SOIL INVERTEBRATES AS SUCCESS INDICATORS FOR LAND RECLAMATION MONITORING

by

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ABSTRACT

Current monitoring of anthropogenically disturbed lands that have been subject to land reclamation mainly focuses on soil physical and chemical properties and vegetation cover. With a global interest in resilience and biodiversity there is debate about whether such two factor monitoring is sufficient for assessing reclamation trajectories and success, and whether biotic indicators in addition to plants are needed. Soil invertebrate assemblages are often monitored as indicators of ecosystem health, function, and stability, making them potentially valuable for assessing reclamation success. However, their effective use in monitoring requires consideration of several factors, including the need for taxonomic expertise, selection of appropriate success indicator candidates, and practical constraints of measurement.

To streamline invertebrate assessment processes, a comprehensive measurement or index is needed. This measurement or index should encompass select soil invertebrate taxa that exhibit disproportionate responses to reclamation methods relative to reference conditions. It must remain consistent across seasons to prevent skewed results based on assessment timing. It should be sensitive to various reclamation techniques and soil materials, reflecting either positive or negative impacts of disturbance. Ideal indicators should be measurable in the field, easily quantifiable, and cost effective to obtain. They should facilitate efficient reclamation assessments without requiring extensive resources or specialized expertise.

I studied two young forest reclamation sites at a coal mine in western Alberta, Canada. The reclamation sites differed in soil reclamation methods (stockpiling versus direct placement), adjacent land uses, and relative distance to an undisturbed proximate forest reference site. My research objectives were to assess effectiveness of soil invertebrates in land reclamation monitoring, impacts of stockpiling and direct placement of salvaged soil on reclamation success, and changes in soil invertebrate assemblages over the growing season to develop reclamation monitoring strategies and recommendations for soil invertebrate indicator taxa. Response variables were vegetation community composition, soil properties, and soil invertebrate abundance and assemblages. Above ground, litter dwelling, and soil dwelling mesofauna and macrofauna were quantified in intensive monthly sampling over two years, using broad taxonomic identification.

Reclamation methods significantly altered soil invertebrate abundance and communities relative to the undisturbed reference. Direct placement of salvaged forest floor material and topsoil, rather than stockpiling, fostered vegetation and soil invertebrate communities more similar to reference sites. Some soil invertebrate taxa responded positively to disturbance, exemplified by higher abundance of ants and true bugs in reclamation areas. Conversely, taxa such as springtails and oribatid mites exhibited negative responses, displaying reduced abundance. While beetles and spiders demonstrated recovery post-disturbance in abundance in reclamation areas, species level examination showed variations in community composition, with reclaimed areas frequently dominated by a single species. Oribatid mites emerged as consistent and sensitive indicators, reliably distinguishing between reclamation sites and references across sampling periods. However, when considering site variability, soil invertebrates did not provide additional insights beyond existing vegetation and soil parameters.

While soil invertebrates might not serve as practical indicators for broad land reclamation monitoring, they hold importance for specific restoration focused areas or detailed monitoring efforts. The homogenization of soil invertebrate communities, coupled with the potential expansion of non-native plant species in reclamation sites, underscore the importance of maintaining soil biodiversity for long-term success and resilience. Thus, monitoring select soil invertebrate groups remains valuable where economically feasible and with time permitting, to ensure the health and sustainability of reclaimed ecosystems.

This research contributes to the knowledge of soil invertebrate assemblages in early forest reclamation sites, particularly in coal mining. It aids in the selection of soil invertebrate taxa for

intensified monitoring efforts and underscores the importance of addressing non-native plant species and singular species dominance in such reclamation sites. Results can be extrapolated to similar disturbances and other forested environments in Alberta and beyond.

Given that oribatid mites emerged as the most consistent and sensitive indicator of ecosystem conditions by abundance, further research is essential to fully leverage their potential in reclamation monitoring. Major next steps include evaluating the effects on oribatid mite assemblages at the species level to determine whether single species dominance and homogenization are occurring, similar to my observations of beetles and spiders, and to develop a photo based sorting algorithm capable of identifying and quantifying oribatid mites relative to other mesofauna.

PREFACE

My supervisor Dr. M. Anne Naeth, and supervisory committee members Dr. Diane Haughland and Dr. Heather Proctor provided guidance throughout this research. Gerad Hilchie assisted with identification of beetle species, and Kirra Kent assisted with identification of spider species.

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DEDICATION

Dedicated to the love of my life, Nicole Oak.

Thank you for being my roots,

For rebuilding my eroded profile,

For restoring my structure and health,

And for reclaiming my classification with new parent material.

I can't wait to marry you.

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1. BACKGROUND

Land degradation has resulted from anthropogenic disturbances associated with natural resource exploration and development, energy sector activities, recreation, agriculture, forestry, urban development, and other land uses; and from natural disturbances such as floods, fires, tornadoes, and volcanic eruptions. Large scale disturbances, such as mining, have caused intensive and extensive degradation; small scale disturbances such as pipelines and well sites, can be equally degrading due to cumulative effects. Land degradation is a growing global concern requiring large scale programs to reclaim and restore ecosystem structure, function, and composition after disturbance (Hobbs and Harris, 2001; Harris, 2003).

Timely and effective land reclamation requires objective and feasible metrics and methods to assess ecosystem status and reclamation trajectories (Harris, 2003). Monitoring is a feedback mechanism necessary to develop and meet environmental management standards and decisions regarding impact on species, populations, ecosystems, and landscapes (Smyth and Dearden, 1998). An important component of land reclamation monitoring is early indication that disturbed areas are on an appropriate trajectory of recovery from a disturbed area to an acceptable, established system (Harris, 2003). Criteria for judging reclamation success of disturbed lands include visually distinguishable above ground indicators, such as soil erosion and compaction and vegetation cover and diversity. However, these criteria fail to capture ecosystem function and account for critical ecosystem components, including microorganisms (Mummey et al., 2002).

The global focus on maintaining sustainable and resilient ecosystems, high in biodiversity, has shifted the interpretation of reclamation success from a green space to a complex and integrated system that will support diverse organisms at various trophic levels. There is concern that current indicators to determine reclamation success are not providing a complete picture. Vegetation metrics are commonly used to assess reclaimed ecosystem performance and requires measurements of selected indices such as biomass, cover, diversity, and spatial structure (Yang et al., 2022). Soil metrics have been mainly focused on physical and chemical properties, measurable in the field or the laboratory. Although vegetation parameters are relatively easy to assess, they may not reflect how well an ecosystem is progressing towards nutrient self-sufficiency as vegetation can be poor indicators of soil quality (Ernst, 2004).

2. LITERATURE REVIEW

2.1. Environmental Disturbance And Recovery

An ecological disturbance is a spatially and temporally discrete event resulting in sustained disruption of ecosystem structure and function (Pickett and White, 1985; Tilman, 1985). Natural and anthropogenic disturbances can alter population, community, and ecosystem structure, affecting resource availability on large spatial and temporal scales (Pickett and White, 1985). Anthropogenic habitat disturbance is a primary cause of reduced global biodiversity (Kamdem et al., 2012). Chapin et al. (2011) classified environmental disturbances into physical, biogenic, and anthropogenic categories. Physical disturbances include fires, hurricanes, floods, droughts, and ice storms; biogenic disturbances include impacts of insects, mammals, and pathogens; and anthropogenic disturbances include logging, agriculture, mining, chemical pollution, and species introductions. Human activities have altered the frequency and size of many natural disturbances, such as fires and floods, and produced new disturbance types such as logging, mining, and wars.

After disturbances, ecosystems undergo succession, a change in ecosystem composition, structure, and function; disturbance severity significantly influences vegetation recovery rate and trajectory (Chapin et al., 2011). For example, a disturbance that removes live or dead organic matter will be colonized by plants that gradually reduce light at the soil surface and change water and nutrient availability (Tilman, 1985). These events happen in sequence, and may retrograde, until the ecosystem has reached a state of equilibrium and resilience. Primary succession occurs when a severe disturbance leaves little or no biological legacy, creating a new surface for colonization; secondary succession occurs when a substantial biological legacy remains after disturbance (Prach and Walker, 2019). Primary succession follows severe disturbances such as glacial retreats, landslides, mining, and flooding, leaving little organic matter, soil, or plant propagules; secondary succession follows disturbances such as fires, hurricanes, logging, and plowing, which leave some soil organic matter and plant propagules; (Chapin et al., 2011).

2.2. Common Disturbances In Alberta

Major mining resources in Alberta include oil sands, coal, limestone, salt, shale, sandstone, and sand and gravel (Government of Alberta, 2020). Alberta has 85 % of coal deposits in Canada and 18 % of global coal deposits (Coal Association of Canada, 2018). Coal mines were first established in Alberta in Medicine Hat, Canmore, Lethbridge, and Edmonton between 1883 and 1891 (Blue, 1924). Great Canadian Oil Sands built the first major commercial oil sands production plant and began operations in 1968 (Chastko, 2004). Mining is an essential component to the economies of Alberta and Canada, although activities can have negative environmental effects and large environmental footprints.

Coal and oil sands surface mining involves vegetation clearing and removal of soil, peat, and overburden; extraction of coal or oil sands; and salvage and subsequent storage of topsoil and subsoil for future use in reclamation areas. Reclamation challenges arise from overburden material characteristics and mining methods; admixing of soil horizons and soil compaction have also been reported (Schori et al., 1989). Oil sands deposits deeper than 75 m require in situ operations, such as steam assisted gravity drainage, where steam, solvents, and other fluids are injected into the oil sands reservoir, allowing it to be pumped to the surface for recovery. These methods disturb less land per unit of production than surface mining, with different landscape impacts. Surface mining primarily creates polygonal features, while in situ mining is characterized by linear features, leading to increased landscape fragmentation (Jordaan et al., 2009).

Large scale linear features resulting from anthropogenic activities include transportation corridors, access roads, transmission lines, pipelines, survey lines, and seismic testing lines (King and Yetter, 2011). Linear developments have a small cumulative footprint and disproportionately large effects on surrounding ecological processes (Trombulak and Frissell, 2000; Whittington et al., 2005). Majority of linear developments occur in northern Alberta's boreal forest, led by forestry and energy sector exploration and resource extraction (Schneider et al., 2003).

Other anthropogenic disturbances include forestry and agriculture. Clearcutting can cause significant changes to precipitation, resulting in increased run-off and soil erosion (Johnson et al., 1991). Many agricultural practices, including tillage, fertilizer application, and use of herbicides degrade soil health, and have various effects on soil animals and microbiota.

2.3. Land Reclamation

2.3.1. Terminology

Numerous terms are associated with land reclamation, including restoration, rehabilitation, ecological restoration, remediation, and revegetation. Many terms are used interchangeably and vary within and among jurisdictions. Clearly distinguishing terms is important for context and to avoid confusion and miscommunication. Land reclamation is defined as the process of converting disturbed or damaged land to its former or other productive uses; it can be considered in a biophysical context, after natural resource exploration and development or when overuse or degradation occurs due to improper management (Naeth, 2016a). Land reclamation is an umbrella term and includes soil reclamation, revegetation, and contaminant remediation (Naeth, 2016a). Land reclamation is a method used to return disturbed land to a useful state (Alberta Environment, 1999). The Environmental Protection and Enhancement Act and the Conservation and Reclamation Regulation, outline that reclamation can include removing equipment, buildings, or other structures; decontaminating buildings, land, water, or other structure; and stabilizing, contouring, maintaining, conditioning, or reconstructing the surface of the land to a state of equivalent land capability (Alberta Energy Regulator, 2024).

Although restoration is often used interchangeably with land reclamation, these terms are not synonymous. Murray et al. (2021) illustrates the interchangeable use of restoration and reclamation in discussing how restoration approaches influence carbon exchange at in-situ oil sands wetland sites, which are actually undergoing reclamation, not restoration (Murray et al., 2021). Ecological restoration definitions include: the process of assisting in the recovery of an ecosystem that has been degraded, damaged, or destroyed (Society for Ecological Restoration, 2024); recreating entire communities of organisms, closely modeled on those occurring naturally (Jordan et al., 1987); and repairing disturbed ecosystems through human intervention, aiming to recreate, initiate, or accelerate ecosystem recovery (Vaughn et al., 2010). Restoration aims to return disturbed areas to a previous state by reintegrating abiotic and biotic site components, encompassing the reconstitution of composition, structure, and function of a community or ecosystem following disturbance (Naeth, 2016a). Ecological restoration is often used for conservation of threatened or endangered habitats or organisms.

Remediation is the removal, or reduction to acceptable levels, of a contaminant or unwanted element or compound in the soil, surface water, or ground water (Government of Alberta, 2024a; Naeth, 2016a). Remediation methods focus on improvement of a contaminated site to prevent, minimize, or mitigate damage to human health or the environment. Soil contaminants can be treated onsite, offsite through excavation, or disposed in a hazardous materials landfill; some soil remediation methods include encapsulating contaminated soil and capping with topsoil, soil vapor extraction, soil washing, electrochemical methods, and thermal treatments (Khan et al., 2021). Bioremediation employs biological entities such as bacteria or fungi to remove or detoxify pollutants, while phytoremediation uses plants for remediation to stabilize, degrade, volatize, or extract contaminants (Greipsson, 2011; Naeth, 2016a; Praveen and Nagalakshmi, 2022). Individual jurisdictions regulate acceptable levels of contaminant concentrations, after remediation, reclamation will likely be required to regain ecosystem functioning (Naeth, 2016a).

