available to the plant community (Brady & Weil, 2008). In the case of the papermill sludge used in the Gunnar mine tailings revegetation experiment, the phosphate present would have been bound in microbial and invertebrate biomass (at an unknown stage of decomposition) as a result of the active biological wastewater treatment processes that produced the secondary sludge. The tailings amended with papermill sludge witnessed enrichment in available phosphate within the first month of their incorporation into the substrate, indicative of a relatively rapidly solubilized organic P source. Land amendment with papermill sludge has consistently improved available phosphorous levels, and the rapid solubilization of this material is not uncommon. In a study examining the effect of papermill sludge incorporation into a volcanic ash derived soil (at a rate of 10 to 40 Mg ha^{-1}) on soil properties and plant growth responses showed that after 105 days, all treatment rates exhibited significantly increased available P (Gallardo et al., 2012). Another study looking at the suitability of various organic additions for kimberlite tailings revegetation found that papermill sludge (4 cm) raised available P from 10 to 24 mg kg^{-1} upon application (Reid & Naeth, 2005b).

5.1.5 Available potassium content:

The high fertilizer amended tailings were the only treatment to respond to the inorganic fertilization with an observable increase in available potassium, despite all treatments having at least 40 kg K ha^{-1} applied at the onset of the experiment. Furthermore, the tailings that were amended with papermill sludge would have received an additional 17 kg K ha^{-1}as a result of the K present in the amendment. That being said, like P, mineral fixation events are common in calcareous, alkaline substrates, resulting in the removal of K from the active, available pool (Brady & Weil, 2008). Also like in the case of P, the amount and type of clay minerals present in the soil will affect availability through the incorporation of fixation of available K to clay structures. In the case of the K applied to the Gunnar mine tailings as part of the high fertilizer rate amendment treatment, the loss of potassium observed over the 2009 growing season was likely attributable to leaching from the tailings substrate that did not have sufficient holding capacity (CEC < 1.0 meq 100 g^{-1}). Given that the excess potassium applied to these plots did not appear to have been used in plant growth (high fertilizer rate treatment had a plant cover equivalent to the low fertilizer rate treatment), and minimal erosion was witnessed throughout the relatively wet summers of 2009 and 2010 (see Appendix A and B), leaching was the most likely process removing the applied potassium. Our inability to observe an increase in potassium in many treatments despite having applied a considerable source may be explained both by plant use during the early establishment of the seeded cover species, as well as mineral fixation events precipitated by the slightly alkaline conditions in the tailings substrate. Furthermore, while the papermill sludge amendment did contain potassium, it is possible that it was present largely in the unavailable pool, in the form of primary minerals that may have settled out of the primary wastewater during initial clarification processes. In addition to organic matter of plant origin, primary clarification will generate a primary sludge containing significant quantities of soil and sediment that has been cleared from wood during processing (personal communication with Tembec waste facility manager).

Over the course of this experiment, symptoms of potassium deficiency were not witnessed in any experimental treatments. . Given the high level of available potassium in the Gunnar mine tailings, both prior to and following amendment application, relative to other potentially growth limiting plant macronutrients (nitrogen and phosphorous), it is unlikely that this nutrient has affected plant establishment or growth over the course of this experiment.

5.1.6 Total metal content

The level of several elements in the Gunnar tailings increased or decreased over the course of the first field season, with some being influenced solely by the sample date while others showed an interaction with the amendment treatment and the sample date. Many of the elements whose levels changed during the 2009 field season represent essential micro and macronutrients in plants and soil organisms, including Mg, Fe, Zn, Cu, Mo, Ni, and Mn. (Brady & Weil, 2008). In fact, four of these seven essential nutrients decreased over time (Mn, Mg, Ni, Zn); it may therefore be hypothesized that the loss of these elements from the substrate could be in part a result of plant uptake. That being said, a reverse trend was observed for copper and molybdenum, with the levels of these metals increasing in the tailings substrate over the course of the 2009-field season. The changes in elemental composition could also be due to the loss of certain elements through leaching or erosive movement off site either as soluble substances or as eroding sediment. Leaching processes have been described in many natural settings and have been exploited as a method of recovery of potentially harmful process chemicals as well as extractable remnant minerals (US EPA, 1994; Lottermoser, 2003). The solubility and speciation of metallic elements is, however, strongly influenced by ambient pH levels, with the availability of many metallic ions increasing considerably under acidic conditions. In fact, in basic substrates many of these ions will form insoluble complexes (most commonly hydroxides, such as iron hydroxide)(Brady & Weil, 2008). Since plants take up most of these elements strictly in their ionic form, conditions that favor the solubilization of these ions and induce more readily absorbed speciation will drastically affect availability (Brady & Weil, 2008). Given the neutral to slightly alkaline pH range exhibited by the Gunnar mine tailings, it is unlikely that many of these elements are present in plant available form and therefore it is unlikely that the loss of these elements from the tailings is a result of plant uptake. That being said, the level of several elements are above the Canadian Council of Ministers of the Environment (CCME) soil quality guidelines for parklands; these are arsenic, chromium, copper, nickel and zinc (CCME, 2007). While levels were not sufficient to elicit observable negative plant responses, the potential hazard that these levels of metals present for human and environmental health underscores the importance of stabilizing these tailings and minimizing their movement off-site.

5.1.7 Cation Exchange Capacity

The low cation exchange capacity of the Gunnar mine tailings is common for this type of mine waste, and can be explained by the low clay content as well as the lack of organic matter of the tailings. Highly weathered soils, as well as sandy, organic poor substrates can commonly possess cation exchange capacities of less than 4 meq 100 g^{-1} (Brady & Weil, 2008). At these levels of cation exchange capacity, the soil substrate may not be capable of retaining whatever mineral nutrients are applied, or become available through mineralization. In fact, in substrates with exchange capacities of less than 4 meq 100 g^{-1}, an increased susceptibility to leaching of water soluble ions may be observable, leading to environmental concerns as it relates to aquifer contamination (Fageria et al., 2011). In addition to the harmful repercussions of loss of mineral ions, when soil substrates are not capable of binding and retaining plant nutritional resources, plant growth is limited and overall ecosystem productivity is restricted. Given the lack of organically derived plant nutrients in the unvegetated mine tailings, the lack of capacity for cation exchange will likely limit the potential for this substrate to satisfy the nutritional demands of a permanent erosion controlling cover community.

The observed increase in cation exchange capacity of the amended Gunnar mine tailings followed the same trend as the organic carbon levels with the effect appearing after one year in papermill sludge amended tailings and two years in the wood chips amended tailings. Organic matter additions to soils and mine tailings have been consistently shown to improve cation exchange capacity (Carpenter & Fernandez, 2000; Ibrahim & Goh, 2004; Camberto et al., 2006). In addition, in natural systems where an initially infertile substrate is gradually colonized by natural vegetation leading to a buildup of organic matter, this increase in organic matter directly correlates with an increased exchange capacity (Anderson, 1977; Schafer et al., 1980). In fact an increase in organic matter of 1% roughly correlates with an increase in cation exchange capacity of 1.7 meq 100 g^{-1} in soil substrates (Fageria et al., 2011). The relative influence that the organic matter addition will have on this soil response will be dictated both by the specific characteristics of the substance, as well as the rate at which it is applied (Carpenter & Fernandez, 2000; Camberto et al., 2006). Papermill sludge (including primary, secondary, and de-inking sludge) has been applied to both degraded agricultural lands as well as mine wastes leading to improvements to soil organic carbon, and as a corollary improvements to cation exchange capacity (Reid & Naeth, 2005b; Camberto et al., 2006). The effect of papermill sludge addition was similar to that of the addition of peat moss in a study looking at the potential of these structural amending materials in the revegetation of kimberlite tailings (Reid & Naeth, 2005b). Working with gold mine tailings of relatively similar physical composition, Ibrahim and Goh (2004) found that additions of humic substances were significantly correlated with increases in organic carbon as well as cation exchange capacity. As a result of the importance of cation exchange on fertility as well as the leaching potential of a substrate, this characteristic may significantly influence the revegetation of both naturalized and managed mine soils.

5.1.8 Water stable aggregate distribution

The initial amendment and cultivation of the Gunnar mine tailings lead to an improvement in water stable aggregation in all treatments, with papermill sludge amended tailings showing the greatest increase in total aggregates. Soil aggregates are formed both by biotic and abiotic means; abiotic processes are most important on the small scale and are predominantly related to the activity of clay minerals, while biological processes act on a larger scale and are the dominant in sandy soils (Brady & Weil, 2008). Furthermore, depending on field moisture content as well as other edaphic factors (e.g. organic content, amount and types of clays, existing structure), mechanical tillage of natural soils can encourage either the creation or the destruction of macro aggregates (Ojeniyi & Dexter, 1979; Tisdall & Adem, 1986). The coarse texture and low organic matter content of the Gunnar mine tailings suggest that the dominant processes acting to create and stabilize macro-aggregates are biological in nature. In natural soils, these include the burrowing and molding activity of soil organisms, the production of extracellular substances that act as binding agents by soil organisms, as well as the physical entanglement of substrate particles by roots and fungal hyphae (Brady & Weil, 2008). That being said, given that some increase in aggregation was observed even in those plots where little plant establishment occurred, some abiotic processes are likely contributing to aggregate stability. These processes are likely attributable to the clay minerals (which represent a very small proportion of the tailings) and their interaction with other mineral constituents that encourage flocculation (multivalent cations, other clay platelets or humic substances) (Brady & Weil, 2008).

The addition of organic materials of many types has lead to an increase in substrate aggregation, with the magnitude and duration of the effect being related to the amount and nature of the material as well as the soil substrate type (Ibrahim & Goh, 2004; Price & Voroney, 2007; Tahboub et al., 2008). Once aggregate formation has occurred, the process of stabilizing these aggregates can commence, where any process that increases cohesion between substrate particles act to stabilize the aggregate structure (Chenu & Cosentino, 2011). Both papermill sludge and wood chips have been shown to improve aggregate stability under field conditions when being amended to soils (Price & Voroney, 2007; Tahboub et al., 2008). Tahboub et al., (2008) found that the incorporation of wood chips into the soil resulted in a rate dependent effect of increasing stability in aggregates in the 0.5 – 1.0 mm size class. This study suggested that slowly decomposable materials, such as wood wastes, have superior long term capacity to stabilize soil aggregates as a result of the relatively high content of hydrophobic substances, such as lignin, that degrade over long periods of time (Tahboub et al., 2008). Similarly, in a study examining the effects of high C to N ratio (>100) papermill sludge application to several soil types, a beneficial effect of application on aggregate stability developing over time in a rate dependent fashion was found (Price & Voroney, 2007). Again, the effect was not apparent at all loading rates after just one year, however after three years all rates of application showed significantly more stable macro-aggregates. The delay in effect can again be attributable to the slow decomposition rate acting on this amendment as a result of its high C to N ratio, which indicates a preponderance of carbon rich, recalcitrant materials such as lignin (Price & Voroney, 2007). Specifically as it concerns organic amendment to mine tailings, in many cases the same benefits of improved aggregate formation and stabilization has been observed (Ibrahib & Goh, 2004; Green & Renault, 2008). The increase in aggregation is directly related to the increase in organic carbon and the effect of organic carbon on soil microbial activity (Camberto et al., 2006). This improvement may be seen rapidly (in the case of readily decomposable materials), or after a significant lag time in the case of materials with high C to N ratio (Camberto et al., 2006; Tahboub et al., 2008).

5.1.9 Bulk density

The results of the Gunnar mine tailings study agree with the general trend of decreasing bulk density over time during spontaneous succession of new or newly disturbed substrates as well as in the years following active revegetation projects

(Crocker & Major, 1952; Tester, 1990; Wali, 1999; Price & Voroney, 2007). In natural succession of newly deposited or exposed substrates, regardless of the rate of plant invasion, the eventual effect of the vegetative community on bulk density is to decrease the density as a result of the action of the plant roots as well as soil biota (Jenny, 1958; Wali, 1999). In fact, during the natural colonization of the Gunnar mine tailings, a decrease in soil compaction was observed with time since vegetative establishment. In this case, the magnitude of the decrease was greatest over the first five meters (most recently colonized) but continued along the vegetative chronosequence (Young et al., 2012). As plant roots proliferate the substrate, through normal growth as well as the eventual senescence and decomposition of tissues, pore space is developed. In addition, plant roots feed and fuel other soil biological activity during their lifetime as well as their decomposition while they provide organic substances to the soil. Processes such as burrowing, physical entanglement, and extracellular polysaccharide production function in aggregate formation (thus pore space creation) and they depend on organic substances present in the soil (Chenu & Cosentino, 2011).

In many cases, the application of organic amendments to natural soils has not provided overarching improvements to bulk density. The chemical and physical characteristics of the amendment (as it relates to their specific density), as well as their indirect effect on bulk density through an improvement in overall soil fertility and biological activity will dictate whether a lowering of the substrate bulk density will be observed (Camberto et al., 2006; Tahboub et al., 2008). In the current study, the decrease in bulk density across amendment treatments over time suggests that while the amendments may have an indirect effect on bulk density through increased plant establishment, the addition of the low rate of these materials did not itself affect bulk density. In a study looking at the effect of fresh pecan (*Caryaillinoinensis* (Wangenh.) K. Koch), wood chip addition to a silty soil, Tahboub et al., (2008) did not witness a reduction in soil bulk density, regardless of the rate of incorporation. The lack of response in terms of lowering the bulk density was attributed to the fact that this organic material did not contribute any readily decomposable substances that would feed the microbial community resulting in decomposition of the added material as well as encouraged aggregate formation and stability (Tahboub et al., 2008). Similarly, the incorporation of more readily degradable organic wastes (sewage sludge, mixed papermill sludge or de-inking sludge, as well as animal manures) has resulted in the lowering of the substrate bulk density (Tester, 1990; Price & Voroney, 2007; Green & Renault, 2008). However, in many cases it is difficult to determine the effect of the organic matter addition removed from the effect of plant growth, and potentially improved plant growth on lowering bulk densities.

5.2 Plant growth

Robust seeded cover vegetation was established within the first field season on tailings amended with papermill sludge and a low rate of fertilizer; no other amendment allowed for equivalent growth. Plant establishment and growth was similar for all other amendment treatments, although the high fertilizer rate treatment did not allow for the survival of any *Picea mariana* seedlings. Seedlings survival was low across the entire experiment, and there was a beneficial effect of being planted along with a seeded grass cover after the first winter. All plots were invaded by volunteer vegetation to a greater or lesser extent depending on the tailings amendment treatment.

5.2.1 Ground cover establishment and plant productivity

Vegetative cover can be established on the non-acid generating mine tailings left by the Gunnar gold mine through minor seedbed improvements (fertilization and cultivation) and direct seeding of herbaceous agronomic plant species. In addition, this establishment was enhanced by further amendment with papermill sludge leading to a more productive and robust vegetative cover. As discussed above, this organic amendment lead to a rapidly observable increase in organic carbon and available phosphate, presumed nitrogen enrichment, in addition to overall improvements to cation exchange capacity, bulk density, and water stable aggregation. These improvements to substrate properties permitted greater plant growth through improved nutrient and moisture dynamics, soil biological activity, as well as by limiting erosion and compaction (Fageria et al., 2011). Overall, cover increased during the first two years of the study after which some dieback was apparent, likely indicative of soil nitrogen levels insufficient to support the established vegetation. The maintenance of a productive vegetative cover on mine tailings or other disturbed substrates lacking nutrient capitol may, in many cases, require continual fertilization (Bloomfield et al., 1982; Johnson et al., 1994). This is especially true as it concerns agronomic species with moderate to rapid growth rates and moderate to high nutrient requirements, such as those selected for this study (USDA, NRC, 2012). One method to overcome the requirement for long-term fertilization is the top-dressing and/or the incorporation of a slowly decomposing source of soil nutrients to the substrate. Papermill sludge has been described as having a two-phase decomposition model; with rapidly solubilized substances degrading rapidly while the more recalcitrant fractions will degrade slowly over the many years that follow (Camberto et al., 2006; Rashid et al., 2006). The decomposition of these materials by the saprophytic soil biota will bind the nutrients in organic form before eventually being mineralized and made available for plant uptake (Rashid et al., 2006). In addition to providing a slow-released source of nutrients, increasing organic content will improve the substrates ability to bind and retain nutrients, lessening the requirement for ongoing inputs (Fageria et al., 2011). This increase in nutrient holding capacity (partially described by the improved cation exchange capacity) was witnessed in the Gunnar mine tailings amended with papermill sludge, and further increases in this property can be expected as senescent plant tissues are incorporated into the soil.

The greatest levels of plant cover were found when grass species were seeded as part of the seed mix. When papermill sludge amended plots were seeded with the grass mixes, a robust and consistent vegetation cover was established within the first growing season. The grass species selected for use in this study are both warm season species commonly used in reclamation as a result of certain desirable characteristics. For example, *Festuca rubra* exhibits a rhizomatous growth habit, a feature that allows this plant to invade adjacent voids in vegetation and create a dense cover and root matrix capable of significant erosion control (Raven, 2005; USDA, NRC, 2012). Conversely, while *Agropyron trachycaulum* exhibits a bunch growth with little rhizomatous spread. This plant is a high biomass producer and has a rapid growth rate, and low moisture requirements (USDA, NRC, 2012).

While the rapidly established, robust vegetative cover will assist greatly in erosion control, in some cases, such a dense cover can in fact be detrimental to reclamation goals. When reclaiming mine spoils and wastes for forestry services, seeding a tree-compatible ground cover can be a critical consideration influencing tree establishment and growth. Generally speaking, a tree-compatible cover is sparse, containing plant members that will not out-compete tree seedlings for soil moisture and nutrients, as well as access to direct sunlight (Burger et al., 2009). Similarly, when reclaiming disturbed lands to a native plant community, a sparse cover with considerable bare earth can allow for the invasion of native plants from adjacent communities (Polster, 1991). In the study at the Gunnar mine tailings, papermill sludge amended sub-plots that were left unseeded encouraged the greatest level of volunteer vegetation. Again, considering the improvements to soil chemical and physical fertility as a result of this amendment, it is not surprising that invasion would occur more rapidly in this treatment. On the other hand, those sub-plots amended with wood chips and left unseeded resisted the invasion of volunteer vegetation throughout the experiment. This effect can be attributed to the rapid immobilization of available soil nitrogen in microbial biomass when these carbon-rich materials are applied to a substrate. Without sufficient soil nitrogen, volunteer plant species that do not have the capacity for N-fixation will struggle to establish and may exhibit poor growth rates or fail to complete their life cycle.

5.2.3 *Picea mariana* seedling survival

Given the harsh substrate conditions in the Gunnar mine tailings, the overall poor *Picea mariana* seedling survival rate was to be expected. While the Gunnar mine tailings reliably had a pH in excess of 7.0, *P. mariana* has a preferred pH range of 4.7 – 6.5, a factor with considerable consequence on plant mineral nutrition (Raven et al., 2005; USDA, NRC, 2012). As discussed in section 5.1.6, the bioavailability of many important plant nutrients as well as potentially harmful metals is greatly influenced by pH levels (Brady & Weil, 2008). At the neutral to slightly alkaline pH of the Gunnar mine tailings, we would expect to witness a decreased availability of micronutrients such as boron, manganese, copper and zinc (Brady & Weil, 2008). Furthermore, under basic conditions phosphorous present within the substrate may be exposed to mineral fixation processes, thereby removing this important nutrient from the available pool (Brady & Weil, 2008). In addition to the potential nutritional impediment presented by the tailings environment, the Gunnar tailings are highly susceptible to drying out depending on prevailing climatic and water table conditions. Given that *P. mariana* is characterized by a very poor drought tolerance, this species is unlikely to survive in situations where soil moisture becomes limiting (USDA, NRC, 2012). Looking at the natural succession of these mine tailings to native forest habitat, it is clear that the recruitment of both *P. mariana* and *Larix laricina* must be preceded by pioneer vegetation (Young et al., 2012). Under natural conditions, *P. mariana* or *L. laricina* establishment was never seen within the first five meters of encroaching vegetation, where the plant community was dominated by herbaceous and shrubby pioneers. The clear reduction in terms of *P. mariana* seedlings establishment witnessed when tailings were amended with a high fertilizer rate was not expected, nor fully explained by any individual measurement. Upon incorporation of the high rate of fertilization, it is possible that the increased conductivity of the soil solution created osmotic impediments to the acquisition of water from the substrate, or direct toxicity through excessive ionic stress (Hopkins, 1999). This effect would be particularly detrimental to the health of seedlings given that *P. mariana* is salt intolerant and has a high moisture use during its active growing period, (USDA, NRC, 2012). In fact, in a study examining the effect of increasing levels of soluble salts on the germination and survival of several common boreal tree species, Croser et al., (2001) found that exposure to levels of NaCl or Na_2SO_4 in excess of 100 mM (equivalent to approximately 11.50 dS m^{-1}) significantly reduced emergence and survival in *P. mariana* seedlings. An alternative hypothesis is that the high rate of fertilization may have stimulated shoot and needle growth rather than root production, as sometimes observed in the case of fertile soils, resulting in an inadequate root system to survive the harshness of over-wintering (Davidson, 1968: Canham et al., 1996). Exacerbating the issues of having an inadequately established root system is the fact that members of the Pinaceae family commonly host ectomycorrhizal fungi on their roots creating a symbiosis that endows significant foraging capacity as well as the ability to endure certain environmental stresses

(Barton & Northup, 2011). The seeded vegetation used during the Gunnar revegetation study did not form this association with ectomycorrhizal fungi, and as such would not have promoted the ingress of these organisms nor supports their establishment should they arrive on-site. That being the case, the development of this symbiotic relationship will be retarded until viable propagules are delivered within the root zone of these tree seedlings (Reeves et al., 1979).

As previously mentioned, the seeding of a tree-compatible ground cover is an important factor to consider when approaching revegetation to a forested environment. A tree-compatible ground cover has been described as being sparsely seeded, with slow and low growing native herbaceous plants in order to minimize competitive interactions (Burger et al., 2009). Following the first winter, there were decreases in *Picea mariana* survival across all amendments; however those seedlings planted in sub-plots with a seeded grass component were more likely to have survived the winter. This phenomenon is not unexpected given that natural establishment of this tree species on site always follows significant establishment of a pioneer community (Young et al., 2012). The protective effect of a grass cover likely relates to the soil binding capacity of the grass roots reducing erosion and soil movement in the vicinity of the planted seedlings. Various types of tailings movement resulted in observed seedling mortality, these include the removal of rooting medium and plant materials by water erosion, the burying of plant materials by windblown particles, as well as the heaving of soil plugs from the tailings during freeze-thaw. This finding illustrates the potential positive impact that a strong grass cover component can have on seedling survival, however the negative impacts were also witnessed within a short time. By the second field season, in papermill sludge amended plots, it became difficult to locate the seedlings among the tall and dense herbaceous cover. This direct competition for soil moisture, nutrients and access to sunlight will eventually result in greatly reduced growth rates if not further reduction in survival.