2.3.2. Land reclamation process

Prior to starting mining operations, companies must apply for approval to carry out activities that could cause environmental disturbance and they must provide security to guarantee reclamation. When mines or well sites have reached end of life, they will be decommissioned; any contaminated areas will be remediated and the reclamation plan approved in the mine plan can begin. The reclamation plan specifies end land use based on regulations and stakeholder input. End land use determines the appropriate land reclamation plan and criteria for reclamation certification. Alberta has reclamation criteria for native grasslands, cultivated lands, forested lands, and peatlands (Alberta Agriculture and Forestry, 2016).

Land reclamation practices for large disturbances, such as surface mining, include backfilling pits, often with overburden, subsoil, or tailings stored during mining operations. End pit lakes are common in surface coal mining reclamation. Topsoil, subsoil, and overburden are sequentially moved from one pit to the next as coal is mined, culminating in an end pit lake to reduce the need for soil materials once the final pit is mined (Bott et al., 2016b). Reclamation practices in plains coal mining includes removing vegetation one to two years before mining, stripping topsoil, and salvaging to the A horizon (approximately 20 cm), then salvaging other overburden material as subsoil (Fedkenheuer and Macyk, 2000; Navus Environmental Inc., 2012). Topsoil and subsoil can be used immediately in an active reclamation area (direct placement) or stockpiled. Subsoil is normally placed to a 1 m depth in an area of final elevation and landscape features; large rocks are removed, compaction alleviated using equipment such as a deep tillage cultivator, and smoothed to provide a level surface for uniform topsoil placement across the site, usually to a thickness of 18 to 20 cm (McQueen et al., 1991). Soils are reconstructed generally using various onsite materials, including stockpiled mineral (topsoil, subsoil, overburden) and organic soils. Offsite materials and amendments are often used to enhance reclaimed soil properties such as fertility, structure, and infiltration and retention of water and nutrients. Common soil amendments include chemical fertilizers, manure, peat, and compost.

Revegetation follows soil reclamation, planting seed mixes or seedlings, aligns with end land use plans. Revegetation guidelines include best practices for native plant selection, maintaining biodiversity, erosion control, seed sources, and weed control (Native Plant Working Group, 2000; Alberta Environment, 2003). When forest is the desired end land use in Alberta, revegetation practice is to seed with barley or native and non-native grasses to improve soil stability and prevent soil erosion, followed by tree seedling planting including white spruce (*Picea glauca*), trembling aspen (*Populus tremuloides*), and jack pine (*Pinus banksiana*) (Rowland et al., 2009).

Continued monitoring after vegetation establishment ensures an appropriate trajectory of ecosystem recovery and confirms that reclaimed areas are diverse and self-sustaining ecosystems that integrate into a surrounding landscape (Alberta Energy Regulator, 2019). Reclamation certificates are only issued when disturbed areas meet all criteria and pass landscape, soil, and vegetation assessments; certificates are granted when it can be demonstrated through monitoring, that sites meet end land use criteria and equivalent land capability (Alberta Energy Regulator, 2019). Environmental variables in monitoring to obtain forest reclamation certification include topography, geotechnical stability, surface and ground water quantity and quality, replaced soil quantity and quality, vegetation composition, wildlife use, and contaminant or waste presence (Alberta Energy Regulator, 2019).

In Alberta, large areas of surface coal mines have been reclaimed to agricultural land and to wildlife habitat, and over 40,000 well sites have been reclaimed to agricultural land (Alberta Environment, 1999). The first oil sands reclamation certificate was issued to Syncrude Canada in March 2008, for the completion of Gateway Hill, a 104 hectare area of deciduous and coniferous forests with wetlands; reclamation activities for this area started in 1983 (Canadian Mining Journal, 2008; Government of Alberta, 2008a).

2.3.3. Alberta reclamation regulations

Alberta has province wide industrial development which presents major reclamation and conservation challenges. Significant changes to Alberta's land conservation and reclamation programs have been driven by regulatory policies and objectives, stakeholder and public expectations, advances in reclamation science and practices, land disturbance type and scale, and intended end land use (Powter et al., 2012). Over the past 20 years, changes to Alberta's reclamation policy have enhanced coherence and consistency of the regulatory framework, aligning policy more closely with intended outcomes (Wellstead et al., 2016).

Enacted in June 1963, the Surface Reclamation Act required reclamation of disturbed land in surveyed areas of the province; it addressed landowner concerns with well sites and defined reclamation as disturbed land in proper condition and well maintained (Alberta Environment, 1999; Powter et al., 2012; Bott et al., 2016a). The Surface Reclamation Act was replaced by the Land Surface Conservation and Reclamation Act, which was enacted between July 1973 and August 1978; reclamation and remediation were not defined, and reclamation certificates were issued when disturbed areas were deemed to be in satisfactory condition by the government

(Alberta Environment, 1999; Powter et al., 2012; Bott et al., 2016a). The Environmental Protection and Enhancement Act (EPEA) consolidated and replaced previous legislation for the protection of air, land, and water; enacted in 1993, it provided clear definitions and regulatory requirements and approval terms and conditions for mineable sites (Government of Alberta, 2023).

Regulatory requirements include formal environmental assessment, a public hearing, decision by the Energy Resources Conservation Board or Natural Resources Conservation Board that the project is in the public interest, review of regulatory applications, consideration of public input, issuance of one or more environmental operating approvals and subsequent amendments and renewals, submission of reclamation security, ongoing monitoring and reporting requirements compliance, and enforcement actions and reclamation certification (Alberta Environment, 1999; Government of Alberta, 2016). Projects that do not need to follow the EPEA approval process are subject to the Act. Activities include education, guideline publication, periodic field inspections, compliance and enforcement actions, and reclamation certification (Powter et al., 2012).

Currently, there are no regulations outlining timelines for reclamation and certification. This has resulted in an incremental increase in the number of sites not being reclaimed or certified (Powter et al., 2012). There have been ongoing concerns that many disturbed sites will fail to be reclaimed and liability will fall to landowners; in response to the concerns reclamation programs, guidelines, and working groups were established. The Land Reclamation Program (1973-1993) developed under the provincial government, funded reclamation projects and restored areas with no responsible operators for the land to be returned to a biophysically productive state (Kryviak, 1982; Alberta Environment, 1999). Alberta’s orphan well program began operations in 2002, to reclaim abandoned orphan wells, pipelines, and facilities (Orphan Well Association, 2013).

Reclamation practices and expectations have evolved parallel to changes in regulations. In 1981, the Alberta Soils Advisory Committee released the Proposed Soil Quality Criteria which was updated and published in 1987 as the Soil Quality Criteria Relative to Disturbance and Reclamation (Soil Quality Criteria Working Group, 1987). Planting guidelines using native species and directed by intended end land use were published by working groups and government agencies (Native Plant Working Group, 2000; Alberta Environment, 2003). There is still room for improvement in Alberta’s reclamation regulation including clearly defined reclamation expectations, direction on the relative priority placed on key reclamation criteria, and decision support tools to assist practitioners and regulators with the reclamation application, review, and decision (Tokay et al., 2019). Certification criteria need to be developed for mines, pits, borrow pits, plant sites, brown field, and renewable energies including wind farms (Powter, 2024).

2.4. Reclamation Success

Historically, the most important aspect of reclamation success was vegetation productivity, especially in agricultural lands (Powter et al., 2012). This was not an appropriate measure because vegetation productivity can be affected by numerous factors including drought, floods, fertilizer, and herbicides, which caused delays in regulatory decisions. In 1983, reclamation objectives focused on the ability of landscape and soils to support intended use, with vegetation evaluation to determine expected performance and potential soil contamination (Brocke, 1982).

Successful reclamation and reducing the overall industrial footprint will address growing concerns about habitat fragmentation, maintaining biodiversity, and cumulative impacts (Powter et al., 2012). There have been continued advances in regulatory conditions and reclamation practices, which have placed increasing pressure on regulators and companies to be industry leaders. Emerging advances in science and technology have increased stakeholder expectations and regulatory requirements. This has led to new reclamation options and reassessing reclamation options previously not economically viable, creating opportunity for advancements in reclamation methods and how reclamation success is assessed and measured.

2.4.1. Reclamation success in Alberta

Reclamation success definitions vary depending on initial objectives and end land use goals for the disturbed area. In Alberta, the Alberta Energy Regulator (AER) determines reclamation criteria and issues reclamation certificates. Current criteria include landscape (drainage, erosion, stability, bare areas, contour, amendments, gravel and rocks, debris), soil (required and minimum depth, consistence, texture, structure, colour, rooting restrictions, pH, organic matter, percent clay, electrical conductivity, sodium adsorption ratio), and vegetation (species composition, plant height, density, weight, litter, plant health, weeds). The primary reclamation objective is to obtain equivalent land capability (Alberta Energy Regulator, 2020).

Equivalent land capability is widely interpreted causing debate among academia, environmental organizations, industry, and the public. Criticisms include restoration to pre-disturbance conditions not being required and lack of accountability for loss of carbon storage during mining (Oil Sands Research and Information Network, 2011; Rooney et al., 2012). Many industrial disturbances in Alberta are only a few hectares in size, allowing for reclamation to return the area to original use and function (Powter et al., 2012). Reclamation of large industrial sites such as mines and quarries aims to return land for beneficial use, often resulting in ecosystems with different landscapes, vegetation, and ecological functions than pre-disturbance, but opportunity for new or improved land use options (Powter et al., 2012). Returning ecosystem function and use of native vegetation have received increased attention, as these strongly affect reclamation goals and determining success (Native Plant Working Group, 2000; Alberta Environment, 2003).

Reclamation goals of most mining companies aim to create sustainable ecosystems. Coal mining reclamation objectives include returning land to prior agricultural, forestry, wildlife, or recreational use by managing soil, maintaining water quality and quantity, protecting wildlife, reducing waste, and limiting noise (Coal Association of Canada, 2015). Oil sands mining reclamation aims to develop environmental performance goals balancing resource development and environmental preservation, ensuring sustainability for future generations by returning disturbed land to a safe, biologically self-sustaining state, and creating landscapes that support diverse land uses and meet stakeholder expectations (Syncrude, 2014; Suncor Energy Inc., 2022). Equivalent land capability can represent numerous types of ecosystems and does not provide well defined standards for measuring reclamation success. Successful reclamation requires well defined target ecosystems for the various physical sites in a region, which Alberta and Canada are currently lacking (Timoney, 2015) in many areas.

Over 75 % of land disturbed by coal mining in Alberta has been reclaimed. Plains coal mines have been reclaimed to agricultural land and wildlife habitats, mountain coal mines to forests and wildlife habitats, with lakes in both regions providing recreational opportunities (Alberta Environmental Protection, 1998; Coal Association of Canada, 2018). Oil sands reclamation is ongoing. In 2020 the total active footprint was 105,541 hectares; 104 hectares are certifiably reclaimed, almost 7,500 hectares are permanently reclaimed, approximately 2,100 hectares are temporarily reclaimed, and an additional 1,110 hectares have had soils placed (Alberta Environment and Parks, 2022). As biodiversity, sustainability, and reclamation or restoration of disturbed areas become global focal points there is a need for clearer definitions of reclamation success and a clear plan on how to obtain this goal for various disturbances and end land uses.

2.4.2. Ecological restoration

It is important to differentiate between ecological restoration and reclamation, particularly considering that many reclamation areas, especially those involving highly modified soils, may never fully resemble pre-disturbance conditions. Understanding these distinctions aids in setting realistic expectations for industry and stakeholders regarding the success of reclamation efforts.

Restoration projects provide insights for identifying reclamation areas suitable for targeted restoration, leveraging site specific conditions and soil materials to restore habitats. The Society for Ecological Restoration (SER International Science & Policy Working Group, 2004) outlines nine ecosystem characteristics to measure restoration success: similar diversity and community structure relative to reference sites; presence of indigenous species; presence of functional groups necessary for long-term sustainability; capacity of the physical environment to sustain reproducing populations; normal functioning; integration with the landscape; elimination of potential threats; resilience of natural disturbances; and self-sustainability. Ecosystem restoration goals attempt to replicate high species diversity, vegetation characteristics, and ecosystem processes found in natural sites (Aronson et al., 1993; van Aarde et al., 1996; Reay and Norton, 1999; Passell, 2000; McCoy and Mushinsky, 2002; Pywell et al., 2003; Ruiz-Jaen and Aide, 2005; Prach and Walker, 2019; Cadier et al., 2020; Rydgren et al., 2020; König et al., 2022).

Vegetation parameters are cover of functional groups including forbs, shrubs, and trees; woody plant density; biomass; and vegetation diversity (Federal Geographic Data Committee, 2008). Caution is needed when using functional groups to predict vegetation or ecosystem function changes such as carbon storage, especially in Arctic systems (Thomas et al., 2019). Ecological processes including nutrient cycling and biological interactions provide information on ecosystem resilience (Ruiz-Jaen and Aide, 2005). Measured parameters and how they are assessed will depend on the ecosystem and restoration goal. Changes in diversity, vegetation, and ecological processes provide information on a disturbed ecosystem trajectory. Criteria to evaluate restoration success should be compared with more than one reference site, to inform temporal and spatial ecosystem dynamics and expectations. Reference (controls) sites are needed to help define restoration goals, provide a template for success, and aid in designing monitoring programs (Brinson and Rheinhardt, 1996). The primary objective of restoration success is to provide ideal conditions for native species, yet this goal is rarely tested (Block et al., 2001).