5.3 Plant stress

The pigment contents of the seeded cover species were influenced in some cases both by amendment as well as the seeding treatments, though the effects were not consistent across species or pigment molecule. Proline levels as well as gas exchange parameters were highly variable during sampling, and no effect of either amendment or seeding was apparent. Symbiotic nitrogen fixation in *Medicago sativa* plants was significantly enhanced in the high fertilizer amended tailings, while all other amendments yielded similar levels of N-fixation. The elemental composition of *M. sativa* and *Agropyron trachycaulum* shoot tissue differed at the time of sampling, and for both species amendment treatment did not seem to control these levels.

5.3.1 Pigment content

In 2010 and 2011, when tissues were collected for photosynthetic pigment content determination, both *Medicago sativa* and *Agropyron trachycaulum* responded to certain amendment applications through increased pigment levels. The pigment levels witnessed in these species were tied to the available nitrogen present in their substrate, with vegetation from the high fertilizer rate or papermill sludge amended tailings exhibiting the greatest levels. Leaf nitrogen is primarily present in proteins and enzymes of the Calvin cycle as well as proteins and pigment complexes of thylakoid membranes (Evans, 1989). As a result, when nitrogen supplies are lacking these molecules cannot be synthesized *de novo*, and their overall content within the leaf tissue will be reduced relative to the same plant grown in an N-rich environment (Venanzi et al., 1993). Many authors who have examined the effect of nitrogen nutrition on photosynthetic capacity, crop production and yield described the direct relationship between increasing soil nitrogen levels and foliar pigment content (Venanzi et al., 1993; Lawlor, 1995). In fact, the correlation is strong enough that for many crop species, photometric analysis of the leaf tissue can sufficiently describe foliar N content as well as fertilizer requirements (Wood et al., 1992; Güler & Büyük, 2007; Reis et al, 2009). In a similar study, the fertility enrichment provided by papermill sludge amendment to mine tailings has lead to increases in the pigment content of existing vegetation (Green & Renault, 2008). In addition to soil nitrogen, other important plant nutrients may influence the levels of pigment molecules in plant tissue; these include micro and macronutrients required for pigment synthesis such as phosphorous, magnesium, manganese and iron (Raven et al., 2005).

Beyond the direct and apparent effect of soil nitrogen on pigment contents, there appeared to be an effect of seed mix on the content of certain pigments in some seeded plant species. For *M. sativa* a greater carotenoid content was witnessed in plants seeded without the grass mix whereas *Festuca rubra* had higher chlorophyll and carotenoid contents when seeded without *M. sativa*. As a result of the antioxidant role of carotenoids, these molecules provide protection against the damaging effects of photooxidation (Raven et al., 2005). In situations where *M. sativa* was planted in the absence of other cover vegetation it is possible that these individuals would be exposed to a greater level of direct photo-irradiance requiring greater quantities of antioxidants (such as carotenoids) to offset the greater free radical concentration (Hopkins, 1999). In addition, competition between plants can affect total foliar pigment content when shading occurs, and can also influence which pigment molecules the plant invests in (Dai et al., 2009; Li et al., 2010). Shaded plants may invest in the production of greater quantities of chlorophyll molecules as compared to non-shaded individuals in addition to lowering its internal chlorophyll a to chlorophyll b ratio (Evans, 1989; Dai et al., 2009; Li et al., 2010). In the study at the Gunnar mine tailings, the only amendment that allowed for great enough plant productivity to create a shading effect was the papermill sludge amended tailings (with cover value ranging above 100%). Considered individually, no seeded plant species appear to have responded to possible shading in papermill sludge amended tailings with an increase in total pigment content. Conversely, both grass species had a lower chl a to chl b ratio when planted without *M. sativa* not only in the papermill sludge amended plots, but also across all amendment treatments. While dense swards composed of both grass species and *M. sativa* were established on the Gunnar mine tailings, shading is not likely to be occurring as a result of their similar overall growth form and age. Furthermore, as mentioned, competitive interaction for soil resources can also affect photosynthetic pigment content. When nitrogen is limiting to growth, being seeded with a competitively dominant species may hinder the ability to acquire necessary soil nutrients to synthesize new pigment molecules. Likewise, under moisture limiting conditions direct competition for soil reserves can impede plant growth and even disrupt membrane integrity, resulting in the loss of function in membrane bound molecules, including the vast majority of pigments (Hopkins, 1999). Given that substrate fertility and moisture availability are chief constraints on natural and introduced vegetation at the Gunnar mine site, plant competition is likely affecting pigment levels in either a direct or indirect fashion (Renault, 2006; Young et al., 2012).

5.3.2 Proline content

Foliar proline content of the seeded vegetation was not influenced by the amendment treatment, nor the seed mix at the time of tissue sampling (summer 2010). Proline accumulation in plant tissues commonly occurs in response to drought stress where it acts as an osmotic adjuster (Hopkins, 1999). In addition, plants may accumulate this amino acid in response to heavy metal toxicity, nutrient deficiency, atmospheric pollution, oxidative stress and salinity; as such it is commonly used as a plant stress marker (Hare & Cress, 1997; Gangopadhyay & Basu, 2000). The action of this molecule during instances of environmental stress is attributed to its ability to mediate osmotic adjustment, scavenge free radicals, stabilize sub cellular structures, as well as provide energy for use during the recovery period (Hare & Cress, 1997). Despite the fact that the major constraints on plant growth expected at the Gunnar mine tailings were nutritional deficiencies as well as moisture availability, neither of these factors were sufficient to elicit the accumulation of proline at levels indicating stress (Rozema et al., 1978; Irigoyen et al., 1992; Nakata, 2007). In fact, the overall content found in all species, regardless of amendment treatment were low, potentially indicating a lack of nutrients required to synthesize and accumulate proline molecules (Hopkins, 1999). Pang et al., (2003) suggest that fertilization may aid in alleviating issues relating to growth in metal rich mine tailings in part by providing the resources required to synthesize substances important during exposure to harsh abiotic stresses. Given the capacity of plants to mobilize nitrogen within their tissues, if an environmental stressor were to appear, more robust individuals such as those growing on the papermill sludge amended tailings would surely exhibit a greater capacity to synthesize and accumulate proline.

5.3.3 Symbiotic nitrogen fixation

At the time of sampling (summer 2011), the rate of nitrogen fixation (as described by acetylene reduction) in *Medicago sativa* root nodules was significantly greater where tailings had been amended with the high fertilizer rate. Except at very low levels, the application of nitrogenous fertilizers and N-rich manures generally does not stimulate nodule formation or N-fixation. In fact some studies have described a decrease in nitrogen fixation in response to nitrate applications (Waterer & Vessey, 1993; Nesheim & Oyen, 1994; Vinther & Jensen, 2000; Bollman & Vessey, 2005). Further improvements to the availability of other soil minerals can be of benefit to this process (Hopkins, 1999; Raven et al., 2005). Iron, sulfur, and molybdenum are all significant components of the dinitrogenase enzyme complex, which is the site of dinitrogen fixation (Hopkins, 1999). In addition, cobalt has been shown to influence biological N-fixation (Delwiche et al., 1961), and given the high energetic cost (approximately 25 moles of ATP per mole of N_2 fixed), it is not surprising that phosphorous nutrition can impact N fixation (Hopkins, 1999; Olivera et al., 2004). The Gunnar mine tailings amended with the high fertilizer rate as well as those amended with papermill sludge exhibited greater available phosphate levels as compared to the other amendment treatments at the final 2011 sample date. This relatively superior phosphorous nutrition may permit for the greater fixation rate in the high fertilizer amended tailings, although this effect was not witnessed in papermill sludge amended plots despite equal phosphate levels. In fact, Allahdadi et al., (2004) found that increasing additions of de-inking paper sludge improved nitrogen fixation in forage legumes during plant establishment, and had a neutral or positive effect later in the plant's life. Inherent variation between individuals sampled may explain the different response. Rates of nitrogen fixation undergo seasonal variation, generally with lowest rates witnessed at the start and end of the growing season and highest rates observed in late summer (Vinther & Jensen, 2000; Goh, 2007). This being the case, it is possible that the sample date influenced the fixation rates witnessed and that the measured fixation rate was not indicative of a maximum fixation capacity.

5.3.4 Gas exchange

All gas exchange measurements increased with increasing light intensities and the maximal rates were reached around 150, 200, and 1000 μmol photons $m^{-2}s^{-1}$ for stomatal conductance, transpiration, and photosynthetic rates, respectively. The maximal photosynthetic rate witnessed was similar to the rate observed in six *M. sativa* cultivars at 1200 μmol photons $m^{-2}s^{-1}$ under greenhouse conditions (Ni et al., 2012). Conversely, the transpiration rate and stomatal conductance of plants growing in the Gunnar mine tailings were high relative to those observed in *M. sativa* cultivars both under control and drought conditions (Ni et al., 2012). The photosynthetic capacity as well as the movement of a plant stomata that controls gas exchanges is affected by a number of internal and environmental factors, including pigment content and composition, light quantity and quality, water relations, temperature and carbon dioxide, as well as plant nutrition and overall health (Hopkins, 1999). The increased pigment content witnessed in the tissue of the papermill sludge and the high fertilizer amended plots did not correlate with an increased photosynthetic capacity at the time of sampling. The lack of notable difference in these responses between amendments suggests that once established, physiological responses relating to gas exchange in *M. sativa* plants are unaffected by the apparent differences in tailings fertility and plant growth. The relatively high transpiration and conductance rates may imply that, across the tested light intensities, access to adequate concentrations of carbon dioxide rather than soil moisture may be limiting to the process of photosynthesis. It is important to note that for this analysis a limited amount of replication was performed (n = 2).

5.3.5 Elemental composition

Despite having observed considerable changes in tailings properties between amendment treatments, in almost all cases these changes were not reflected by differential elemental composition in plant tissues. The relatively low content of some elements (potassium, magnesium and manganese) in *Agropyron trachycaulum* and *Medicago sativa* tissue at the time of sampling agrees with the observed dieback at this point in the study. It is possible for the timing of tissue collection to affect the content of some minerals. Studying the content of plant nutrients in several grasses, Fleming & Murphy (1968) found a reliable trend of increasing levels over the active growing period, followed by an eventual decrease late in the season. The plants were sampled in this study in late July of 2011 for elemental analysis, and given the cool season phenology of *A. trachycaulum* it is reasonable to consider that at this point in the summer this plant has completed its period of active growth.

M. sativa benefits from adequate molybdenum and cobalt availability, at least partially, as a result of their requirement during the process of nodulation and nitrogen fixation (Delwiche et al., 1961; Barton & Northup, 2011). Additionally, this affinity for uptake of these elements may increase the likelihood of uptake of elements of similar size, charge, geometry of coordination and electronic configuration through similar pathways (Fageria et al., 2011). That being the case, the levels of elements such as cobalt, molybdenum, boron, copper, arsenic, cadmium and chromium relative to *A. trachycaulum* may be indicative of this diverse mineral nutrient requirement. Other factors affecting the uptake of metals in *M. sativa* plants include soil pH, rhizobia and other soil biota as well as fertilization and life stage

(Chapin, 1980; El-Kherbawy et al., 1989; Peralta-Videa et al., 2004). Furthermore, the higher boron content present in *M. sativa* as compared to *A. trachycaulum* may simply be explained by a greater requirement of this mineral in eudicotyledon as compared to monocotyledons (Raven et al., 2005).

Chapter 6: Conclusion and Recommendations

Environmental impacts of the mining industry can be devastating when proper management practices are not carefully employed throughout a mine's lifetime, as well as upon closure. As a result of lax environmental regulation throughout much of our country's history, we have been left with a legacy of mine sites posing various levels of hazards to both natural ecosystems and local residents. The burdens to reclaim these sites and return the disturbed lands to an acceptable ecological state has and will often fall on the shoulders of the Federal and Provincial governments, and by proxy, the Canadian taxpayers. That being the case, in order to facilitate remedial efforts and to encourage the remediation of a greater number of these sites, effective low-cost techniques must be developed to assist in the process.

Mine tailings present a great impediment to the natural colonization and establishment of plant species, often requiring considerable amendment before any plant establishment can occur. The current study successfully re-introduced a robust vegetative cover to the Gunnar mine tailings with the use of a low-cost locally sourced waste material, papermill sludge. While allowing for substantial plant growth, this amendment improved soil characteristics directly related to fertility including organic content, bulk density, water stable aggregation, as well as cation exchange capacity. In fact, after the two-year study period, organic carbon levels in the organically amended tailings approached the levels found on portions of the tailings dump that had been naturally revegetated for decades. The economic importance of mining to Canada and the Province of Manitoba ensures that this industry will persist and likely continue to expand development in the decades to come. That being the case, more focus must be placed on the creation of simple, low-input reclamation strategies that stimulate the natural progression from an inhospitable substrate to one that can support a healthy plant community.

The following site and study specific recommendations have been generated as a result of this study:

- The incorporation of a small quantity of papermill sludge and a low rate of fertilizer (as described in the Section 3.4) is capable of encouraging the rapid establishment and robust growth of seeded cover vegetation
    - Improvement of many key soil properties (such as organic carbon, cation exchange capacity, bulk density and water stable aggregation) created a substrate highly conducive to plant growth
    - Higher organic matter content and better overall structure permitted plant roots to proliferate the tailings, which will lead to further improvement to, and overall development of the soil substrate
    - Prior to land application, amendments such as papermill sludge should be properly tested for the presence and abundance of noxious and total weed seeds; weed control practices will be required
    - If the tested amendment is not available in the quantities required for a full scale revegetation project, I recommend the use of a structure improving material as well as a slowly released, low rate organic fertilizer (such as composted organic wastes, manures, and/or biosolids).

- Fertilization of the Gunnar mine tailings without organic amendment resulted in poor overall establishment and growth of the seeded cover vegetation, regardless of the rate of application tested
    - Within two years, all of the maximum 250 kg N ha^{-1} had leached from the tailings, with the great majority being lost within the first year
    - The incorporation of organic matter capable of binding and holding applied nutrients in the root zone is critical if any fertilization is to be done to these tailings.
- Though not effective at promoting plant growth, wood chips incorporation into the tailings did improve certain soil properties and limited the invasion of the tailings by weed volunteer vegetation
    - To overcome the microbial immobilization of applied nutrients and provide sufficient nutrient for plant growth, very large quantities of fertilizer would be required
    - The rough surface structure that these amended tailings maintained throughout the experiment will be less susceptible to erosion
    - The eventual breakdown of the wood chips may improve fertility characteristics and lead to a substrate suitable for normal plant growth.
- *Medicago sativa* establishment did not cause a detectable increase in available nitrogen in the tailings over the two year study
    - Because of its taproot root structure, this species contributes very minimally to soil stabilization and erosion control

- The strongly competitive nature of this species, its bunched growth form and its large annual seed set do not favor the invasion of the tailings by the surrounding native vegetation
    - Native forbs with high pollinator value should replace this species in any subsequent plantings at the Gunnar mine site.
- *Picea mariana* seedlings establishment was low across the entire experiment, however the high fertilizer rate fostered conditions that resulted in no seedling survival past the first field season
    - A positive effect of being planted in conjunction with a seeded cover that included a grass component was found following the first winter in the field; this effect must be balanced against the competition that this cover vegetation will impose on the seedlings
- Planting cover vegetation with differing growth forms has clear advantages in terms of soil stabilization and overall productivity
    - In the most successful areas of cover establishment, the tall bunch grass (*Agropyron trachycaulum*) and the short rhizomatous grass (*Festuca rubra*) grew together and created a dense and complete ground cover
    - This should be expanded on this in any subsequent plantings, with an emphasis being placed on diversity of growth forms and habits allowing for the natural formation of a heterogeneous community

The Gunnar mine tailings site has provided an excellent experimental area to test the effect of various amendment techniques on a reasonably large and feasible scale. Depending on the future direction of reclamation works at this site, there is a great potential for further research, including:

- Examining the potential use of other low-cost, locally sourced amendment materials such as composts derived from plant material and/or biosolids or food residues from the nearby cottage and camp grounds (the towns of Lac du Bonnet or Bissett may also provide options)
- Depending on the decided land use, any number of native boreal plant species could be studied with respect to their capacity for use in reclamation, their response to environmental stressors, or their interactions with their environment (competition, facilitation, soil ecology and soil development)
- Relatively little attention has been given to soil biological responses during tailings amendment, or other reclamation methods (such as capping); important research points would include: characterizing soil biota present in the tailings, in any amendment materials, the changes to both following application, the succession of soil organisms or guilds during plant establishment and succession
- The examination of site use by native wildlife could provide direction for reclamation decisions (for example land form and features, plant selection, and human accessibility) as well as valuable information pertinent to wildlife conservation

- Downstream effects of the reclamation work should be studied, including the ramifications for the adjacent wetland and wetland vegetation as well as Beresford Lake and aquatic communities influenced by the presence of the mine site.

The findings of the current study may provide insight into the handling and eventual revegetation of degraded soil substrates such as non-acid generating mine tailings. The resounding improvements to substrate characteristics and the subsequent robust plant growth accompanying papermill sludge incorporation validate the effectiveness of this material as a soil amendment in degraded landscapes. While mine wastes are as unique as the geological formations from which they originate, these findings may be characteristic of drastically degraded mineral soil substrates whose major impediments to plant growth include nutrient deficiency, compaction, and a lack of organic matter.

References:

Alday, J. G., Y. Pallavicini, R. H. Marrs, and C. Martínez-Ruiz. 2011. Functional groups and dispersal strategies as guides for predicting vegetation dynamics on reclaimed mines. Plant Ecology 212:1759–1775.

Allahdadi, I., Beauchamp, C.J., Chalifour, F-P. 2004. Symbiotic Dinitrogen Fixation in Forage Legumes Amended with High Rates of De-Inking Paper Sludge. Agronomy Journal. 96: 956-965.

Anderson, D.W. 1977. Early stages of soil formation on glacial till mine spoils in a semi-arid climate. Geoderma. 19: 11-19.

Anielski, M. and Wilson, S. 2002. Counting Canada's Natural Capitol: Assessing the real value of Canada's Boreal Ecosystems. Pembina Institute, Canadian Boreal Initiative, Drayton Valley, AB; Ottawa, ON. Pp. 90.

Bamburak, J.D. 1990 Metallic Mines and Mineral Deposits of Manitoba. Manitoba Energy and Mines, Geological Services. Pp 111.

Barbour, A.K. 1994. Mining Non-ferrous Metals. In: Mining and its Environmental Impact. Edited By: R.E. Hester and R.M. Harrison. Royal Society of Chemistry (Great Britain). Pp. 1-15.

Barton, L.L., and Northup, D.E. 2011. Chapter 10: Microbial processes contributing to biogeochemical cycles. In: Microbial Ecology, First Edition. Edited by: L.L. Barton and D.E. Northup. John Wiley & Sons, Inc. Hoboken, New Jersey. Pp. 273-298.

Bayne, E.M., Habib, L., Boutin, S. 2008. Impacts of chronic anthropogenic noise from Energy-Sector Activity on Abundance of Songbirds in the Boreal forest. Conservation Biology. 22: 1186-1193.

Belden, S.E., Schuman, G.E., Depuit, E.J. 1990. Salinity and moisture responses in wood residue amended bentonite mine spoil. Soil Science. 150: 874-882.

Bell, M.A.M. and Meidinger, D.V. 1977. Native species in reclamation of disturbed lands. Proceedings of the 1st Annual British Columbia Mine Reclamation Symposium in Vernon, BC, 1977. The Technical and Research Committee on Reclamation. Pp. 143-158.

Black, T.A., Hartog, G.D., Neumann, H.H., Blanken, P.D., Yang, P.C., Russell., C., Nesic, Z., Lee, X., Chen, S.G., Staebler, R., and Novak, M.D. 1996. Annual cycles of water vapor and carbon dioxide fluxes in and above a boreal aspen forest. Global Change Biology.2: 219-229.

Bloomfield, H.E., Handley, J.F., Bradshaw, A.D. 1982. Nutrient deficiencies and the aftercare of reclaimed derelict land. Journal of applied ecology. 19: 151-158.

Blowes, D.W. and Ptacek, C.J. 1994. Acid-neutalization mechanisms in inactive mine tailings. In: The Environmental Geochemistry of Sulfide Mine-Wastes. Mineralogical Association of Canada, Waterloo, ON. Pp. 271-285.

Blowes, D.W., Ptacek, C.J., Jurjovec, J. 2003. Mill tailings: hydrogeology and geochemistry. In: Environmental Aspects of Mine Wastes. Edited by: J.L. Jambor, D.W. Blowes and A.I.M. Ritchie. Mineralogical Association of Canada. Pp 95-115.

Blundon, D.J. and Dale, M.R.T. 1990. Dinitrogen fixation (acetylene reduction) in primary succession near Mount Robson, British Columbia, Canada. Arctic and Alpine Research. 22: 255-263.

Bollman, M.I. and Vessey, J.K. 2005. Differential effects of nitrate and ammonium supply on nodule initiation, development, and distribution on roots of pea (*Pisum sativum*). Canadian Journal of Botany. 84: 893-903.

Bonan, G.B., Pollard, D., Thompson, S.L. 1992. Effects of boreal forest vegetation on global climate. Nature. 359: 716-718.

Borden, R.K. and Black, R. 2005. Volunteer Revegetation of Waste Rock Surfaces at the Bingham Canyon Mine, Utah. Journal of Environmental Quality. 34: 2234-2242.

Bradshaw, A.D. 1983. The Reconstruction of Ecosystems. Journal of Applied Ecology. 20: 1-17.

Bradshaw, A.D. 2000. The use of natural processes in reclamation – advantages and difficulties. Landscape and Urban Planning. 51: 89-100.

Bradshaw, A.D., Humphreys, M.O., Johnson, M.S. 1978. The value of heavy metal tolerance in the revegetation of metalliferous mine wastes. In: Environmental Management of Mineral Wastes. Edited by: G.T. Goodman and M.J. Chadwick. Siijthoff & Noordhoff International, Netherlands. Pp 311- 334.

Bradshaw, A.D. and Johnson, M.S. 1992. Revegetation of metalliferous mine waste: the range of practical techniques used in Western Europe. In: Minerals, Metals and the Environment. Institute of Mining and Metallurgy, London. Pp. 481.

Brady, N.C. and Weil, R.R. 2008.The Nature and Properties of Soils, Fourteenth edition. Prentice Hall Publishing, Columbus Ohio, USA. Pp. 881.

Brink, V.C. 1980. Use of native species in mine land reclamation. Proceedings of the 4th Annual British Columbia Mine Reclamation Symposium in Vernon, B.C., 1980. The Technical and Research Committee on Reclamation. Pp. 231-241.

Bulmer, C. 2000. Reclamation of forest soils with excavation tillage and organic amendments. Forest Ecology and Management. 133: 157-163.

Burger, J., Graves, D., Angel, P., Davis, V., Zipper, C. 2005. The Forestry Reclamation Approach. Forest Reclamation Advisory No. 2. The Appalachian Regional Reforestation Initiative (ARRI). Pp. 4.