2.4.3. Novel ecosystems

Differentiating between reclamation and restoration is important. However, another aspect to consider is that reclamation sites may transition into, or already represent, novel ecosystems. A novel ecosystem is an anthropogenically influenced system of abiotic, biotic, and social components that self-organizes and exhibits unique qualities without intensive human management (Hobbs et al., 2013). Novel ecosystems arise from areas that have been transformed past a point where practical restoration methods are feasible (Higgs, 2017). Novel ecosystems are considered to have three common characteristics; they are comprised of native and exotic organisms with distinct biophysical conditions and selection pressures, they are resilient and require little human intervention, and they cannot be restored to original conditions

(Hobbs et al., 2013; Morse et al., 2014; Truitt et al., 2015). Some consider novel ecosystems to include those that may require human intervention to be sustainable (Naeth, 2024).

There is concern that industry and government will divert funds for research, mitigation, or restoration, saying novel ecosystems will provide much needed ecosystem services (Murcia et al., 2014). This could marginalize the inherent value of nature and cause a shift to a human focused conservation ethic, creating artificial ecosystems that lack value (Marris, 2009). Others maintain successful reclamation outcomes can be achieved if policy and regulatory requirements have the necessary scope and economic flexibility to account for development of hybrid and novel ecosystems among disturbed mine sites (Audet et al., 2015). Novel ecosystems in reclamation is an ongoing issue; regardless it is imperative to implement reclamation techniques that will lead to successful, sustainable, and resilient ecosystems.

2.5. Reclamation Success Indicators

Reclamation success is difficult to define, and definitions vary depending on the source. There is still considerable confusion about the meaning and application of equivalent land capability as the legislated reclamation objective (Powter, 2024). Extensive land reclamation monitoring and sampling is required to ensure that the reclamation plan is being followed, that there are no site issues, and that criteria will be met to receive a reclamation certificate. A wide ranging reclamation monitoring method is to examine ecosystem function, which can be measured through bioavailable nutrients, plant community composition, litter decomposition rate, soil cation concentrations, and development of a soil surface organic layer (Rowland et al., 2009). These indicators can serve as targets during early reclamation stages, and will monitor ecological development and inform on compliance to reclamation objectives (Doley and Audet, 2014). Current focus for measuring and assessing reclamation success is on soil and vegetation. More recently soil microbial communities and soil invertebrates have been included in reclamation monitoring research, although these parameters have not been included in official industrial reclamation monitoring methods (Gervan et al., 2020; McMahen et al., 2022; Allingham et al., 2023; Mahoney et al., 2023; Santana-Martinez et al., 2024).

Creating reclaimed ecosystems that are sustainable and resilient to disturbances, requires information on recovery of various trophic levels and ecosystem processes (SER International Science & Policy Working Group, 2004; Ruiz-Jaen and Aide, 2005). For assessment of reclamation success, soil invertebrates that accurately reflect the recovery state and are relatively easy to collect and identify are needed (Kremen et al., 1993). The end goal in reclamation is to have a functioning and evolving ecosystem which includes knowing more about diversity and community composition of microorganisms and soil invertebrates that actively contribute to ecosystem functioning, nutrient availability, and soil forming processes. Although there are specific biological indicators being used, combining various parameters has been proposed as early indicators of ecosystem stress (Nannipieri et al., 2002). Incorporation of biological indicators into the understanding of how site physical, chemical, and biological processes create landscape dynamics has not yet been achieved (Johnson and Miyanishi, 2008; Quideau et al., 2013).

2.5.1. Vegetation

Vegetation properties needed to obtain reclamation certification in Alberta, outlined by the Environmental Protection and Enhancement Act (2016), include ecosite type, trends, and performance (cover, density, productivity). Some monitoring programs assess planted vegetation, including seed mixes and planting densities. Target ecosite trajectories need to be identified and vegetation must be compared to forest regeneration standards. Weeds must be characterized and a management plan addressed. Vegetation monitoring programs include assessment of plant species, canopy structure, and productivity (Rochdi et al., 2014). Vegetation cover is considered an indicator of vegetation development, with degree of cover related to resource availability at a reclamation site, number of species and their productivity, and the root system (Jochimsen, 2001). Alternative vegetation parameters for measuring ecological recovery of reclaimed well sites include basal area (m^2/ha), stocking density (plants/ha), volume of coarse woody debris (m^3/ha), and cover of canopy, shrub, total vegetation, wood, and litter (Huggard, 2016). Plant species composition and number and cover of non-native plants are also parameters to consider.

2.5.2. Soil

The EPEA has parameters required for reclaimed soils to be assessed when applying for reclamation certificates. Companies must provide a soil survey map of the reclaimed area, outline soil building materials and placement, soil profile description, substrate and landform development, and soil morphological and physical properties. Reclamation cover materials used must be fully described, and it must be confirmed that soil was replaced under approval conditions, and the capping materials used must be outlined and described.

Soil quality in reclaimed systems reflect soil quality and function prior to mining (Ojekanmi and Chang, 2014). Parameters of interest include nitrogen, microbial respiration, enzyme activities, cation exchange capacity, bulk density, pore volume, and soil water retention capacity (Kong et al., 1980). Cation exchange capacity is used to measure soil fertility and nutrient retention capacity. Soil organic matter in reconstructed soils in oil sands reclamation, was related to time since reclamation and served as a reliable soil monitoring parameter (Turcotte et al., 2009). Soil particle size distribution is important for reclamation success as it will influence revegetation through water holding capacity, bulk density, soil water availability, and nutrient contents and availability (Dickinson et al., 2005). Soil porosity can affect reclamation outcomes; higher porosity has resulted in increased site productivity (Rodrigue and Burger, 2004). Soluble salts, often found in reclaimed soil, affect tree seedling survival and growth; increasing concentrations can decrease productivity and therefore must be monitored (Torbert et al., 1988; Rodrigue and Burger, 2004).

Nitrogen, organic carbon, and phosphorus content in reclaimed soil will initially be low, followed by higher concentrations after organic matter development and incorporation into mineral topsoil. Soil organic matter in reclaimed sites will progressively increase with time due to vegetation growth, decomposition, and microbial activities (Ananyeva et al., 2008). Soil carbon and nitrogen rapidly accumulate in reclamation soils 15 years after reclamation (Šourková et al., 2005). Over 20 years are required for soil microbial biomass and diversity to recover in disturbed soils (Anderson et al., 2008; Banning and Murphy, 2008). When soil indicators are used in an integrative index, they are highly suitable for estimating soil quality and provide stronger indication of reclamation status in degraded areas (Gil-Sotres et al., 2005).

2.5.3. Soil microorganisms

The EPEA does not require monitoring or information on soil microorganisms to apply for a reclamation certificate. The microbial ecology, response, and connection with other abiotic and biotic variables in a reclaimed site are not well understood, it is important to assess effectiveness of reclamation practices on main biotic components, such as soil microorganisms and invertebrates (Dimitriu et al., 2010). Measuring bacteria assisted rates of ammonification, nitrification, and nitrogen fixation can aid in assessing soil fertility and overall effects of environmental disturbances on soil quality. Ammonification is a relatively insensitive parameter as it involves a variety of soil microorganisms; nitrification is sensitive to disturbance as only a few microorganisms are involved in the process (Somerville et al., 1987; Grzyb et al., 2021).

Soil microbial biomass is closely related to reclaimed soil age (Insam and Domsch, 1988). The ratio of soil microbial respiration to total microbial biomass was a strong indicator for soil recovery in forests 1 to 60 years following agricultural abandonment (Zak et al., 1990). Using soil respiration and litter decomposition as success indicators is not always acceptable as they are influenced by soil water content and temperatures, which generally fluctuate within 24 hours (Visser, 1988).

Diversity and frequency of occurrence of soil bacteria and fungi are higher in soil with a wide range of functional attributes (Zak et al., 1992). Soil quality is degraded by surface mining, causing a decrease in soil microbial diversity, likely due to loss of soil organic matter (Visser et al., 1984).

Soil microbial community function and structure can be quite different in reclaimed sites relative to any natural analogs. For example, reclaimed sites had higher dissimilarity when tailings sand was used, and community composition responded differently depending on the reclamation treatments that were implemented (Dimitriu et al., 2010). Both soil microbial community structure and enzyme activities could be used as potential reclamation success indicators. More research is needed to understand the relationships between vegetation composition, environmental factors, and the resulting soil microbial community development (Hahn and Quideau, 2013). This will aid in improving current reclamation practices and regulations.

2.5.4. Soil invertebrates

The EPEA does not include soil invertebrate monitoring in reclamation certificate applications. Soil invertebrates can be used to assess effects of anthropogenic activities, as changes in soil invertebrate diversity and species composition often correspond with ecosystem changes and provide information on ecosystem health, complexity, function, and stability (Majer, 1983; Majer, 1990; Ferguson and Berube, 2004). Invertebrates are commonly used to assess health of fresh water and marine habitats, and to monitor environmental changes in agricultural practices and management (Majer et al., 2007; Aspetti et al., 2010). Soil invertebrates that colonize reclamation sites will be influenced by substrates, vegetation, distance from undisturbed areas, and migration barriers on site (Majer et al., 2007; Macdonald et al., 2015).

Soil invertebrates have been used as biological indicators of ecosystem health, response to anthropogenic activities, and restoration status. For example, the role of soil invertebrates to help ascertain ecosystem recovery after mining operations has been explored with bauxite mining in Australia (Majer et al., 1984; Cuccovia and Kinnear, 1999; Majer et al., 2007; Orabi et al., 2010), brown coal opencast mining in Germany (Dunger et al., 2001; Topp et al., 2010), agriculture restoration in Italy (Santorufo et al., 2012), and forest restoration in China (Ren et al., 2017; Huang et al., 2019), to name a few. Soil invertebrates could be used as indicators of ecosystem health and function or could be used as surrogate indicators of overall diversity and biodiversity. In Australia, invertebrates in manganese and bauxite mining reclamation sites have been extensively studied (Majer, 1983; Majer et al., 1984; Majer, 1989). Formicidae (ants) provided indication of ecosystem recovery after mining and increases in ant fauna after mining reflected general ecosystem recovery. Further research found that invertebrate density, abundance, and composition reflected changes in the environment, and composition of other animal taxa better than plants, terrestrial vertebrates, and birds (Orabi et al., 2010).

2.6. Ecological Importance Of Soil Invertebrates

The soil biological community plays vital roles in ecosystem function and is an essential component of soil quality. Soil invertebrates are grouped by size as microfauna, mesofauna, and macrofauna (Lavelle et al., 2006). Mesofauna are most abundant and include Acari (mites) and Collembola (springtails). Soil fauna assist with organic debris decomposition, soil formation and modification, soil structure improvement, organic matter formation, nutrient cycling and turnover, net primary production, trace gas production, carbon and nitrogen fixation and sequestration, and water infiltration, purification, and storage (Hutson, 1980; Setala and Huhta, 1991; Freckman et al., 1997; Naeem and Li, 1997; Groffman and Bohlen, 1999; Althoff et al., 2009; Colloff, 2011). Without soil fauna it may take 500 to 1000 years to create an inch of topsoil (Gupta et al., 2007).

Soil fauna are very diverse and can exceed above ground faunal and floral diversity by orders of magnitude in many ecosystems around the world (Anderson, 2009). A temperate woodland with one dominant tree species can contain 1000 species/m^2 in the soil (Schaefer and Schauermann, 1990). Biodiversity can enhance ecosystem stability and sustainability. Naeem and Li (1997) suggest large numbers of species should enhance ecosystem reliability, increasing the probability of a system providing consistent performance over time. Soil invertebrate biodiversity is important and has become a focus of international environmental policies such as the European Union Soil Thematic Survey (2006) and the Biodiversity Plan for Agriculture (EU 2001) (Menta, 2012).

Soil invertebrates play numerous and various roles in an ecosystem, including predators, parasites, herbivores, sacrophages, and pollinators, making them ecologically significant and economically important (Rosenberg et al., 1986). Soil invertebrates, especially larger bodied arthropods, play significant roles due to their burrowing, drilling, mixing, and general soil substrate processing activities (Colloff, 2011). These activities provide the soil matrix with spatial complexity consisting of structures such as burrows, pores, and tunnels. Soil invertebrates can be considered ecosystem engineers by assisting soil aeration and water infiltration, which are critical components for biogeochemical reactions and build the foundation for ecological succession.

Soil invertebrates have diverse body sizes, vagilities, growth rates, population sizes, reproductive potentials, generation times, and can occupy various positions in the food web (Peck et al., 1998;

McIntyre et al., 2001; Longcore, 2003). Soil invertebrate diversity and community composition have frequently been directly linked to ecosystem health, biodiversity, function, and stability (Orabi et al., 2010). These characteristics allow soil invertebrates to be used as biological indicators. Proposed criteria for strong biological indicators (Noss, 1990) include: sensitivity to changes for early problem detection, present in a range of geographical areas, and capable of providing continuous assessment over various stressors. Ideal reclamation indicators are sensitive, responsive to management practices, ubiquitous, representative, easy to sample and identify, functionally important (Majer, 1983). Soil invertebrates meet these outlined categories and are ideal surrogates for studying environmental impacts of anthropogenic activities (McIntyre, 2000).

Soil invertebrates can provide alternative and valuable information relative to chemical and physical soil properties and soil microbial biomass (Barbercheck et al., 2009). Measuring soil chemical and physical properties provides information on current soil conditions; while soil invertebrates are exposed to the range of soil and climate conditions and stressors, all these effects have been integrated (O'Neill et al., 2010). Soil texture and pH, and the fungal to bacterial biomass ratio predicted less than 24 % of variation in soil nematode communities (Neher and Campbell, 1994). Soil invertebrates can be sensitive to pesticides, fertilizers, vegetation cover changes, and management practices (Paoletti, 1999; Cole et al., 2005; Eggleton et al., 2005; O'Neill et al., 2010). Soil invertebrates are abundant, diverse, functionally important, and sensitive to soil condition changes (Nahmani and Lavelle, 2002; Andersen and Majer, 2004). The presence or absence of specific groups of soil invertebrates can greatly influence reclamation outcomes (Majer et al., 2007). These observations strongly suggest that soil invertebrates should be used as biological indicators to assess reclamation sites and determine reclamation success.

2.7. Major Soil Invertebrate Groups In Reclamation Monitoring

2.7.1. Acari and Collembola

Acari (mites) and Collembola (springtails) are often researched in connection with soil health and reclamation. They represent the majority of soil mesofauna and occupy every trophic level in the soil (Cuccovia and Kinnear, 1999). Hutson (1980) found despite harsh reclamation conditions in Northumberland, England, mites and Collembola rapidly colonized soil with high densities within two years. The reclaimed area supported large and diverse soil fauna groups within a year of reclamation, although they differed significantly from undisturbed sites (Hutson, 1980). In contrast, in some areas more than ten years were required for reclaimed areas to acquire mite abundance and diversity similar to surrounding undisturbed areas (Cuccovia and Kinnear, 1999). Research in the Russian Arctic approximately 30 years post coal mining, found the ratio of species richness and abundance of Collembola and two mite groups (Mesostigmata, Oribatida) was immature and developing relative to a reference site, likely due to colder temperatures (Coulson et al., 2015).

Species richness of mites and collembolans has a strong positive correlation with the age of a reclamation site (Cuccovia and Kinnear, 1999). In New Zealand mine sites, mite abundance and capture frequency reflected changes in vegetation and were responsive to successional changes over time (Rufaut et al., 2010). Increasing collembolan abundance was associated with increasing vegetation cover and plant species richness, and specific invertebrate species were associated with litter cover (Majer et al., 2007). Ecosystem and invertebrate responses to disturbances and reclamation methods will not be universal. A study in Australia determined mites identified to coarse taxonomic levels were not useful indicators due to high abundance in all samples which overwhelmed contributors from other invertebrate groups. Few environmental variables were significantly correlated with mite abundance or ratios (Nakamura et al., 2003).

2.7.2. Coleoptera

Coleoptera (beetles) have a high number of species and diversified ecological habitat (Topp et al., 2010). Colonizing species were commonly omnivores, scavengers, and generalist feeders which fed on seeds, weedy plants, and living and dead insects (Parmenter and Macmahon, 1987). Beetle abundance increased when canopy cover increased, while species richness was strongly influenced by soil surface structure (Topp et al., 2010). Establishing pre-disturbance beetle community composition is difficult and requires considerable time (Assmann, 1999). Sites 11 to 15 years post-mining and reforestation, had a 32 % similarity of beetles caught in pitfall traps relative to those caught in reference stands (Cooke and Johnson, 2002). Soil and vegetation conditions in reclaimed mine sites may be too different and that restoring pre-disturbance soil faunal communities will not be viable (Parmenter and Macmahon, 1987).

2.7.3. Formicidae

Formicidae (ants) are easily sampled, ubiquitous, and correlated with soil and management (Peck et al., 1998). Ants have been suggested as ideal candidates for terrestrial indicators (Peck et al., 1998; Majer et al., 2007). Nakamura et al. (2003) examined communities of ants, centipedes, millipedes, isopods, true bugs, carabid beetles, and carrion beetles in remnant rainforest, pasture, and revegetated sites in Australia. Ants were the only group to distinguish pasture land from rainforest (Nakamura et al., 2003). Australian bauxite mining reclamation sites had strong positive associations between ant species richness and abundance or richness of other taxonomic groups and reclamation site variables (Majer et al., 2007). The previous research evolved the use of ants as biological indicators of mine reclamation and land management, which was widely accepted in Australia and other parts of the world (Andersen and Majer, 2004).

2.7.4. Hemiptera

Many hemipterans, true bugs, feed on plants and contain both generalist and specialist plant feeding species (Orabi et al., 2010); thus they likely reflect changes to the vegetation community and to vegetation cover. Hemipteran community composition in reclaimed bauxite mining sites was associated with specific plant species (Orabi et al., 2010). Reclamation methods successfully achieved similar species density to undisturbed reference sites, but not overall composition, even though reclamation started 20 years prior. Hemipterans can be used as a good biological indicator as they reflect environmental conditions and change (Orabi et al., 2010). Although there may be similarities between reclaimed and reference areas in richness and abundance of soil invertebrates, species composition usually takes longer to recover after disturbance, especially hemipterans (Moir et al., 2005; Orabi et al., 2010)

2.7.5. Lumbricidae and Nematoda

Lumbricidae (earthworms) and nematodes are sensitive to physical and chemical changes in environment, have limited locomotion, are straightforward to identify to family, and determine soil suitability for almost all soil organisms (Tischer, 2009). Earthworm abundance severely decreases when soil is disturbed, but recovers well with time (Althoff et al., 2009). Nematodes are influenced by disturbance, and community structure can provide a comprehensive assessment of the status of the soil food web and to indicate ecosystem recovery post-disturbance (Althoff et al., 2009). Earthworm assemblages can rapidly develop in reclaimed areas, and forest reclamation was more suitable to their development than agricultural reclamation (Hlava and Kopecký, 2013). Earthworms are sensitive to reclamation methods and time since disturbance, with abundance in older reclamation is similar to natural reference sites (Hlava and Kopecký, 2013).

2.7.6. Other soil invertebrate taxa

Other major soil invertebrate groups include Diplopoda (millipedes), Myriapoda (centipedes), Araneae (spiders), and Isoptera (termites). Millipedes aid in decomposition and influence soil nitrogen, carbon, and magnesium concentrations (Smit and van Aarde, 2001). Termites assist in decomposition and influence soil structure (Lobry de Bruyn and Conacher, 1990). Termite abundance and diversity were positively correlated with time since reclamation and adding coarse woody debris to reclamation sites can accelerate termite recolonization and associated ecosystem processes (Majer et al., 2007). After 19 years, termite species composition in reclaimed bauxite mine areas was comparable to undisturbed forest (Majer et al., 2007). Spider assemblages responded to structural aspects of their habitat (Majer et al., 2007). In Australia, millipede and centipede taxon richness was significantly correlated with environmental variables such as tree spacing and litter index (Nakamura et al., 2003). Spiders show sensitivity to changes in leaf litter depth and presence and abundance of standing vegetation used as attachment points for web building (Uetz, 1979). The thickening of standing vegetation and availability of prey may influence spider recolonization after mining (Simmonds et al., 1994; Majer et al., 2007).

2.8. Soil Invertebrates And Reclamation Monitoring

Soil invertebrates as biological indicators for reclamation assessment have been researched mainly in Australia and Europe (Dunger et al., 2001; Nakamura et al., 2003; Majer et al., 2007; Hartley et al., 2008; Orabi et al., 2010; Topp et al., 2010; Hendrychová et al., 2012; Hlava and Kopecký, 2013; Coulson et al., 2015). Soil invertebrate groups researched most frequently included mites, earthworms and nematodes, and ants. Numerous ecosystems (coastal, prairie, Arctic, subtropical) and disturbance types (agriculture, chemical spill, mining) were researched. Common soil invertebrate sampling methods include pitfall trapping, soil cores with extraction using Tullgren funnels, sweeping and beating, and manual collection. Taxonomic resolution of specimen identification is often to species level, though morphospecies, broader taxonomic classification to family and sometimes order, and overall soil invertebrate abundance can be used.

Hendrychová et al. (2012) found species richness of various soil invertebrate groups was affected by soil properties and management practices. Mining companies and reclamation practitioners have been advised to revegetate disturbed areas using diverse methods to maximize return of soil biodiversity and support to its full range (Majer et al., 2007; Hendrychová et al., 2012). A review of regulatory performance standards and monitoring requirements in nine North American jurisdictions, to evaluate land reclamation success, suggested that as an alternative to existing approaches, development of the soil invertebrate community, including trophic organization, species dominance, turnover rates, and community composition be used to assess land reclamation (Smyth and Dearden, 1998). Invertebrate functional groups that influence ecosystem processes, such as pollinators, herbivores, predators, and detritivores were recommended as indicators. The abundance of the invertebrate indicators can be compared between sites which will then be used to evaluate overall establishment and maintenance.

Common soil invertebrate measurements currently include a diversity index (Shannon-Wiener index), abundance, frequency, density, biological index of soil quality, and disturbance effect index. Some soil invertebrate groups are cost effective and easy to sample. They are strong representatives of the arthropod community, correlated well to soil and management practices, and taxonomically understood. However, using one single group is unlikely to be appropriate or effective across all ecosystems and establishing a universal indicator is unlikely to occur without advancement in both knowledge and technology.

The impact of reclamation on soil invertebrates after bauxite mining in the Jarrah Forest of Western Australia has been extensively researched. There have been over 20 studies including arthropods in soil and leaf litter, understory vegetation, and the tree canopy (Majer et al., 2007). Projects included various trophic groups (decomposers, predators, herbivores); invertebrate groups included Hemiptera, spiders, scorpions, termites, Collembola, mites, earthworms, and ants. Research projects have been long term with regular sampling. Results show species composition of most invertebrate groups becomes more similar to the surrounding unmined forest given enough time, but distinct differences remain. Determining causes and mechanisms surrounding these differences is an ongoing challenge. Future research work will focus on soil fauna and their relationship with soil structure and the role of invertebrates in pollination. The bauxite mining operation, Alcoa World Alumina, has invested significantly in reclamation and restoration research, leading to advances in understanding the role of soil invertebrates in reclamation, optimizing invertebrate sampling techniques, and documenting biodiversity in southwestern Australia. This commitment could serve as a model for mining companies in Alberta, potentially encouraging them to monitor soil invertebrates in their reclamation sites, thereby enhancing ecological outcomes and demonstrating corporate responsibility.

3. RESEARCH CHALLENGES, SIGNIFICANCE AND OBJECTIVES

3.1. Research Challenges

Currently, the largest challenge to incorporating soil invertebrates into reclamation practices and monitoring is the high level of taxonomic expertise needed to measure diversity (McGeoch and Chown, 1998). A large proportion of soil invertebrate species are undescribed and very little is known about their biology, distribution, and functional roles in the ecosystem (Greenslade, 2007). This challenge is termed the ‘taxonomic impediment’ (Greenslade, 2007). In North America, it has been estimated that 75 % of oribatid mite species have not yet been described (Behan-Pelletier and Bissett, 1992). It is more economically efficient to complete vegetation assessments and collect soil cores for analytical processes. Numerous resources and courses are available for vegetation identification, and required equipment at most, is a hand lens and taxonomic keys. Soil chemical analysis can be costly; however, time and equipment for sample collection is minimal. There is little incentive to include soil invertebrates in reclamation criteria and monitoring, as assessment of current indicators requires less time, money, specialized knowledge, and can be completed by most trained individuals. There are strong arguments that indicators for reclamation and restoration need to include more than soils and plants as soil invertebrates could give a better understanding of ecosystem function and reclamation trajectory (Majer et al., 2007).

Use of coarse taxonomic resolution has been proposed to overcome the taxonomic impediment (Nakamura et al., 2003). A broader approach to taxonomy will require little expertise and decrease time required for assemblage composition identification (Nakamura et al., 2003). When using broad taxonomic approaches, there is a risk that order level aggregation will obscure variation important to habitat assessment, especially if families, genera, and species react differently to environmental conditions (Andersen, 1999). However, taxonomically precise studies may not be appropriate surrogates for overall diversity (Andersen, 1999). More studies on taxonomic breadth and soil invertebrate response to environmental depth is needed (Meehan et al., 2019).

A method for broader taxonomic assessment, a frequency score (number of quadrats per site with a particular group present), was an acceptable measure of revegetation in research reclamation sites in Australia (Nakamura et al., 2003). Frequency measures eliminate the need to count individual invertebrates, which in turn will be more financially feasible. When using coarse taxonomic methods, if a reclaimed site shows similarity to the natural analog, species composition and species richness can still vary within the coarse soil invertebrate groupings. When sorted to species, invertebrate responses to ecological differences between reclaimed and natural analog sites are better detected. Frequency scores were sufficient for achieving separation of site types.

Another proposed soil fauna index eliminates complex taxonomic identification and overcomes the limitation of presence/absence data (Yan et al., 2012). The novel index uses diversity of the soil faunal community in conjunction with its functional traits, and abundance (Abundance-based Fauna Index [FAI]). FAI values were associated with soil quality and could provide information linking soil fauna functional to the environmental conditions below ground. Further research and testing will be required to examine the ecological validity and long-term repeatability.

Alternative synthetic parameters can be measured such as mite:springtail ratio and soil biological quality index (QBS) (Santorufo et al., 2012). The QBS index categorizes soil invertebrates based on morphology and assumes high soil quality will be reflected in higher abundance of soil invertebrates well adapted to soil habitats (Parisi et al., 2005). Although easy to calculate, it can be difficult to interpret the parameters. A high value for the mite:springtail ratio can suggest high soil quality, as the number of mite species can decrease with soil degradation; however, this index is not reliable across all ecosystems and values are sometimes not comparable (Santorufo et al., 2012). A single metric called the Indicator Value (IV) is a proposed approach for identifying indicator taxa that integrates the degree of uniqueness, abundance, and occurrence of a taxon in a particular habitat (Dufrêne and Legendre, 1997). A high score reflects high information content and a high probability of being sampled (McGeoch et al., 2002). Since this value is absolute and calculated independently from other species, it can overcome the limitations that are commonly associated with using parametric and multivariate methods (Dufrêne and Legendre, 1997).