Burger, J., Davis, V., Franklin, J., Zipper, C., Skousen, J., Barton, C., Angel, P. 2009. Tree-compatible ground covers for reforestation and erosion control. Forest Reclamation Advisory No. 2. The Appalachian Regional Reforestation Initiative (ARRI). Pp. 6.

Burger, J.A., Zipper, C.E., Skousen, J. 2010. Establishing Groundcover for Forested Postmining Land Uses. Virginia Cooperative Extension, Powell River Project, Reclamation Guideline for Surface-Mined Lands. Publication 460-124. Pp. 8.

Burykin, A.M. 1985. Soil formation rates in man-made landscapes as related to recultivation. Pochvovedeniye. 2: 81-93.

Cabral, F., Vasconcelos, E., Goss, M.J., Cordovil, C.M.d.S. 1998. The value, use, and environmental impacts of pulp-mill sludge additions to forest and agricultural lands in Europe. Environmental Reviews. 6: 55-64.

Calace, N., Petronio, B.M., Pietrantonio, M., Pietoletti, M. 2011. Metal Polluted Soil Remediation Using Inexpensive Amendments. In: Soil Remediation. Editors: L. Aachen and P. Eichmann. Nova Science Publishers, Inc, New York. Pp. 281-296.

California Department of Conservation. 2007. Surface Mining and Reclamation Act and Associated Regulations. Office of Mine Reclamation. Pp. 87.

Camberto, J.J., Gagnon, B., Angers, D.A., Chantigny, M.H., Pan, W.L. 2006. Pulp and paper mill by-products as soil amendments and plant nutrient sources. Canadian Journal of Soil Science. 86: 641-653.

Canadian Boreal Initiative. 2005. The Boreal in the Balance: Securing the Future of Canada's Boreal Region. Canadian Boreal Initiative. Ottawa, ONT. Pp. 72.

Canadian Boreal Initiative. 2008. Mineral Exploration Conflicts in Canada's Boreal Forest. International Boreal Conservation Campaign and Canadian Boreal Initiative. Pp12.

Canadian Boreal Initiative. 2011. About Canada's Boreal: Fast Facts & Manitoba www.borealcanada.ca

Canadian Council of Ministers of the Environment (CCME). 2007. Canadian Soil Quality Guidelines for the Protection of Environmental and Human Health. Update 7.0. Pp. 6.

Canham, C.D., Berkowitz, A.R., Kelly, V.R., Lovett, G.M, Ollinger, S.V., Schnurr, J. 1996. Biomass allocation and multiple resource limitation in tree seedlings. Canadian Journal of Forest Research. 26: 1521-1530.

Canadian Centre for Mineral and Energy Technology (CANMET). 1999. Construction and instrumentation of a multi-layer cover. MEND report 2.22.4a. Pp. 267.

Carlson, M., Wells, J. Roberts, D. 2009. The Carbon the World Forgot: Conserving the Capacity of Canada's Climate Change. Boreal Songbird Initiative and Canadian Boreal Initiative, Seattle, WA, and Ottawa, ON. Pp 33.

Carlson, M., Chen, J., Elgie, S., Henschel, C., Montenegro, A., Roulet, N., Scott, N., Tarnoca, C., Wells, J. 2010. Maintaining the role of Canada's forests and peatlands in climate regulation. The forestry chronicle. 86: 434-443.

Carlsson, G. and Huss-Danell, K. 2003. Nitrogen fixation in perennial forage legumes in the field. Plant and Soil. 253: 353-372.

Carpenter, A.F. and Fernandez, I.J. 2000. Pulp sludge as a Component in Manufactured Topsoil. Journal of Environmental Quality. 29: 387-397.

Caruccio, F.T.,Hossner, L.R., Geidel, G. 1988. Pyritic materials: acid drainage, soil acidity and liming. In: Reclamation of Surface Mined Lands Vol. 1. Edited By: L.R. Hossner. Pp. 159-189.

Chalker, B.E. 1981. Simulating Light-Saturation Curves for Photosynthesis and Calcification by Reef-Building Coral. Marine Biology. 63: 135-141.

Chambers, J.C., Brown, R.W., Johnston, R.S. 1987. A comparison of soil and vegetation properties of seeded and naturally revegetated pyritic mine spoil and reference sites. Landscape and Urban Planning. 14: 507-519.

Chapin, S.F. III. 1980. The Mineral Nutrition of Wild Plants. Annual Review of Ecology and Systematics. 11: 233-260.

Chapman, H.D. 1965. Cation-exchange capacity. In: Methods of Soil Analysis, Part 2. Agronomy No. 9. Edited by: C.A. Black. Pp. 891-901.

Chenu, C. and Cosentino, D. 2011. Microbial Regulation of Soil Structure Dynamics. In: The Architecture and Biology of Soils: Life in Inner Space. Edited by: K Ritz and I Young. CAB International. Pp. 37-70.

Clymo, R.S. 1991. Peat Growth. In: Quaternary landscapes. Edited by: L.C. Shane and E.J. Cushing. University of Minnesota Press, Minneapolis, MN. Pp. 76-112.

Conyers, M.K. and Moody, P.W. 2009. A conceptual framework for improving the P efficiency of organic farming without inputs of soluble P fertilizer. Crop and Pasture Science. 60: 100-104.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neil, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M. 1997. The value of the world's ecosystem services and natural capitol. Nature. 387: 253-260.

Cranstone, D. A. 2002. A History of Mining and Mineral Exploration in Canada and Outlook for the Future. Natural Resources Canada. Minister of Public Works and Government Services Canada. Pp 47.

Crocker, R.L. and Major, J. 1955. Soil Development in Relation to Vegetation and Surface Age at Glacier Bay, Alaska. The Journal of Ecology. 43: 427-448.

Croser, C., Renault, S., Franklin, J., Zwiazek, J. 2001. The effect of salinity on the emergence and seedling growth of *Picea mariana, Picea glauca*, and *Pinus banksiana*. Environmental Pollution. 115: 9-16.

Dai, Y., Shen, Z., Wang, L., Hannaway, D., Lu, Hongfei. 2009. Effects of shade treatments on the photosynthetic capacity, chlorophyll fluorescence, and chlorophyll content of *Tetrastigma hemsleyanum* Diels et Gilg. Environmental and Experimental Botany. 65: 177-182.

Davidson, R.L. 1968. Effects of Soil Nutrients and Moisture on Root/Shoot Ratios in *Lolium perenne* L. and *Trifolium repens* L. Annals of Botany. 33: 571-577.

Delwiche, C.C., Johnson, C.M., Reisenauer, H.M. 1961. Influence of Cobalt on Nitrogen Fixation by *Medicago*. Plant Physiology. 36: 74-78.

Dollhopf, D.J. and Postle, R.C. 1988. Physical parameters that influence successful minesoil reclamation. In: Reclamation of Surface-Mined Lands, Vol. 1. Edited by: L.R. Hossner. CRC Press. Pp 81-105.

Dudal, R. 2004. The sixth factor in soil formation. Presented at the International Conference on Soil Classification, August 2004. Pp. 13.

Dudka, S. and Adriano, D.C. 1997. Environmental Impacts of Metal Ore Mining and Processing: A Review. Journal of Environmental Quality. 26: 590-602.

El-Kherbawy, M., Angle, J.S., Heggo, A., Chaney, R.L. 1989. Soil pH, rhizobia, and vesicular-arbuscular mycorrhizae inoculation effects on growth and heavy metal uptake of alfalfa (*Medicago sativa* L.). Biological Fertility of Soils. 8: 61-65.

Eugster, W., Rouse, W.R.,Pielke, R.A., McFadden, J.P., Baldocchi, D.D., Kittle, T.G.F., Chapin, F.S., Liston, G.E., Vidale, P.L., Vaganov, E., Chambers, S. 2000. Land-atmosphere energy exchange in Arctic tundra and boreal forest: available data and feedbacks to climate. Global Change Biology. 6: 84-115.

Evans, J.R. 1989. Photosynthesis and nitrogen relationships in leaves of C3 plants. Oecologia. 78: 9-19.

Fageria, N.K.,Baligar, V.C., Jones, C.A. 2011. Growth and Mineral Nutrition of Field Crops, Third Edition. CRC Press. New York, NY. Pp. 560.

Feagley, S.E., Valdez, M.S., Hudnall, W.H. 1994. Papermil Sludge, Phosphorous, Potassium, and Lime Effect on Clover Grown on a Mine Soil. Journal of Environmental Quality. 23: 759-765.

Fleming, G.A. and Murphy, W.E. 1968. The uptake of some major and trace elements by grasses as affected by season and stage of maturity. Presented at the Winter Meeting of the British Grasslands Society in London, March 1968. 174-187.

Foley, B.J. and Cooperband, L.R. 2002. Paper Mill Residuals and Compost Effects on Soil Carbon and Physical Properties. Journal of Environmental Quality. 31: 2086-2095.

Foy, C.D., Chaney, R.I., White, M.C. 1978. The physiology of metal toxicity in plants. Annual Review of Plant Physiology. 29: 511-566.

Franz, R. 1992. Colonization and early succession on anthropogenic soils. Journal of Vegetation Science. 3: 201-208.

Gallardo, F., Cea, M., Tortella, G.R., Diez, M.C. 2010. Effect of pulp mill sludge on soil characteristics, microbial community, and vegetal production of *Lolium Perenne*. Journal of Environmental Management. 95: S193-S198.

Gagnon, J.Y. and Crowder, A.A. 1984. Successional plant assemblages at the Beattie tailings, Duparquet, Quebec. Ontario Ecology and Ethology Colloquium. Pp. 1.

Gangopadhyay, G. and Basu, S. 2000. Proline as osmotic stress marker in *in vitro* system. In: Advances in Plant Physiology – vol III Edited by: A. Hemantaranjan. Scientific Publishers, Jodhpur, India. Pp. 283 – 304.

Gelbard, J.L. and Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology. 17: 420-432.

Goh., K.M. 2007. Effects of Multiple Reference Plants, Season, and Irrigation on Biological Nitrogen Fixation by Pasture Legumes using the Isotope Dilution Method. Communications in Soil Science and Plant Analysis. 38: 1841-1860.

Goins, G.D. and Russelle, M.P. 1996. Fine root demography in alfalfa (*Medicago sativa* L.). Plant and Soil. 185: 281-291.

Gonzalez, G., Gould, W.A., Hudak, A.T., Nettleton-Hollingsworth, T. 2008. Decay of Aspen (*Populous tremuloides* Michx.) Wood in Moist and Dry Boreal, Temperate and Tropical Forest Fragments. Royal Swedish Academy of Sciences. 588-599.

Government of Alberta. September 2011. Understanding the Oil Sands. www.oilsands.alberta.ca

Government of Manitoba. 2008. Manitoba Mining: Rich in Opportunities. Pp. 17.

Gray, N.F. 1997. Environmental impact and remediation of acid mine drainage: a management problem. Environmental Geology. 30: 62-72.

Green, A.J. 1978. Physical Analyses. In: Manual on Soil Sampling and Methods of Analysis. Editor: J.A. McKeague. Canadian Society of Soil Science. Pp 4-36.

Green, S. and Renault, R. 2008. Influence of papermill sludge on growth of *Medicago sativa, Festuca rubra* and *Agropyron trachycaulum* in gold mine tailings: A greenhouse study. Environmental Pollution. 151: 524-531.

Grewling, T., Peech, M. 1960. Chemical soil tests. Cornell University Agricultural Experimentation Station, Bulletin 960. Pp. 34-36.

Groninger, J., Skousen, J., Angel, P., Barton, C., Burger, J., Zipper, C. 2007. Mine reclamation practices to enhance forest development through natural succession. The Appalachian Regional Reforestation Initiative (ARRI). Forest Reclamation Advisory No. 5. Pp. 5.