Various methods are available to identify indicators for ecological monitoring, each with distinct parameters and strengths that must be carefully considered in the selection process. Indicator Species Analysis is commonly used in conservation and restoration practices with a wide range of organisms (O'Neill et al., 2010). Based on research by Santorufo et al. (2012), QBS seems to be the most appropriate measurement for soil quality. It is important when researching soil invertebrate communities to measure more than one community parameter. The Simpson and Shannon-Wiener diversity indices should be used with caution when evaluating soil quality and invertebrate community structure (Santorufo et al., 2012). Further research is needed to ascertain community parameters best suited for invertebrates, ecosystem types, and disturbance type and severity. While unlikely, a unified universal explanatory parameter for measuring soil invertebrates would allow for more concise research and make comparing results more feasible.

Soil invertebrate succession and contribution to ecological succession during reclamation is not well understood. Succession behaviour and trajectories for many plant species is clear, but lacking for soil invertebrates. There is an absence of baseline data that are available to compare reclamation sites (McGeoch and Chown, 1998). Stages of soil invertebrate colonization in clean soils remediated from oil pollution are described as simple invertebrate communities dominated by predators (mesostigmatid mites), followed by Collembola, then there is an appearance of oribatid mites (Melekhina et al., 2021). Newly restored or reclaimed areas will commonly have a high abundance of generalist species, whereas the surrounding undisturbed natural areas will often have high levels of specialist species (Majer et al., 2007). Majer et al. (2007) suggested studies involving a chronosequence and following succession by revisiting reclamation sites repeatedly over extended periods of time. Long-term studies of arthropod community development in response to ecosystem disturbance are critical for determining direction, mediating variables, and eventual succession outcome (Parmenter and Macmahon, 1987). Determining successional stages that soil invertebrates follow post disturbance and during and after reclamation will greatly enhance the current level of understanding.

Key soil invertebrate groups are useful in ascertaining ecosystem recovery and have met other indicator requirements. To provide value to ecosystem recovery and reclamation monitoring, soil invertebrate groups need to be applicable across regions and across various land uses (Barbercheck et al., 2009). Research objectives need to include numerous disturbances, end land uses, and ecosystems. Soil mites have potential as indicators for disturbances in forests, wetlands, and agricultural operations (Donegan et al., 2001). Barbercheck et al. (2009) found collembolan communities were a promising indicator, as monitoring results were consistent in forest, wetland, and agricultural ecosystems. Ants are another potential indicator as they respond negatively to disturbance and have shown influence over changes in physical and chemical soil properties, plants, and other soil organisms (Majer, 1983; Peck et al., 1998).These major invertebrate groups require further research in various ecosystems, using a variety of reclamation methods, following different disturbances types.

3.2. Research Significance

Alberta's expanding disturbance footprint from various sources creates a need for refined reclamation methods and monitoring. Disturbances occur in all ecosystems. There is a unique opportunity now to research soil invertebrate response to reclamation methods and successional stages by monitoring sites regularly to provide baseline data for future reference. Moving this research and monitoring approach forward will enhance our current understanding of soil invertebrate biodiversity in various regions across Canada.

Disturbed soil physical and chemical properties will affect the structure and function of corresponding soil fauna (Althoff et al., 2009). The rate at which soil invertebrate communities recover from these various disturbances provides a beneficial indicator of soil and ecosystem function, and ultimately of resilience (Lavelle et al., 2006; Maggiotto et al., 2019; Auclerc et al., 2022). Given the increased environmental impacts from anthropogenic activities and from climate change, ecosystem resiliency to stressors and change is very important. Continued research is needed to link resilience in reclamation with continued monitoring of soil invertebrates and adjusting current reclamation goals to include pre-disturbance invertebrate community composition, diversity, and abundance.

3.3. Research Objectives

The overall objective of my research is to advance the science of land reclamation success indicators, and to identify the most ecologically effective indicators while maximizing resource use in monitoring and assessment to meet government criteria and other regulatory requirements.

Specific research objectives include: evaluating differences in soil invertebrate groups in forest reclamation and undisturbed forest reference sites, determining which reclamation indicator best captures site variability, determining if addition of soil invertebrates to current indicators alters reclamation assessment results, and recommending soil invertebrate groups of interest in reclamation monitoring, including appropriate sampling methods and collection timing.

4. THESIS STRUCTURE

This thesis is structured with an Introduction, followed by four chapters addressing different aspects of soil invertebrate dynamics in reclaimed sites, and a synthesis chapter. The structure of my thesis is designed to mirror a reclamation site assessment; beginning with a detailed analysis of plant communities, dominant plant species, and soil properties. It then incorporates soil invertebrate taxa abundance to evaluate changes in site interpretation and to determine whether these indicators enhance our understanding of reclaimed sites. This is followed by an examination of the response of various soil invertebrate taxa to reclamation methods across the growing season, assessing which groups serve as consistent indicators regardless of sampling time. The thesis then delves into species level responses of soil invertebrates, specifically beetles and spiders, to reclamation methods, exploring the potential of species level data as indicators and their relationship to broader soil invertebrate taxa abundance trends.

Chapter I serves as an introduction, provides background information, and outlines research significance, challenges, and objectives. Chapter II investigates reclaimed soil chemical properties and vegetation community composition relative to the reference, and the impact of incorporating soil invertebrate metrics. It aims to identify potential soil invertebrate indicators and determine sensitivity relative to soil and vegetation properties. Chapter III focuses on soil invertebrates across the growing season, comparing reclaimed assemblages to reference conditions, evaluating consistency and indicator recommendation. Chapters IV and V delve into two well studied soil invertebrate groups, beetles and spiders, examining species level data with abundance to determine if community composition is affected. Chapter VI synthesizes research findings, discusses limitations, and explores future applications for reclamation efforts.

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CHAPTER II: CAN SOIL INVERTEBRATES ENHANCE RECLAMATION MONITORING PRACTICES IN RECLAIMED FORESTED LANDS

1. INTRODUCTION

Canada's boreal forest covers 270 million hectares, comprises over 25 % of the global boreal zone, and provides critical ecosystem services including water and air purification, carbon storage, climate regulation, economically significant resources, and cultural significance (Kayes and Mallik, 2020; Natural Resources Canada, 2021). Ongoing anthropogenic disturbances attributed to forestry and energy industries, have caused habitat fragmentation and loss of complex wetlands, while introducing non-native species and decreasing biodiversity (National Council for Air and Stream Improvement Inc., 2008; Langor et al., 2014; Venier et al., 2014; Kayes and Mallik, 2020). Analysis of temporal trends of landscape pattern indices for Canada's boreal forest, found the largest decline in forest cover in the Boreal Shield, Boreal Plain, and Boreal Cordillera ecozones and increased forest edge density in all ecozones (Pickell et al., 2016).

Energy related disturbances in the boreal forest include surface mining of oil sands and coal, in situ oil sands operations, seismic lines, pipelines, fracking, and associated infrastructure; the cumulative effects of these activities are difficult to quantify (Mahon et al., 2019; Crosby et al., 2023). Researchers estimating cumulative effects in the boreal region on migratory songbird species distribution suggest effective management of cumulative effects includes regulation and planning spanning organizational levels, which reflects hierarchically nested spatial scales that align with scale domains of relevant ecological processes for species and management objectives (Crosby et al., 2023). Energy sector stressors often had additive or interactive effects with forestry stressors on abundance of 27 land bird species in the boreal; these interactive cumulative effects from multiple sectors present a major challenge for impact assessments (Mahon et al., 2019). An assessment of cumulative effects of natural and anthropogenic disturbances on forest carbon stocks and fluxes at a 1.3 million ha pilot study area in Alberta's oil sands region, found the study area changed from a net carbon sink to a net carbon source over 28 years (Shaw et al., 2021).

The scale of disturbance in Alberta is substantial and land reclamation will be tasked with mitigating environmental impacts from energy sector activities; therefore evaluating existing reclamation monitoring and outcomes is important to ensure reclaimed areas can be self-sustaining (Hawkes and Donald, 2012). Effective monitoring is paramount for understanding ecosystem recovery after natural or anthropogenic disturbances, including managing biodiversity, assessing long-term impacts, and modeling to predict future ecological challenges. Environmental monitoring programs, such as the Alberta Biodiversity Monitoring Institute (ABMI) that tracks changes in Alberta’s wildlife and their habitats, and the Oil Sands Monitoring Program that assesses long-term cumulative environmental impacts from oil sands, show the scope of long-term monitoring provincially (Alberta Biodiversity Monitoring Institute, 2014; Government of Alberta, 2024b). However, even systematic, collaborative monitoring programs face continued challenges with consistent data collection and comparability (Ellingsen et al., 2017). By learning from these challenges, we can design a standardized, province-wide reclamation monitoring program that ensures the collection of meaningful, scientifically consistent data, enabling accurate assessment and reporting of reclamation sites, tracking of reclamation trajectories, and the development of predictive models for reclamation success and certification timelines. To advance reclamation criteria, reclamation targets should be based on current and anticipated conditions of post-disturbance landscapes to determine appropriate ecosystem and management practices for existing site conditions, ideally leading to ecosystems with the greatest ecological resilience under future climate conditions and changing environmental stressors and drivers (Audet et al., 2015).

In Alberta, under the Environmental Protection and Enhancement Act (EPEA) and the Conservation and Reclamation Regulations, areas disturbed by oil and gas activities are required to return disturbed areas to equivalent land capability (Government of Alberta, 2023). Companies have a duty to reduce land disturbance; remediate contamination; salvage, store, and replace soil; and revegetate impacted areas (Alberta Energy Regulator, 2023). Many industrial land disturbances in Alberta are only a few hectares in size, allowing for reclamation to return the area to original land use and function; for larger footprints, such as surface mines, the primary goal is to return the area to useable land that will provide a net benefit to landowners and society, resulting in an ecosystem that diverges in landscape, vegetation, and ecological functions from pre-disturbance (Powter et al., 2012). The range of interpretations for equivalent land capability has caused debate among the public, academia, environmental organizations, and industry. Criticisms include that restoration to pre-disturbance conditions is not required and well defined standards for measuring reclamation success are not clear (Oil Sands Research and Information Network, 2011; Rooney et al., 2012). Successful reclamation requires well defined target ecosystems for the variety of physical sites in a region, which Alberta, and to a larger extent Canada, is lacking (Timoney, 2015). There are suggested targets for upland forest ecosystems, but as biodiversity, sustainability, and restoration of disturbed areas become global focal points, clearer definitions of reclamation success and how to obtain it are needed (Gosselin et al., 2010).

Early reclamation objectives, primarily aimed at vegetation establishment mainly for erosion control through use of agronomic species and fertilizer applications, have shifted towards a focus on integrating native plant species and fostering self-sustaining ecosystems. As land reclamation science has evolved, projects increasingly prioritized native plant species establishment, wildlife habitat creation, and natural ecosystem trajectories, which introduced an emphasis on returning ecosystem function, which in turn impacts reclamation goals and definitions of success (Native Plant Working Group, 2000; Alberta Environment, 2003). Some believe accountability is lacking for the loss of carbon stored in soil and vegetation and released during mining (Rooney et al., 2012). Soil organic matter in reclaimed soils is significantly lower, while salinity can be significantly higher, presenting issues when planting native species not accustomed to high salt concentrations (Purdy et al., 2005; Turcotte et al., 2009).

The Alberta Energy Regulator (AER) controls reclamation criteria, which vary depending on land type, and will issue reclamation certificates. Currently there are criteria for landscape (drainage, erosion, stability, bare areas, contour, amendments, gravel and rocks, debris), soil (required and minimum depth, consistence, texture, structure, colour, rooting restrictions, pH, organic matter, percent clay, electrical conductivity, sodium adsorption ratio), and vegetation (species composition, plant height, density, weight, litter, plant health, weeds). A company's reclamation efforts are deemed successful when a reclamation certificate is obtained, indicating the reclamation site is within 80 % of the reclamation criteria (Alberta Energy Regulator, 2019).

Reclamation criteria in Alberta have evolved, with notable updates in 1995 and 2010 shifting from the pre-1995 emphasis on rapid vegetation cover establishment and reducing soil erosion often utilizing agronomic species. Shifts include significant regulatory adjustments to incorporate woody vegetation and vegetation strata requirements into forest reclamation criteria (Land Conservation and Reclamation Council, 1982; Shergill, 1995; Sinton, 2011). Current land reclamation programs do not guarantee equivalent land capability as directed by provincial legislation and there are still long-term biological impacts on newly reclaimed sites using updated criteria (Janz et al., 2019; Lupardus et al., 2020). An assessment of pre-1995 and 2010 reclamation criteria on mineral surface leases in the boreal forest, found 2010 reclamation criteria appear to be more effectively promoting ecosystem recovery than pre-1995 criteria (Baah-Acheamfour et al., 2022). Some certified reclaimed well pads in the boreal forest are in a state of arrested succession, with vegetation remaining dissimilar to the forest reference up to 48 years after reclamation (Lupardus et al., 2019). Changes to the reclamation criteria from 1995 to 2010 have proven beneficial, particularly in areas like soil quality indicators, woody stem requirements, and native plant coverage (Baah-Acheamfour et al., 2022). Updating reclamation criteria has ecologically benefitted forest reclamation sites, and continued updating of criteria and associated legislation, especially in areas related to soil quality, could provide additional benefits and further enhance reclamation outcomes. This approach is needed to ensure long-term ecological recovery of areas that have been disturbed by oil and gas activities.

Soil invertebrates can be effective indicators of forest disturbance and recovery (Battigelli, 2000; Meehan et al., 2019) and reclamation practices (Battigelli, 2011; Hammond et al., 2018; McAdams et al., 2018; Hammond et al., 2022). However, depending on the end land use, soil invertebrate indicators may not be overly appropriate for estimating soil biological quality, especially in cultivated agricultural areas (Lupardus et al., 2021). A nationwide project in the United Kingdom assessed soil mesofauna high level taxonomic abundance and found that it accurately reflected ecoregions and that it helped measure total national soil biomass (George et al., 2017). Soil invertebrate high level taxonomic abundance could thus be an important addition to reclamation monitoring and in gaining a deeper understanding of Canadian soil biodiversity. In Alberta, there is currently limited research linking above ground vegetation composition and below ground soil and soil invertebrate assemblages and abundance in reclamation sites. There is research on soil invertebrate response to reclamation methods on well pads in cultivated sites (Lupardus et al., 2021), to operational oil sands mine reclamation (Hammond et al., 2018), to soil preparation methods during in situ oil and gas reclamation (Hammond et al., 2022), and recovery in forested sites following oil sands mining (McAdams et al., 2018). For soil invertebrate abundances to be included in reclamation criteria, the assessed metrics must enhance our current monitoring practices, be economically feasible, and be easily carried out by general reclamation practitioners to ensure consistent, accurate reporting.

To guide revisions of evolving reclamation practices and criteria, I have addressed four research objectives. These objectives are i) to evaluate the effects of land reclamation methods (direct soil placement and natural revegetation versus stockpiled soil and planting) on vegetation health, cover, and community composition, and soil properties; ii) to determine whether vegetation, soil, and/or soil invertebrates best capture reclamation site variability; iii) to determine if adding soil invertebrate taxa abundance to our current indicators alters success interpretations for reclamation sites; and iv) to determine which soil invertebrate groups contribute the most to differentiating reclaimed and undisturbed forest sites. We examine two reclamation methods and their effects on vegetation composition and soil properties, and whether quantitative soil invertebrate metrics are strongly impacted.

2. MATERIALS AND METHODS

2.1. Research Area And Sites

The research was conducted at the Genesee coal mine, located approximately 70 km southwest of Edmonton, Alberta (53°54'N 113°49'W). The mine footprint covers 73.2 km^2 and is located in the Dry Mixedwood Natural Subregion of the Boreal Forest Natural Region (Natural Regions Committee, 2006; Natural Resources Canada, 2012). Genesee Mine reclaimed 0.47 km^2 in 2015, and in total over the last 25 years has reclaimed 9.53 km^2 (Capital Power, 2016). Land use within the Genesee area includes cultivation agriculture and pasture, and contains fragmented forests and peatlands. There is a high level of disturbance and agronomic species cover in the area.

The area is characterized by moderately well drained wooded soils developed on dark coloured, medium to fine textured, lacustrine material, that contain the Macola and Maywood Soil Series (Lindsay et al., 1968). Macola soils have a thick dark coloured Ah horizon with a thin Ae horizon; Maywood soils have a pronounced light coloured Ae horizon underneath the LFH layer (Table A.1). The research area is located in a transitional region between Black Chernozems and Gray Luvisols; dominant soils include Dark Gray Luvisols, Humic Gleysols, and Mesisols (Navus Environmental Inc., 2012; Stantec Consulting Ltd., 2013). Water bodies in the area are the cooling pond, sewage lagoon, effluent settling pond, and some natural open water wetlands.

Monthly minimum, maximum, and mean air temperature and precipitation were obtained from the Saint Francis Meteorological Station, 10 km east of the research area (Figure A.1). Recorded daily air temperatures for the research years (2017 to 2019) were between 34.6 and 6.3 °C for the growing season (May to October). Cumulative precipitation for the growing season (May to October) of each sampling year was 90.1 mm (2017), 123.4 mm (2018), and 345.6 mm (2019). In 2019, June and July recorded precipitation was over 100 mm, while September precipitation was over 20 mm. In 2017 and 2018 September had the highest precipitation.

Three research sites were selected based on reclamation prescriptions, end land use, and proximity to an undisturbed forest reference (Figure 2.1). Sites had comparable slope, aspect, topography, and drainage. Reclamation sites were named after respective tree species and received different soil reconstruction treatments and revegetation methods. Post-mining, subsoil was replaced and areas recontoured to be well drained with a gentle slope (5 to 9 %). The aspen reclamation site (hereafter Aspen) had direct placement of salvaged forest surface soil (LFH, Ae) to 20 cm depth in January 2009; soil amendments included salvaged coarse woody debris and straw spread to a depth of 2 to 5 cm. Revegetation resulted from seeds and vegetative propagules from the forest soil seedbank and non-native species from surrounding agro-ecosystems (Navus Environmental Inc., 2012). The spruce reclamation site (hereafter Spruce), had soils placed in 2010 to a 15 cm depth with white spruce trees planted in 2013. Soil was stockpiled, but stockpiling duration, amendments, and fertilizer applications were not recorded. A forage mixture (cover crop) was likely seeded in 2010 after topsoil placement to avoid erosion. The undisturbed forest reference (hereafter Reference) was closer to Aspen than to Spruce, and was a relatively undisturbed late seral aspen dominated mixedwood stand (Figure 2.2).

At each research site, ten plots (5 x 5 m) were established, selected for similar drainage and slope while capturing site variability, and at least 5 m buffer region between plots. Each plot was divided into a grid system with a total of twenty-five 1 by 1 m subplots. In July 2017, five random subplots were used for vegetation assessments, followed by pitfall trap installation, and collection of litter and soil samples for soil invertebrate extraction, and soil cores (0 to 15 cm, 15 to 30 cm) for soil chemical and physical analysis. In July 2018 and 2019 new random subplots were used for vegetation assessments; if the random number fell on a square that already had a pitfall trap installed a new one was selected. Soil invertebrates were also sampled at that time.

2.2. Vegetation Assessments

Vegetation assessments were conducted on July 27, July 25, and August 2, from 2017 to 2019. Percent cover for living plant species was determined using a nested quadrat system. Total moss cover was determined in 10 by 10 cm (0.01 m^2) quadrats, herbaceous plant species in 50 by 50 cm (0.25 m^2) quadrats, shrubs in 1 by 1 m (1 m^2) quadrats, and trees in 5 by 5 m (25 m^2) quadrats. Vegetation identification was completed in the field using Common Plants of the Western Rangelands Volume 1: Grasses and Grass like species, Volume 2: Trees and Shrubs, and Volume 3: Forbs (Alberta Government et al., 2003a, 2003b, 2003c). Identifications were confirmed with Flora of Alberta (Moss and Packer, 1994). Grasses and forbs were categorized as native or non-native based on origin information in the Flora of Alberta (Moss and Packer, 1994) and Native Plant Revegetation Guidelines for Alberta (Native Plant Working Group, 2000); weeds and noxious weeds were determined as per Alberta's Weed Control Act (Government of Alberta, 2010, 2008b). Vegetation species nomenclature follows Moss and Packer (1994).

Overall cover was visually assessed for live vegetation by species, bare ground, litter, and other (scat, rock) using 1 m^2 quadrats. Plant health was visually assessed by species and scored using a five point scale (Naeth, 2016b). A score of 1 = plant is healthy and > 90 % green, with little necrosis or chlorosis; 2 = plant is mostly healthy and 75 to 90 % green, with minor chlorosis; 3 = plant is less healthy and 25 to 75 % green, with necrosis, chlorosis and wilting; 4 = plant is dying and < 25 % green, with necrosis, chlorosis and wilting; 5 = plant is dead.

2.3. Sample Collection And Analysis

Soil was sampled August 1, 2017 using a Dutch auger at two depth intervals, 0 to 15 cm (topsoil) and 16 to 30 cm (subsoil). Depth increments were used to ensure sampling consistency. A description of the soil profile including characterization of horizons (depth, colour, texture) and determination of rooting depth was conducted for each site using a soil pit (Table A.2); soil texture ranged from sandy loam to clay loam (Table A.3). Soil profiles within the reclamation sites were generally moderately drained to a depth of 60 cm, had few coarse fragments, and showed little evidence of pedogenic processes below 20 cm. Soil samples were collected in sealed plastic bags, stored in coolers with ice packs for transport, and taken within 24 hours to a commercial laboratory for analysis (Table 2.1).

Soil invertebrates were collected from five randomly located subplots per plot, on July 27 (2017), July 12 (2018), and July 11 (2019), totaling 50 samples per site, per year. Pitfall traps were used to sample above ground invertebrates such as beetles and ants (Greenslade, 1964). Traps were made of clear 16 oz plastic containers (height 7.62 cm, top diameter 11.75 cm) filled with 200 mL of propylene glycol, and a styrofoam plate roof was anchored with bamboo skewers no more than 3 cm above the trap. Traps were left in the field for one week at a time. Below ground invertebrates were collected from leaf litter (LFH) and mineral soil using a 10 by 10 cm quadrat. All litter and organic matter were collected until mineral soil was reached, then mineral soil was sampled to a depth of 10 cm with a 5 cm diameter metal soil core. Invertebrate extraction was performed using the Berlese-Tullgren funnel method (Berlese, 1905; Crossley and Blair, 1991; Tullgren, 1918). within 7 days of field collection (Alberta Biodiversity Monitoring Institute, 2009). Samples were under 20 watt lightbulbs for 7 days and were not sieved after extraction.

Pitfall trap specimens were sorted in the laboratory into coarse taxonomic categories, mostly to order except for ants (Formicidae), using field guides for North America (Borror et al., 1998; Eaton and Kaufman, 2007). Harvestmen (Opiliones) were not differentiated from spiders (Araneae) and they were grouped together for analysis. Abundance is the number of individuals per taxon per pitfall trap. Litter and soil extracted specimens were preserved in 95 % ethanol until processing. Soil mesofauna were separated into Collembola and major mite orders including Astigmata, Oribatida other than Astigmata, Mesostigmata and Prostigmata (suborder). Specimens were identified using a dissecting microscope and unpublished keys from the 2018 Summer Acarology Program at Ohio State University (Walter and Beaulieu, 2014). Abundance is the number of individuals per taxon per sample.

Additional soil invertebrate data, including monthly abundance over the growing season, and species specific trends, is described in the following chapters.

2.4. Statistical Analyses

Plots at each research site were treated as subsamples. We calculated summary statistics for soil properties and vegetation cover by species and functional group including calculations of mean, standard error, and standard deviation.

All statistics were performed with the R statistical package (version 4.1.3) (R Core Team, 2023). Partial and simple Mantel tests (vegan) were used to calculate correlations between dissimilarity matrices for vegetation community composition, soil properties, and soil invertebrate taxa abundance. Mantel R values ranged from −1 (negative correlation) to 1 (positive correlation), with 0 representing no relationship between matrices. Partial mantel tests measure correlation between two dissimilarity matrices while controlling for effects of a third matrix. We used Bray-Curtis dissimilarity, Pearson correlation, and 999 permutations.

2.4.1. Unconstrained ordination

Soil properties, vegetation composition, and soil invertebrate assemblages were visualized using non metric multidimensional scaling (NMDS) unconstrained ordination with the metaMDS function from the vegan package, and Bray-Curtis as the distance measure (Oksanen et al., 2015). Vegetation cover data were Hellinger transformed, with the decostand function in the vegan package, to account for low counts and zeros (Legendre and Gallagher, 2001). Number of dimensions were determined using the dimcheckMDS function in the goeveg package and evaluating scree plots and reduction in stress with decreasing dimensionality (McCune and Grace, 2002). Significant correlated vegetation species, soil invertebrate taxa, soil properties, and environmental variables were visualized as vectors on the ordination plot, using vec and envfit functions in the vegan package (alpha = 0.001, permutations = 999). Vectors for soil properties were selected when Pearson R > 0.6, while vegetation species vectors and soil invertebrate taxa vectors were selected when Pearson R > 0.3. Permutational multivariate ANOVAs (perMANOVA) were performed with the adonis function in the vegan package, to determine significant differences between soil properties, vegetation composition, and soil invertebrate assemblages recorded in Reference relative to reclamation sites (alpha = 0.001, permutations = 999). P-values were corrected by Bonferroni for repeated tests using the RVAideMemoire package.

We evaluated differences in soils, vegetation, and soil invertebrate assemblages using homogeneity of dispersion tests (betadisper), followed by ANOVAs to compare mean distance-to-centroid of vegetation composition and soil properties. Significant ANOVAs (alpha < 0.05) were followed by Tukey's HSD post-hoc test to identify pairwise differences among the research sites.