Güler, S. and Büyük, G. 2007. Relationships among Chlorophyll-Meter Reading Value, Leaf N and Yield of Cucumber and Tomatoes. Proceedings of the 3rd Balkan Symposium on Vegetablers and Potatoes. Acta Horticultura. 729 – 735.

Hardy BBT Limited. 1989. Manual of Plant Species Suitability for Reclamation in Alberta. Alberta Land Conservation and Reclamation Council. Pp. 444.

Hare, P.D. and Cress, W.A. 1997. Metabolic implications of stress-induced proline accumulation in plants. Plant Growth Regulation. 21: 79-102.

Hart, R., MiningWatch Canada, Hoogeveen, D. 2012. Introduction to the Legal Framework for Mining in Canada. MiningWatch Canada. Pp. 13.

Hilson, G. and Murck, B. 2000. Sustainable development in the mining industry: clarifying corporate perspective. Resource Policy. 26: 227-238.

Hogan, C.M. and Tremblay, G.A. 2008. NOAMI and partners – a six-year retrospective. NAOMI and MEND Secretartiat. Pp. 9.

Hopkins, G.W. 1999. Introduction to plant physiology, 2nd Edition. John Wiley & Sons, Inc. Pp. 512.

Huag, M.D. and Pauls, G. 2001. A Review of non-traditional dry covers. Mining Environmental Neutral Drainage (MEND) 2.21.3b. Pp. 142

Ibrahim, S.M. and Goh, T.B. 2004. Changes in Macroaggregation and Associated Characteristics in Mine Tailings Amended with Humic Substances. Communications in Soil Science and Plant Analysis. 35: 1905-1922.

Innis, H.A. 1948. Mining Industry. In: The Encyclopedia of Canada. Edited by: W. S. Wallace. Toronto University Association of Canada.4: 297-305.

Insam, H. and Domsch, K.H. 1988. Relationship Between Soil Organic Carbon and Microbial Biomass on Chronosequences of Reclaimed Sites. Microbial Ecology. 15: 177-188.

Irigoyen, J.J., Emerich, D.W., Sànchez-Díaz. 1992. Water stress induced changes in concentrations of proline and total soluble sugars in nodulated alfalfa (*Medicago sativa*) plants. Physiologia Plantarum. 84: 55-60.

Jansen, I.J. and Melsted, S.W. 1988. Land shaping and soil construction. In: Reclamation of Surface-Mined Lands, Vol. 1. Edited by: Hossner, L.R. CRC Press, United States. Pp 125-137.

Jenny, H. 1958. Role of the plant factor in the pedogenic functions. Ecology. 39: 5-16.

Jennings, S.R., Neumann, D.R., Blicker, P.S. 2008. Acid Mine Drainage and Effects on Fish Health and Ecology: A Review. Reclamation Research Group Publication, Bozeman, MT. Pp 29.

Jha, A.K. and Singh, J.S. 1991. Spoil characteristics and vegetation development of an age series of mine spoils in a dry tropical environment. Vegetation. 97: 63-76.

Johnson, M.S. and Bradshaw, A.D. 1977. Prevention of heavy metal pollution from mine wastes by vegetative stabilization. Transactions of the Institute of Mining and Metallurgy. A86: 47- 55.

Johnson, M.S., McNeilly, T., Putwain, P.D. 1977. Revegetation of metalliferous mine spoil contaminated by lead and zinc. Environmental Pollution. 12: 261- 277.

Johnson, M.S., Cooke, J.A., Stevenson, S.K.W. 1994. Revegetation of Metalliferous wastes and land after metal mining. In: Mining and its Environmental Impact. Edited By: R.E. Hester and R.M. Harrison. Royal Society of Chemistry (Great Britain). Pp. 31-47.

Johnson, E.A. and Miyanishi, K. 2008. Creating New Landscapes and Ecosystems: The Alberta Oil Sands. New York Academy of Sciences. 1134: 120-145.

Kalra, Y.P. and Maynard, D.G. 1991. Methods Manual for Soil and Plant Analysis. Forestry Canada. Northwest Region. Northern Forestry Center. Edmonton, AB. Pp. 116.

Karst, A. 2010. Conservation Value of the North American Boreal Forest from an Ethnobotanical Perspective. Canadian Boreal Initiative, David Suzuki Foundation and Boreal Songbird Initiative; Ottawa, ON; Vancouver, BC; Seattle, WA. Pp. 32.

Kemper, W.D., Rosenau, R.C. 1986. Aggregate stability and size distribution. In: Methods of Soil Analysis, Part 1. Physical and Mineralogical methods. 2nd Edition. Edited by: A. Klute. American Society of Agronomy. Madison, WI, USA. Pp. 425-442.

Kozlowski, T.T. 1999. Soil compaction and growth of woody plants. Scandinavian Journal of Forest Research. 14: 596-619

Kramer, P.A., Zabowski, D., Scherer, G., Everett, R.L. 2000. Native Plant Restoration of Copper Mine Tailings: 1. Substrate Effect on Growth and Nutritional Status in a Greenhouse Study. Journal of Environmental Quality. 29: 1762-1769.

Lambert, C.M. 2001. Preliminary Study of the Mineralogy and Geochemistry of the Gunner Mine Tailings Pile, Nopiming Park, Manitoba, Canada.Bachelor of Science Thesis, University of Manitoba. Pp. 68.

Larney, F.J. and Angers, D.A. 2012. The role of organic amendments in soil reclamation: A review. Canadian Journal of Soil Science. 92: 19-38.

Lawlor, D.W. 1995. Photosynthesis, productivity and environment. Journal of Experimental Botany. 46: 1449-1461.

Lea, E.C. 1982. Use of vegetation survey data in reclamation. Proceedings of the 6th Annual British Columbia Mine Reclamation Symposium in Vernon, B.C., 1982. The Technical and Research Committee on Reclamation. Pp. 253-273.

Leisman, G.A. 1957. A vegetation and soil chronosequence on the Mesabi iron range spoil banks, Minnesota. Ecological Monographs. 27: 221-245.

Li, H., Jiang, D., Wollenweber, B., Dai, T., Cao, W. 2010. Effects of shading on morphology, physiology and grain yield of winter wheat. European Journal of Agronomy. 33: 267-275.

Livens, F.R. 1991. Chemical Reactions of Metals with Humic Materials. Environmental Pollution. 70: 183-208.

Lottermosser, B. 2003. Mine Wastes: Characterization, Treatment and Environmental Impacts, 2nd Edition. Springer publishing. Pp. 277.

Lyle, E.S. 1987. Primary factors affecting seed germination, plant establishment and growth. In: Surface mine reclamation manual. Elsevier Press, New York, NY. Pp. 3 -30.

MacKinney, G. 1941. Absorption of light by chlorophyll. Journal of Biological Chemistry. 315-322.

Manitoba Innovation, Energy and Mines. 2011. Manitoba's Mineral Industry: A Rich History. minesinfo@gov.mb.ca

Markham, J.H., Renault, S., Young, I., Halwas, S., Kunkel, S. 2008. Natural revegetation of Gunnar minesite, Manitoba. Report on Activities 2008, Manitoba Science, Technology, Energy and Mines, Manitoba Geological Survey. Pp. 139-143.

Marsh, A.S., Arnone, J.A. III, Bormann, B.T., Gordon, J.C. 2000. The role of *Equisetum* in nutrient cycling in an Alaskan shrub wetland. Journal of Ecology. 88: 999-1011.

Martínez-Ruiz, C., Fernández-Santos, B., Gómez-Gutiérrez, J.M. 2001. Effects of substrate coarseness and exposure on plant succession in uranium-mining wastes. Plant Ecology. 155: 79-89.

Mendez, M.O. and Mainer, R.M. 2008. Phytostabilization of Mine Tailings in Arid and Semiarid Environments – An Emerging Remediation Technology. Environmental Health Perspectives. 116: 278-283.

Meriño-Gergichevich, C., Alberdi, M., Ivanov, A.G., Reyes-Dìaz. 2010. Al^{3-}-Ca^{2+} Interactions in plants growing in acid soils: Al-phytotoxicity response to calcareous amendments. Journal of Soil Science and Plant Nutrition. 10: 217-243.

Mining Environmental Neutral Drainage (MEND). 2001. MEND Manual Volume 1 – Summary. MEND 5.4.2a. Pp. 115.

Mines Branch. 2009. A History of Mine Site Rehabilitation in Manitoba. Manitoba, Canada. Mineral Resources Division. http://www.manitoba.ca/iem/mrd/mines/oa_rehab_history.html

The Mining Association of Canada. 2011. A report on the state of the Canadian mining industry: Facts and Figures 2010. Pp 112.

Minister of Supply and Services Canada. 1996. Mining . In: The State of Canada's Environment. Government of Canada, Ottawa. Pp. 798.

Misra, V., Pandey, S.D., Viswanathan, P.N. 1996. Environmental Significance of Humic Acid in the Sequestration of Metals. Chemistry and Ecology.13: 103-112.

Mulvaney, R.L. 1996. Nitrogen-Inorganic forms. In: Methods of Soil Analysis. Part 3. Chemical Methods. Edited by D.L. Sparks. Soil Science Society of America, Madison, WI. Pp. 1123 - 1184.

Mummy, D.L., Stahl, P.D., Buyer, J.S. 2002. Microbial biomarkers as an indicator of ecosystem recovery following surface mine reclamation. Applied Soil Ecology. 21: 251-259.

Myers, N. 1996. Environmental services of biodiversity. Ecology. 93: 2764-2769.

Myrold, D.D., Ruess, R.W., Klug, M.J. 1999. Dinitrogen fixation. In: Standard soil methods for long term ecological work. Edited by: G.P. Robertson, D.C. Coleman, C.S. Bledsoe, and P. Sollins. Oxford University Press, Oxford. Pp. 241–257.

Nakata, C.K. 2007. Petroleum coke and plants: Impact on growth and physiology. Masters of Science Thesis. University of Manitoba, Winnipeg, Manitoba. Pp. 152.

National Orphaned/Abandonded Mines Initiative (NOAMI). 2004. Guidelines for legislative review. Submitted to the 61st Annual Mines Ministers' Conference, Iqaluit, Nunavut July 20, 2004. Pp. 13.

Neel, C., Bril, H., Courtin-Nomade, A., Dutreuil, J-P. 2003. Factors affecting natural development of soil on 35-year-old sulphide-rich mine tailings. Geoderma. 111: 1-20.

Nesheim, L. and Oyen, J. 1994. Nitrogen-fixation by Red-clover (*Trifolium-pratense* L) Grown in Mixtures with Timothy (*Phleum-pratense* L) at Different Levels of Nitrogen-Fertilization. Acta Agriculturae Scandinavica Section B- Soil and Plant Science. 44: 28-34.

Ni, Y., Guo, Y.J., Guo, Y.J., Han, L., Tang, H., Conyers, M. 2012. Leaf cuticular waxes and physiological parameters in alfalfa leaves as influenced by drought. Photosynthetica. 50: 458-466.

Niemela, J. 1999. Management in relation to disturbance in the boreal forest. Forest Ecology and Management. 115: 127-134.

Northwatch, MiningWatch Canada and Associates. 2008. The Boreal Below: Mining Issues and Activities in Canada's Boreal Forest. Pp. 205.

Nowak, R.S. 1993. Assessing physiological requirements of plant species as a prerequisite for their use in mine reclamation. Presented at the 1993 National Meeting of the American Society for Surface Mining and Reclamation, Spokane, Washington, May 1993. 23-29.