Indicator species analysis (ISA) determines significant indicators based on aspecificity (A), which is the probability the surveyed site is part of the target site group; and given a species presence and sensitivity (B), probability of finding the species in sites belonging to the target site group. The indicator value index assesses predictive value of a species as an indicator of a combination of site groups, but fails to account for species absences inside and outside the site group combination (De Cáceres et al., 2010). Correlation indices assess positive or negative preference of a species within the site group combination, compared to the remaining sites; negative index values suggest a species avoiding particular site groups (Chytrý et al., 2002; De Cáceres et al., 2010). ISA was used to determine indicator values and point-biseral correlation coefficients (r_{pb}), which can be used with species abundance data (De Cáceres and Legendre, 2009). In our study, r_{pb} is the Pearson correlation between a binary variable indicating whether a site belongs to a site group combination (reclaimed or reference), and a quantitative variable containing species abundance. Analysis was performed using the multipatt function from the indicspecies package (Dufrêne and Legendre, 1997; De Cáceres and Legendre, 2009) to determine which plant species, soil properties, and soil invertebrate taxa were indicators of Reference or reclamation sites (whether certain vegetation species or invertebrate taxa were more abundant in undisturbed or reclaimed). We used a quantitative or binary response with a randomization test (alpha = 0.001, permutations = 999). The point-biseral correlation coefficient (r_{pb}) was calculated when no appropriate indicators were selected based on aspecificity and sensitivity scores.

2.4.2. Constrained ordination

Distance based redundancy analysis (db-RDA) constrained ordination was used to evaluate relationships between below ground soil chemical properties and invertebrate assemblages and abundance from each quadrat with above ground vegetation community composition and cover. Prior to db-RDA analysis, plant species cover data were Hellinger transformed using the decostand function in vegan to give lower weight to rare species (Legendre and Gallagher, 2001; Borcard et al., 2011). We used the capscale function in vegan and Bray-Curtis as a distance measure. The global model was tested using ANOVA ($p > 0.001$). Forward step-wise selection of soil variables was done with ordistep function and 999 permutations (alpha = 0.001). To visualize plant species and environmental variables we used ordiplot, with scaling 2 for species and site scores where species are scaled proportional to eigenvalues; sites are unscaled and have weighted dispersion equal on all dimensions. Points in ordination space were coded by site (Aspen, Spruce, Reference). Significant soil and soil invertebrate variables in db-RDA were tested using anova.cca () function. We partitioned variation in vegetation species composition between soil and soil invertebrate variables using the varpart () function in vegan and visualized the results with a Venn diagram. We tested the significance of all fractions using the anova.cca () function.

3. RESULTS

3.1. Soil Properties

Exchangeable sodium in topsoil and subsoil of reclamation sites was often below detection limits. In Aspen 3 of 50 samples were above detection limit in topsoil and 11 in subsoil. Spruce had only 1 topsoil sample above detection limit and 12 of 50 in subsoil. Reference had 19 of 50 above detection limit in topsoil and 28 in subsoil. Due to low levels in Aspen and Spruce, sodium was removed from analysis. Salinity is not a concern in these young reclamation sites, due to non-saline parent material. Exchangeable potassium was below detection limit in reclamation sites; Aspen had 14 of 50 topsoil samples below detection limit, Spruce had 15. Reference had 2 topsoil and 5 subsoil samples below detection limit. This parameter was not removed from analysis and samples below detection (0.50) were assigned a value of 0.3. Reference subsoil had 18 of 50 samples below detection for total inorganic carbon (0.050) and carbonate (0.40). This parameter was not removed and samples below detection limit were assigned 0.03 and 0.25, respectively.

Reference had higher total nitrogen and total organic carbon in topsoil and subsoil than Aspen and Spruce, with the reclamation sites more similar to each other than to Reference (Table 2.2). Cation exchange capacity in subsoil was similar among sites, and in topsoil was highest in Reference, followed by Spruce. Available sodium in reclamation sites was lower than Reference; soil sodicity is not a concern (Table 2.2). Aspen had lower exchangeable potassium in topsoil and higher soluble sulfate in topsoil and subsoil than Spruce and Reference. Exchangeable magnesium in all sites was classified as high (> 2.5 meq/100 g) and could contribute to potassium deficiency. Exchangeable calcium in topsoil and subsoil was lower in Reference than reclamation sites (Table 2.2). Higher calcium in reclamation sites was likely due to admixing during soil salvage and placement, as soils in the area have highest calcium, magnesium, and sulfur in the B horizon (Table A.1). Saturation percentage in subsoil was similar among sites, and highest in Reference topsoil (96.9 %); reclamation sites were lower (Spruce 85.2 %, Aspen 83.9 %), likely due to lower soil organic matter and development of soil structure and pore space (Table 2.2).

3.2. Vegetation Cover And Community Composition

Tree and shrub cover were greatest in Reference; native and non-native forb and grass cover were greatest in reclamation sites (Figure 2.3A). Reference vegetation community composition was consistent among years. In Aspen, non-native forb cover decreased in 2018 remaining low in 2019; native grass cover increased in 2019; tree, shrub, and weed cover were consistent among study years (Figure 2.3B). In Aspen greatest cover was non-native forbs and grasses in 2017; trees and non-native grasses in 2018; native grasses followed by trees, shrubs, and non-native grasses in 2019 (Figure 2.3). In Spruce non-native grasses had greatest cover each year; tree and shrub cover increased over time and non-native forb and native grass cover decreased (Figure 2.3). Dandelion (*Taraxacum officinale*) was the most common weed at, with 5 times higher cover in reclamation sites than Reference, and highest in Aspen. Kentucky bluegrass (*Poa pratensis*) was the most common non-native grass; with 15 and 21 times higher cover in Aspen and Spruce than Reference, respectively. Reference had the highest woody cover, then Aspen.

Differences in vegetation health and ground cover among sites were small (Table 2.3). Vegetation health across all sites remained relatively stable, with scores never exceeding 1.5 on a scale of 1 to 5, with 1 being healthy and 5 being dead, indicating overall plant health. Ground cover was lower in Reference than reclamation sites, as expected in a forested area versus reclamation sites that have higher herbaceous vegetation cover. Spruce exhibited lower ground cover than Aspen. Both ground cover and vegetation health showed little variation over the sampling years.

3.3. Mantel Correlations

Vegetation dissimilarity was significantly positively correlated with soil property dissimilarities (Mantel R = 0.609, p = 0.001) and with soil invertebrate taxa abundance dissimilarity (R = 0.644, p = 0.001). Correlation between soil properties dissimilarities and soil invertebrate taxa abundance dissimilarity was significant (R = 0.561, p = 0.001). Correlation between vegetation community dissimilarity and soil properties dissimilarity, while controlling for the effects of soil invertebrate taxa abundance, was significant, but correlational strength was decreased (R =

0.279, p = 0.001). Correlation was positive and significant between vegetation community dissimilarity and soil invertebrate taxa abundance dissimilarity, while controlling for the effects of soil properties dissimilarity (R = 0.448, p = 0.001). Correlation between soil properties dissimilarity and soil invertebrate taxa abundance dissimilarity, while controlling for the vegetation community effects, was not significant (R = 0.047, p = 0.121).

3.4. Unconstrained Ordination

3.4.1. Soil properties

Unconstrained ordination of soil properties showed separation of reference and reclamation sites with topsoil and subsoil separate (Figure 2.4). Combining them created significant overlap in 95 % confidence interval ellipses between reclamation sites and Reference; Aspen and Spruce did not overlap (Figure 2.4A). NMDS two-dimensional solution final stress was 0.106, very strong non-metric (R^2 = 0.989) and linear (R^2 = 0.959) fits. The best solution was not repeated after 100 tries. Soil vectors with strong correlation to the ordination (Pearson R > 0.6) were soluble magnesium (R = 0.86), calcium (R = 0.83), sulfate (R = 0.75), and potassium (R = 0.61); electrical conductivity (R = 0.76), saturation percentage (R = 0.70), exchangeable calcium (R = 0.70), and cation exchange capacity (R = 0.63) (Figure 4A). Vectors were not strongly associated with sites, except electrical conductivity and soluble sulfate in Aspen. There were significant site effects (PERMANOVA, df = 2, F = 57.8, R^2 = 0.22, p < 0.001), sampling depth (PERMANOVA, df = 1, F = 97.5, R^2 = 0.19, p < 0.001), and interaction (PERMANOVA, df = 2, F = 5.1, R^2 = 0.02, p < 0.001). Post-hoc testing found there were significant differences in soil properties among the sites (p = 0.003) and the sampling depths (p = 0.001), thus topsoil and subsoil were analyzed separately.

Topsoil NMDS two-dimensional solution had a final stress of 0.0998, the best solution repeated 1 in 22 tries (Bray-Curtis) (non-metric fit, R^2 = 0.99; linear fit, R^2 = 0.97). The 95 % confidence interval ellipse in Reference did not overlap with either reclamation site, but was closer to Spruce than Aspen; Aspen had minor overlap with Spruce (Figure 4B). Soil vectors strongly correlated (Pearson R > 0.6) with the ordination were soluble calcium (R = 0.86) and magnesium (R = 0.85), cation exchange capacity (R = 0.77), saturation percentage (R = 0.77), total carbon (R = 0.77), organic carbon (R = 0.77), inorganic carbon (R = 0.67), and nitrogen (R = 0.76); electrical conductivity (R = 0.73), and exchangeable calcium (R = 0.64) and potassium (R = 0.62). Calcium, electrical conductivity, and total inorganic carbon were associated with reclamation sites; total carbon, organic carbon, and nitrogen; cation exchange capacity, and exchangeable potassium were associated with Reference (Figure 2.4B). Site was significant (PERMANOVA, df = 2, F =

15.2, $R^2 = 0.17$, $p < 0.001$) even after post hoc testing ($p = 0.003$). Dispersion of topsoil properties differed with site (betadisper, ANOVA, df = 2, F = 27.5, $p < 0.001$); post hoc testing found slight significant variation between Reference and Aspen (Tukey's HSD, 95% CI = 0.020 to 0.108, p = 0.002), significant variation between Spruce and Aspen (Tukey's HSD, 95% CI = -0.119 to -0.030, p = 0.0003), and Spruce and Reference (Tukey's HSD, 95% CI = -0.183 to -0.094, $p < 0.001$).

Subsoil NMDS two-dimensional solution final stress was 0.108, the best solution repeated 1 in 83 tries (Bray-Curtis) (non-metric fit, $R^2 = 0.988$; linear fit, $R^2 = 0.957$). The 95 % confidence interval ellipses of each site were clearly separated (Figure 2.4C). Soil vectors strongly correlated (Pearson R > 0.6) with the ordination were electrical conductivity (R = 0.90), soluble calcium (R = 0.899), magnesium (R = 0.88), and sulfate (R = 0.83), and exchangeable calcium (R = 0.71). These vectors selected in topsoil; subsoil had 6 strongly correlated vectors and topsoil 10. Calcium and magnesium were associated with reclamation sites, electrical conductivity and soluble sulfate with Aspen, and no vectors with Reference (Figure 2.4C). Site effects were significant (PERMANOVA, df = 2, F = 54.8, $R^2 = 0.427$, $p < 0.001$) even after p values were adjusted for multiple comparisons (p = 0.003). Dispersion of subsoil properties differed by site (betadisper, ANOVA, df = 2, F = 12.26, p = 0.00002), with subsoil more variable in Aspen than Spruce (Tukey's HSD, 95 % CI = -0.096 to -0.026, p = 0.0002), in Reference than Spruce (95 % CI = -0.101 to -0.031, p = 0.00005), and with no differences in Reference-Aspen (p = 0.939).

Although soil properties are not 'species', per se, we used indicator species analysis (ISA) to determine significant topsoil and subsoil properties for sites. The point-biserial correlation coefficient (r_{pb}) was used to calculate a correlation index to determine soil property indicators, as no appropriate indicators were selected based on aspecificity and sensitivity values. There were many significant (p = 0.001) topsoil and subsoil properties, so only strongly correlated properties (Pearson R < 0.6) were considered (Table 2.4). Three subsoil properties were selected as indicators for Aspen: soluble sulphate, electrical conductivity, and soluble calcium; no soil properties were selected as indicators for Spruce. Soluble chloride in the subsoil was the only indicator selected for Reference. We were most interested in the soil properties selected as indicators for both reclamation sites. Topsoil and subsoil pH were the strongest indicators for Aspen + Spruce, followed by exchangeable calcium and total inorganic carbon in subsoil.

3.4.2. Vegetation composition

Unconstrained ordination separated vegetation community composition between the reclamation sites and Reference, and highlighted vegetation ecological groups and species responsible for the separation. NMDS two-dimensional solution (Bray-Curtis) final stress was 0.204 (non-metric fit, $R^2 = 0.958$; linear fit, $R^2 = 0.82$); the best solution was not repeated after 100 tries. The 95 % confidence interval ellipse of Reference did not overlap reclamation sites, which had some overlap with each other (Figure 2.5). Reference vegetation community was associated with greater shrub cover (Figure 2.5A) and the native forb bunchberry (*Cornus canadensis*, Pearson R = 0.515) (Figure 2.5B). Reclamation sites were associated with greater non-native grass and weed cover (Figure 2.5A). Reclamation sites were associated with the native forb wild strawberry (*Fragaria virginiana*, Pearson R = 0.30), non-native grasses smooth brome (*Bromus inermis,* Pearson R = 0.622*)* and Kentucky bluegrass (*Poa pratensis*, Pearson R = 0.472), the weed dandelion (*Taraxacum officinale*, Pearson R = 0.305) and noxious weed Canada thistle (*Cirsium arvense*, Pearson R = 0.430). Reference was associated with shrubs saskatoon berry (*Amelanchier alnifolia,* Pearson R = 0.416), common snowberry (*Symphoricarpos albus,* Pearson R = 0.465), and prickly rose (*Rosa acicularis,* Pearson R = 0.337). Spruce was associated with *Bromus inermis* and *Cirsium arvense*; Aspen with *Fragaria virginiana* and *Taraxacum officinale* (Figure 2.5B). Correlations for total vegetation, moss, forb, and tree covers, and plant health were low.