Ojeniyi, S.O. and Dexter, A.R. 1979. Soil Factors Affecting the Macro-structures Produced by Tillage. Transactions of the American Society of Agricultural Engineers. 339-344.

Okonski, A., Levin, S., Tish, B., Bekett, P., McCreath, D. 2003. Changes in Chemical and Biological Properties of Paper Sludge Used for Revegetation of Copper Tailings. Proceedings of the Mining and the Environment III Conference, Canadian Land Reclamation Association. Sudbury, Ontario. Pp. 6.

Olivera, M., Tejera, N., Oceana, A., Lluch, C. 2004. Growth, nitrogen fixation and ammonium assimilation in common bean (*Phaseolus vulgaris*): effect of phosphorous. Physiologia Plantarum. 121: 498-505.

Ontario Forestry Research Institute. 2011. Queen's Printer for Ontario, 2012. http://www.mnr.gov.on.ca/en/Business/OFRI/index.html

Osman, N. and Barakbah, S.S. 2011. The effect of plant succession on slope stability. Ecological Engineering. 37: 139-147.

Pang, J., Chan, G.S.Y., Zhang, J., Liang, J., Wong, M.H. 2003. Physiological aspects of vetiver grass for rehabilitation in abandoned metalliferous mine wastes. Chemosphere. 52: 1559-1570.

Peech, M. 1965. Hydrogen-ion activity. In: Methods of Soil Analysis Part 2; Editor: C.A. Black. Pp.914-926.

Pentreath, R.J. 1994. The Discharge of Waters from Active and Abandoned Mines. In: Mining and its Environmental Impact. Edited By: R.E. Hester and R.M. Harrison. Royal Society of Chemistry (Great Britain). Pp 121-131.

Peralta-Videa, J.R., de la Rosa, G., Gonzalez, J.H., Gardea-Torresdey, J.L. 2004. Effects of the growth stage on the heavy metal tolerance of alfalfa plants. Advances in Environmental Research. 8: 679-685.

Persson, J., Högberg, P., Ekblad, A., Höberg, M.N., Nordgren, A., Näsholm, T. 2003. Nitrogen acquisition from inorganic and organic sources by boreal forest plants in the field. Oecologia. 137: 252-257.

Pichel, J.R., Dick, W.A., Sutton, P. 1994. Comparison of Amendments and Management Practices for Long-Term Reclamation of Abandoned Mine Lands. Journal of Environmental Quality. 23: 766-772.

Pierret, A., Hartmann, C., Maeght, J.L., Pages, L. 2011. Biotic Regulations: Plants. In: The Architecture and Biology of Soils: Life in Inner Space Edited by: K. Ritz and I. Young. CAB International. Pp. 86 – 103.

Pierzynski, G.M., Schnoor, J.L., Banks, M.K., Tracy, J.C., Licht, L.A., Erickson, L.E. 1994. Vegetative Remediation at Superfund Sites. In: Mining and its Environmental Impact. Edited By: R.E. Hester and R.M. Harrison. Royal Society of Chemistry (Great Britain). Pp. 49-69.

Polster, D.F. 1991. Natural Vegetation Succession and Sustainable Reclamation. Proceedings of the 15th Annual British Columbia Mine Reclamation Symposium in Kamloops, BC, 1991. The Technical and Research Committee on Reclamation. 59-69.

Polster, D.F. 2009. Natural Processes: The Application of Natural Systems for the Reclamation of Drastically Disturbed Sites. British Columbia Mine Reclamation Symposium. Pp 8.

Prach, K. and Hobbs, R.J. 2008. Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restoration Ecology. 16: 363-366.

Price, W.A. 2003. Challenges posed by metal leaching and acid rock drainage, and approaches used to address them. In: Environmental Aspects of Mine Wastes. Edited by: J.L. Jambor, D.W. Blowes and A.I.M. Ritchie. Mineralogical Association of Canada. Pp. 1-10.

Price, G.W. and Voroney, R.P. 2007. Papermill Biosolids Effect on Soil Physical and Chemical Properties. Journal of Environmental Quality. 36: 1704-1714.

Rashid, M.T., Barry, D., Goss, M. 2006. Paper mill biosolids application to agricultural lands: benefits and environmental concerns with special reference to situation in Canada. Soil & Environment. 25: 85-98.

Raven, P.H., Evert, R.F., Eichhorn, S.E. 2005. Biology of Plants, seventh edition. W.H Freeman and Company. Pp. 686.

Reeves, F.B., Wagner, D., Moorman, T., Kiel, J. 1979. The Role of Endomycorrhizae in Revegetation Practices in the Semi-Arid West. I. A Comparison of Incidence of Mycorrhizae in Severely Disturbed Vs. Natural Environments. American Journal of Botany. 66: 6-13.

Reid, N.B. and Naeth, M.A. 2005a. Establishment of a Vegetative Cover on Tundra Kimberlite Mine Tailings: 1. A Greenhouse Study. Restoration Ecology. 13: 594-601.

Reid, N.B. and Naeth, M.A. 2005b. Establishment of a Vegetative Cover on Tundra Kimberlite Mine Tailings: 2. A Field Study. Restoration Ecology. 13: 602-608.

Reis, A.R., Favarin, J.L., Malavolta, E., Júnior, J.L., Moraes, F. 2009.Photosynthesis, Chlorophylla, and SPAD Readings in Coffee Leaves in Relation to Nitrogen Supply. Communications in Soil Science and Plant Analysis. 40: 1512-1528.

Renault, S., Nakata, C., Sabra, A., Davis, L., Overton, D. 2006. Revegetation of tailings at the Gunnar minesite, Manitoba: preliminary observations on plant growth in tailings amended with paper-mill sludge. Manitoba, Report of Activities 2006 Manitoba Science, Technology, Science and Mines, Manitoba Geological Survey. Pp 231-233.

Rensburg, L.V. and Morgenthal, T. 2004. The effect of woodchip waste on vegetation establishment during platinum tailings rehabilitation. South African Journal of Science.100: 294-300.

Rescan Environmental Services, LTD. 1990. Geochemical assessment of sub-aqueous tailings disposal in Anderson Lake, Snow Lake area, Manitoba. Mining Environmental Neutral Drainage (MEND) 2.11.1b-b. Pp. 179.

Ripley, E.A., Redmann, R.E., Crowder, A.A. 1996. Environmental effects of mining. St. Lucie Press. Florida, USA. Pp. 356.

Roberts, R.D., Marrs, R.H., Skeffington, R.A., Bradshaw, A.D. 1981. Ecosystem development on naturally colonized china clay wastes 1. Vegetation changes and overall accumulation of organic matter and nutrients. Journal of Ecology. 69: 153-161.

Roberts, J.A., Daniels, W.L., Bell, J.C., Burger, J.A. 1988. Early Stages of Mine Soil Genesis in a Southwest Virginia Spoil Lithosequence. Soil Science Society of America Journal. 52: 716-723.

Rowe, J.S. and Halliday, W.E.D. 1972. Forest regions of Canada. Information Canada, Ottawa. Pp. 172.

Rowe, J.S. and Scotter, G.W. 1973. Fire in the boreal forest. Quaternary Research. 3: 444-464.

Rozema, J., Rozema-Dijst, E., Freijsen, A.H.J., Huber, J.J.L. 1978. Population Differentiation within *Festuca rubra* L. with Regard to Soil Salinity and Soil Water. Oecologia (Berl.). 34: 329-341.

Runesson, U. 2011. borealforest.org. Faculty of Natural Resource Management, Lakehead University. www.borealforest.org.

Russelle, M.P., Allan, D.L., Gourley, C.J.P. 1994. Direct assessment of symbiotically fixed nitrogen in the rhizosphere of alfalfa. Plant and Soil. 159: 233-243.

Rykaart, M and Hockley, M. 2009. Mine Waste Covers in Cold Regions. Mining Environmental Neutral Drainage (MEND)1.61.5a. SRK Consulting (Canada) Inc. Vancouver, B.C. Pp. 119.

Schafer, W.M., Nielsen, G.A., Nettleton, W.D. 1980. Minesoil Genesis and Morphology in a Spoil Chronosequence in Montana. Soil Science Society of America Journal. 44: 802-807.

Schofield, R.K. and Taylor, A.W. 1955. The measurement of soil pH. Proceedings of the Soil Society of America. 19: 164-167.

Schuman, G.E. and Belden, S.E. 1991. Decomposition of Wood-Residue Amendments in Revegetated Bentonite Mine Spoils. Soil Science Society of America Journal. 55: 76-80.

SENES Consultants Limited. 1996. Review of use of an elevated water table as a method to control and reduce acidic drainage from tailings. Mining Environmental Neutral Drainage (MEND) 2.17.1. Pp. 165.

Sesták, Z., Catský, J., Jarvis, P.G. 1971. Determination of chlorophylls a and b. In: Plant photosynthetic production. Manual of Methods. Edited by: Z. Sesták, J. Catský, and P.G. Jarvis. Pp. 818.

Shaw, P.J.A. 1992. A preliminary study of successional changes in vegetation and soil development on unamended fly ash (PFA) in southern England. Journal of Applied Ecology. 29: 728-736.

Shrestha, R. K. and Lal R. 2006. Ecosystem carbon budgeting and soil carbon sequestration in reclaimed soil. Environment International 6: 781–796.

Shu, W.S., Ye, Z.H., Zhang, Z.Q. Lan, C.Y., Wong, M.H. 2005. Natural Colonization of Plants on Five Lead/Zinc Mine Tailings in Southern China. Restoration Ecology. 13: 49-60.

Skousen, J. and Zipper, C. 1996. Revegetaion Species and Practices. Powel River Project.Reclamation Guidelines for Surface Mined Land in Southwest Virginia. Publication 460-122. Pp. 18.

Slivitzky, M.S.C. 1996. The Manitoba Model Forest: an assessment of mineralogical development, with recommendations for ecosystem based management. Masters of Resource Management Thesis. University of Manitoba, Winnipeg, Manitoba. Pp 158.

Sly, P.G. 1996. Review of MEND studies on the subaqueous disposal of tailings (1993-95). Mining Environmental Neutral Drainage (MEND) 2.11.1e. Pp. 39.

Smith, L and Beckie, R. 2003. Hydrologic and geochemical transportation processes in mine waste rock. In: Environmental Aspects of Mine Wastes. Edited by: J.L. Jambor, D.W. Blowes and A.I.M. Ritchie. Mineralogical Association of Canada. Pp 51-72.

Smith, F.C., Johnson, A.H., Dranoff, M., Wibiralske, A. 1997.Biomass and Nutrient Accumulation During Natural Afforestation of Iron-Smelting Slag. Restoration Ecology. 5: 56-65.

Sofo, A., Dichio, B., Xiloyannis, C., Masia, A. 2004. Lipoxygenase activity and proline accumulation in leaves and roots of olive trees in response to drought stress. Physiologia Plantarum. 121: 58-65.

Standing Committee on Natural Resources. 1996. 2nd Report of the Standing Committee on Natural Resources. Ottawa, Canada. Pp. 21.

Stark, J.M. and Redente, E.F. 1985. Soil-plant Diversity Relationships on a Disturbed Site in Northwestern Colorado. Soil Science Society of America Journal. 49: 1028-1034.

Strohmayer, P. 1999. Soil stockpiling for reclamation and restoration activities after mining and construction. Restoration and Reclamation Review. 4: 1-6.

Swift, R.S. 2001. Sequestration of Carbon by soil. Soil Science. 166: 858-871.

Szczerski, C., Naguit, C., Markham, J., Goh, T.B., Renault, S. 2013. Short- and Long-Term Effects of Modified Humic Substances on Soil Evolution and Plant Growth in Gold Mine Tailings. Water, Air, and Soil Pollution. 224: 1471-1485.

Tahboub, M.B., Lindemann, W.C., Murray, L. 2008. Chemical and Physical Properties of Soil Amended with Pecan Wood Chips. Horticultural Science. 43: 891-896.

Tate, R.L. III. 2000. Soil Microbiology, Second Edition. John Wiley & Sons, Inc New York, NY. Pp. 508.

Tester, C.F. 1990. Organic Amendment Effects on Physical and Chemical Properties of a Sandy Soil. Soil Science Society of America Journal. 54: 827-831.

Tisdall, J.M. and Adem, H.H. 1986. Effect of water content of soil at tillage on size-distribution of aggregates and infiltration. Australian Journal of Experimental Agriculture. 26: 193-195.

Tisdall, J.M., and J.M. Oades. 1982. Organic matter and water-stable aggregates in soils. Journal of Soil Science. 33: 141–163.

Torbert, J.L., Tuladhar, A.R., Burger, J.A., Bell, J.C. 1988. Minesoil property effects on the height of ten-year old white pine. Journal of Environmental Quality. 17: 189-192.

Tordoff, G.M., Baker, A.J.M, Willis, A.J. 2000. Current approaches to the revegetation and reclamation of metalliferous mine wastes. Chemoshpere. 41: 219-220.

Tremblay, G.A. and Hogan, C.M. 2009. Caps and Covers for acidic drainage control – the Canadian Experience. Natural Resources Canada. Ottawa, Ont. Pp. 6.

USDA, NRC, 2012. The PLANTS Database (http://plants.usda.gov, 13 December 2012). National Plant Data Team, Greensboro, NC 27401-4901 USA.

U.S. Environmental Protection Agency, Office of Solid Waste. 1994. Technical Document: Acid Mine Drainage Prediction. PB94-201829 EPA530-R-94-036

Vagstad, N., Broch-Due, A., Lyngstag, I. 2001. Direct and residual effects of pulp and paper mill sludge on crop yield and soil mineral N. Soil Use and Management. 17: 173-178.

Vasconcelos, E., Cabral, F. 1992. Use and environmental implications of pulp-mill sludge as an organic fertilizer. Environmental Pollution. 80: 159-162.

Vinther, F.P. and Jensen, E.S. 2000. Estimating legume N_2 fixation in grass-clover mixtures of a grazed organic cropping system using two 15^N methods. Agriculture, Ecosystems and Environment. 78: 139-147.

Venanzi, G., Lupattelli, M., Poccheschi. N., Frenguelli, G. 1993. Phototsynthetic characteristics of *Helianthus annus* L. as affected by nitrogenous fertilizers. Photosynthetica. 28: 29-35.

Wali, M.K. 1999. Ecological sucession and the rehabilitation of disturbed terrestrial ecosytems. Plant and Soil. 213: 195-220.

Walker, D.J., Clemente, R., Bernal, M.P. 2004. Contrasting effects of manure and compost on soil pH, heavy metal availability and growth of *Chenopodium album* L. in a soil contaminated by pyritic mine waste. Chemosphere. 57: 215-224.

Walkley, A. and Black, I.A. 1934. An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromatic acid titration method. Soil Science. 37: 29-38.

Wallace, I. 1996. Restructuring in the Canadian Mining and Mineral- Processing Industries. In: Canada and the Global Economy: The Geography of structural and technological change. Edited by: J. N.H. Britton. McGill-Queen's University Press. Pp. 123-136.

Waterer, J.G. and Vessey, K. 1993. Effect of low static nitrate concentrations on mineral nitrogen uptake, nodulation, and nitrogen fixation in field pea. Journal of Plant Nutrition. 16: 1775-1789.

Watson, M. E. and Hoitink, A.J. 1985. From waste to resource: Long term effects of papermill sludge in stripmine reclamation. Ohio report on research and development in agriculture, home economics, and natural resources - Ohio Agricultural Research and Development Center. Pp. 3.

Wells, J., Roberts, D., Lee, P., Cheng, R., Darveau, M. 2010. A forest of Blue – Canada's Boeal Forest: The Worlds Waterkeeper. International Boreal Conservation Campaign, Seattle, WA. Pp. 74.

Wilson, G.W. 2003. The design and performance of mine waste cover systems. In: Environmental Aspects of Mine Wastes. Edited by: J.L. Jambor, D.W. Blowes and A.I.M. Ritchie. Mineralogical Association of Canada. Pp 377-384.

Winterhalder, K. 1993. The roles of colonization and succession in the reclamation of mine sites. Presented at the 1993 National Meeting of the American Society for Surface Mining and Reclamation, Spokane, Washington, May 1993. 30-53.

Winterhalder, K. 1995. Approaches to engendering biodiversity in tailings revegetation. Paper presented at the Sudbury 1995, Conference on Mining and the Environment, Sudbury, Ontario. Pp 1239-1248.

Wood, C.W., Tracy, P.W., Reeves, D.W., Edmisten, K.L. 1992. Determination of cotton nitrogen status with a handheld chlorophyll meter. Journal of Plant Nutrition. 15: 1435-1448.

Ye, Z.H.,Shu, W.S., Zhang, Z.Q., Lan, C.Y. Wong, M.H. 2002. Evaluation of major constraints to revegetation of lead/zinc mine tailings using bioassay techniques. Chemosphere. 47: 1103-1111.

Yoshikawa, K., Overduin, P.P., Harden, J.W. 2004. Moisture Content Measurements of Moss (*Sphagnum* spp.) using Commercial Sensors. Permafrost and Periglacial Processes. 15: 309-318.

Young, I., Naguit, C., Hawlas, S., Markham., J., Renault, S. 2012. Natural revegetation of a boreal gold mine tailings pond. Restoration Ecology. In Press.

Zabowski, D., Everett, R., Scherer, G. 1993. Nutrient availability and demand relative to successional status of acid mine revegetation species. National Meeting of the American Society for Surface Mining and Reclamation, Spokane, Washington. Pp. 54-63.

Appendix:

Appendix A. Meteorological data from Pinawa, Manitoba weather station for experimental period (2009 - 11) (data from National Climate Data and Information Archive, online).

Month	Max Temperature (°C)	Min Temperature (°C)	Precipitation (mm)
May 2009	22.1	-5.1	77.5
June 2009	29.0	-1.6	141.2
July 2009	25.5	5.6	131.8
August 2009	31.5	4.9	99.2
September 2009	28.9	-1.2	108.8
October 2009	15.1	15.1	20.9
November 2009 - May 2010	23.7	-35.3	100.4
Total			679.8
May 2010	28.1	-2.3	156.1
June 2010	28.4	3.8	128.8
July 2010	29.7	7.8	113.4
August 2010	31.5	5.4	186.8
September 2010	23.9	-0.8	75.5
October 2010	25.9	-4.7	65.1
November 2010- May 2011	15.8	-38.0	125.5
Total			851.2
May 2011	26.2	-5.3	45.8
June 2011	31.8	2.5	63.2
July 2011	35.1	7.6	15.6
August 2011	35.4	6.5	23.0
September 2011	30.7	-2.9	68.3
October 2011	27.9	-7.0	19.4
November 2011 - February 2012	13.1	-34.4	56.1
Total			291.4

Appendix B. Meteorological data from Bissett, Manitoba weather station for the experimental period (2009 - 11) (data from National Climate Data and Information Archive, online).

Month	Max Temperature (°C)	Min Temperature (°C)	Precipitation (mm)
May 2009	21.9	-4.8	54
June 2009	29.3	0.4	130
July 2009	25.7	7	114.8
August 2009	30.8	6.3	115
September 2009	28.9	0.5	41.8
October 2009	13.5	-3.9	75.5
November 2009 - May 2010	22.6	-33.7	72.5
Total			603.6
May 2010	28.1	-2.4	85.8
June 2010	27.2	5.5	109
July 2010	29.4	8.9	67.4
August 2010	31.8	5	101.8
September 2010	24.5	0.4	72.5
October 2010	25	-4.8	53.3
November 2010- May 2011	20.1	-36.8	224
Total			713.8
May 2011	25.6	-5.7	55.8
June 2011	30.8	3.3	64.2
July 2011	34.9	8.5	39
August 2011	36.3	8.2	12.8
September 2011	31.1	-0.8	67.7
October 2011	26.7	-2.5	31
November 2011 - March 2012	22.7	-33.6	65.6
Total			336.1

Appendix C

1. Organic carbon increase

Tailings bulk density (BD): 1.40 g cm^{-3} = 1400 kg m^{-3}

Depth of incorporation (d): 15 – 20 cm = 0.15 – 0.20 m

Papermill sludge application rate: 3.70 kg m^{-2} (OM)

Wood chips application rate: 3.50 kg m^{-2} (OM)

Organic matter (OM) = 1.724 * Organic carbon (OC)

% Increase in OC = {[(BD * d + OC) / (BD * d)] – 1} * 100

Sample calculation:

% Increase in Papermill sludge treatment at 15 cm depth of incorporation:

= {[(1400 kg m^{-3} * 0.15 m + 2.15 kg OC m^{-2}) / (1400 kg m^{-3} * 0.15 m)] – 1} * 100

= 1.02 %

2. Sub-plot level plant nitrogen contents

Assuming an average plant tissue concentration of 12 g N kg^{-1} (1.2 %)

Sample calculation:

Subplot: Papermill sludge amended; seeded with grasses and *Medicago sativa*

Total biomass: 93.90 g m^{-2} (shoot) + 78.70 g m^{-3} (root) = 172.60 g m^{-2} to 0.20 m

N within plant biomass: 172.60 g m^{-2} * 12 g N kg^{-1} = 2.07 g N m^{-2} = 20.70 kg N ha^{-1}

Appendix D.

Figure D1. Plant growth on papermill sludge amended tailings seeded with all three cover species in early summer 2010 (June).

Figure D2. Plant growth on papermill sludge amended tailings seeded with just the grass species mix in early summer 2010 (June).

Figure D3. Plant growth on papermill sludge amended tailings seeded *Medicago sativa* in early summer 2010 (June).

Figure D4. Plant growth on papermill sludge amended tailings left unseeded in early summer 2010 (June)

Figure D5. Plant growth on wood chips amended tailings seeded with all three cover species in early summer 2010 (June)

Figure D6. Plant growth on wood chips amended tailings seeded with just the grass species mix in early summer 2010 (June).

Figure D7. Plant growth on wood chips amended tailings seeded *Medicago sativa* in early summer 2010 (June).

Figure D8. Plant growth on wood chips amended tailings left unseeded in early summer 2010 (June).

Figure D9. Plant growth on high fertilizer rate amended tailings seeded with all three cover species in early summer 2010 (June)

Figure D10. Plant growth on high fertilizer rate amended tailings seeded with just the grass species mix in early summer 2010 (June).

Figure D11. Plant growth on high fertilizer rate amended tailings seeded *Medicago sativa* in early summer 2010 (June).

Figure D12. Plant growth on high fertilizer rate amended tailings left unseeded in early summer 2010 (June).

Figure D13. Plant growth on low fertilizer rate amended tailings left unseeded in early summer 2010 (June).

Figure D14. Plant growth on low fertilizer rate amended tailings seeded *Medicago sativa* in early summer 2010 (June).

Figure D15. Plant growth on low fertilizer rate amended tailings seeded with all three cover species in early summer 2010 (June)

Figure D16. Plant growth on low fertilizer rate amended tailings seeded with just the grass species mix in early summer 2010 (June).

Figure D17. Moss establishment in a papermill sludge amended experimental plot in July 2010.

Figure D18. Wild blown tailings from unvegetated areas of the Gunnar mine tailings (June 2011).