Effects were significant for site (PERMANOVA, df = 2, F = 113.96, $R^2 = 0.334$, $p < 0.001$) and sampling year (PERMANOVA, df = 2, F = 3.28, $R^2 = 0.010$, $p < 0.001$), and interactions (PERMANOVA, df = 4, F = 1.98, $R^2 = 0.012$, p = 0.002). Post hoc testing gave significant site differences (p = 0.003). 2018 and 2019 (p = 1.00), and 2017 and 2018 (p = 0.063) were not significantly different; 2017 and 2019 had slight differences (p = 0.018); likely due to precipitation, as June and July 2019 had highest precipitation of all months (Figure A.1). Dispersion of vegetation community composition did not differ depending on site (betadisper, ANOVA, df = 2, F = 4.21, p = 0.016). No differences were found between Reference and Aspen (p = 0.296) and Reference and Spruce (p = 0.338), with slight differences between Spruce and Aspen (p = 0.011).

Indicator species analysis (ISA) selected multiple significant (p = 0.001) indicator plant species, so only strongly correlated species (Pearson R < 0.6), with aspecificity (A) and sensitivity (B) greater than 0.5 were considered (Table 2.5). Aspen indicator species were the shrubs wild black currant (*Ribes americanum*) and red raspberry (*Rubus idaeus*), and native blue joint grass (*Calamagrostis canadensis*). Indicators for Spruce included native forb American vetch (*Vicia americana*), and non-native grasses smooth brome (*Bromus inermis*) and couch grass *(Elytrigia repens*). Reference indicators were four woody shrubs (*Amelanchier alnifolia*, *Rubus pubescens*, *Symphoricarpos albus, Rosa acicularis*) and the native forbs bunchberry (*Cornus canadensis*) and meadow rue (*Thalictrum spp.*). These indicators were the same as species vectors in our ordination, with the addition of dwarf red raspberry. More interesting, were indicators selected for both reclamation sites, Aspen + Spruce: native forbs Goldenrod (*Solidago canadensis*) and strawberry (*Fragaria virginiana*), non native forb alsike clover (*Trifolium hybridum*), non native Kentucky bluegrass (*Poa pratensis*) and Timothy grass (*Phleum pratense*), and weedy dandelion (*Taraxacum officinale*), Canada thistle (*Cirsium arvense*), and perennial sow thistle (*Sonchus arvensis*). Plant species were assigned to Reference +Aspen and Reference + Spruce, but low sensitivity and correlation meant they were not selected as appropriate indicators. ISA of general vegetation parameters only assigned indicators to Aspen + Spruce, all were significant ($p < 0.001$) and had high aspecificity, sensitive, and correlation values (Table 2.6). Not surprisingly, non native forbs, non native grasses, and weeds were vegetation indicators for both reclamation sites.

3.4.3. Soil invertebrate assemblages

Unconstrained ordination separated soil invertebrate assemblages among sites; it separated reclamation sites from Reference, and reclamation sites Aspen and Spruce from each other. The NMDS two-dimensional solution (Bray-Curtis) final stress was 0.189 (non-metric fit, $R^2 = 0.975$; linear fit, $R^2 = 0.87$), the best solution was repeated one time after 20 tries. The 95 % confidence interval ellipses of Reference did not overlap reclamation sites, reclamation sites did not overlap with each other, and Aspen was more similar to Reference than Spruce (Figure 2.6). Soil invertebrate taxa vectors for ants (Formicidae, Pearson R = 0.635), true bugs (Hemiptera, Pearson R = 0.422), beetles (Coleoptera, Pearson R = 0.319), springtails (Collembola, Pearson R = 0.332), spiders and harvestmen (Aranaea and Opiliones, Pearson R = 0.388), and earthworms (Lumbricidae, Pearson R = 0.440) were associated with reclamation sites. Ants, true bugs, and beetles were more associated with Spruce, while springtails, spiders and harvestmen, and earthworms were more associated with Aspen. The soil invertebrate taxa vector for oribatid mites (Oribatida, Pearson R = 0.511) showed a clear association with Reference.

Soil invertebrate groups contributing most to differentiation between reclamation sites and Reference can be categorized by size and mobility. Reclaimed sites had a higher abundance of macrofauna collected with pitfall traps such as beetles, ants, and true bugs; Reference had a higher abundance of mesofauna groups, including mites and springtails, particularly oribatids.

There were significant effects for site (PERMANOVA, df = 2, F = 29.54, $R^2 = 0.327$, $p < 0.001$), sampling year (PERMANOVA, df = 1, F = 11.9, $R^2 = 0.066$, $p < 0.001$), and interactions (PERMANOVA, df = 2, F = 12.76, $R^2 = 0.141$, $p = 0.001$). Post hoc testing found significant differences among sites ($p = 0.003$) and sampling year ($p = 0.006$). Dispersion of soil invertebrate assemblages did not differ depending on site (betadisper, ANOVA, df = 2, F = 0.976, p = 0.381). No differences were found between Reference and Aspen (p = 0.947), Reference and Spruce (p = 0.379), and Spruce and Aspen (p = 0.566).

Indicator species analysis (ISA) was used to determine significant soil invertebrate taxa for Reference and reclamation sites. The point-biserial correlation coefficient (r_{pb}) was used to calculate a correlation index to determine soil invertebrate taxa indicators, as no appropriate indicators were selected based on aspecificity and sensitivity values. No soil invertebrate taxa were selected as indicators for Aspen, ants and true bugs were significant (p = 0.001) indicators for Spruce, and oribatid mites, prostigmatid mites, and collembolans were indicators for Reference (Table 2.7). Beetles were the strongest indicator selected for Aspen + Spruce (Pearson R = 0.543, p = 0.001), along with earthworms (Pearson R = 0.285, p = 0.014). True flies and bees and wasps were indicators for Reference + Spruce.

3.5. Constrained Ordination

Constrained ordination separated vegetation community composition of each site, species contributing to separation, and soil properties driving it. The first four db-RDA axes were all significant (p < 0.001) and accounted for 40.83 % of vegetation-soil variance (Table 2.8A). Sites separated along the first two db-RDA axes; reclamation sites were separated by axis 2, and Reference was split along axis 1 (Figures 2.7, 2.8). Seven topsoil and subsoil properties were significantly correlated (p < 0.05) with vegetation community composition and explained 90.7 % of model variation (Table 2.9A). Subsoil pH and soluble sulfate explained 46.7 % and 8.8 % of the variance, respectively; soluble sodium and potassium in topsoil explained 9.3 % and 4.5 % of the variance, and exchangeable magnesium and potassium explained 4.1 % and 2.9 %.

Site association of soil properties and plant species were similar with unconstrained ordination results. Soluble sulfate in topsoil and subsoil was strongly associated with Aspen, exchangeable calcium and magnesium, and pH in topsoil and subsoil were associated with reclamation sites, and sodium properties in the topsoil were strongly associated with Reference (Figure 2.7).

Eight woody species were associated with Reference, *Ribes americanum* and *Ribes idaeus* with Aspen, spruce (*Picea glauca*) and *Salix* spp*.* with Spruce (Figure 2.8A). Red osier dogwood (*Cornus stolonifera)* was between Aspen and Reference, and an Aspen indicator. *Cornus canadensis* was the only non-woody species associated with Reference (Figure 2.8B). Reclamation sites were associated with native forbs *Fragaria virginiana* and *Solidago canadensis*, and non-native forbs alsike clover (*Trifolium hybridum*) and yellow sweet clover (*Melilotus officinalis*), the latter more with Aspen than Spruce (Figure 2.8B). The non-native grass *Poa pratensis* was associated with reclamation sites and axis 1; *Calamagrostis canadensis* with Aspen, and *Bromus inermis* with Spruce; no grass species were associated with Reference (Figure 2.8C). Perennial sow-thistle (*Sonchus arvensis*) and *Taraxacum officinale* were associated with reclamation sites, Reference had no associated weeds; *Cirsium arvense* was associated with Spruce (Figure 2.8D).

A second constrained ordination separated vegetation community composition of each site and soil invertebrate taxa driving the separation. The first three db-RDA axes were significant ($p < 0.001$), accounting for 29.7 % of vegetation-invertebrate variance (Table 2.8B). Sites separated similar to the vegetation-soil constrained ordination, along the first two db-RDA axes; reclamation sites were separated by axis 2, Reference by axis 1 (Figure 2.9). Oribatid mites, springtails (Collembola), and four insect taxa were significant ($p < 0.05$) (Table 2.9B). Ants (Formicidae) and true bugs (Hemiptera) collected in pitfall traps explained 41.6 % and 7.9 % of variance, respectively (Table 2.9B); both were more strongly associated with Spruce than Aspen (Figure 2.9). Oribatid mites and springtails from soil explained 13.7 % and 8.5 % of the variance (Table 2.9B), and were both associated with Reference and axis 1 (Figure 2.9). Springtails collected in pitfall traps were associated with Spruce and differed greatly from springtails in soils (Figure 2.9).

A final constrained ordination of the vegetation community composition of each site and driving soil properties and invertebrate taxa selected two soil invertebrate taxa for the model, springtails collected in pitfall traps ($p = 0.001$) and mesostigmatid mites from litter ($p = 0.068$) (Figure 2.10A). Significant topsoil and subsoil properties and site associations were similar (Figure 2.10B, C).

3.6. Partitioning Of Variance

Partitioning vegetation community composition variance among soil properties and invertebrate taxa abundance found soil properties explained 19.7 % of variation, major soil invertebrate taxa abundance explained 1.7 %; jointly they explained 23.0 % of variation in vegetation cover and community composition (Figure 2.11A). Significance testing found all testable fractions in variance partitioning statistically significant at $p < 0.001$, except major soil invertebrate taxa abundance which was significant at $p = 0.003$ (Table 2.10). Variation of soil invertebrate taxa abundance explained by vegetation community composition and soil properties was 8.3 % and 3.3 %, respectively; combined they explained 26.8 % (Figure 2.11B). Soil properties without controlling for vegetation composition and vegetation without controlling for soil were significant $p < 0.001$;

individual soil and vegetation fractions were not ($p < 0.05$) (Table 2.10). Vegetation community composition explained 41.1 % of variation of soil properties, soil invertebrate taxa abundance explained 5.1 %; jointly they explained 13.1 % (Figure 2.11C). Soil invertebrate taxa abundance was not significant ($p = 0.277$), while all other testable fractions were ($p < 0.001$) (Table 2.10).

4. DISCUSSION

Successful forest land reclamation returns disturbed areas back to equivalent land capability, including function and community composition similar to undisturbed forest reference areas. Our first objective was to evaluate the effects of two different land reclamation methods, direct soil placement and natural revegetation versus stockpiled soil and planting, on vegetation community composition and soil properties. Our results show that reclamation sites were significantly different from each other and Reference, as expected, given the different soil reclamation and revegetation methods. We were interested in factors where reclamation sites were more similar to each other and dissimilar to Reference. Our ordination plots showed similarity between sites when examining topsoil properties, with Spruce being more similar to Reference; subsoil properties showed greater differentiation by site. Vegetation community composition of reclaimed sites were more similar to each other than Reference, but overall, Aspen was more similar to Reference than Spruce. Oribatid mites and springtails from soil were highest in Reference, while ants, true bugs, and springtails from pitfall traps were most abundant in Spruce. Soil properties explained the largest amount of variation in our data, and led to the greatest separation of sites in multivariate space. Soil invertebrates explained the lowest amount of variation and had the highest residual variation, suggesting factors not addressed in this study were affecting their abundance. Almost all of the variation attributable to soil invertebrates could be explained by soil.

4.1. Soil Properties

We expected Aspen soil properties to be more similar to Reference than Spruce, as it had direct placement of forest litter and Ae mineral soil horizon. Yet topsoil and subsoil properties of the two reclaimed sites were more similar to each other than to Reference. This suggests directly placed cover soil does not provide obvious soil property benefits comparable to those from revegetation. Similarly, Howell et al. (2017) found soil nutrient supply rates were more similar between reclamation sites with forest floor mineral mix or peat mineral mix cover soil, rather than undisturbed benchmark soils. The effects of topographic microsites and soil amendments on native grass and forb establishment in southern Alberta grassland reclamation sites found reclaimed site conditions and plant species played a larger role in native grass and forb establishment than soil reclamation methods and amendments (Naeth et al., 2018). Reclamation or site history, explains why reclaimed soils with different amendments and methods respond similarly, and differently from natural forest reference sites (Hahn and Quideau, 2013).

The higher calcium and magnesium in Aspen and Spruce relative to Reference in our study was similar to reclaimed soils in the Athabasca Oil Sands Region (Rowland et al., 2009; Howell et al., 2017). The lower exchangeable potassium, higher soluble sulfate, and more acidic pH in Aspen soil while total nitrogen and organic carbon, and cation exchange capacity were similar is likely due to soil handling. Soil admixing during salvage and placement disturbs the soil nutrient profile by increasing calcium, magnesium, and sulfur at the surface due to the Luvisolic Bt horizons, and lowering potassium and phosphorus through dilution (Yarmuch, 2003; Lavkulich and Arocena, 2011; Gupta et al., 2015). Over stripping or under stripping can contribute to admixing and loss of the A horizon during soil salvaging (Janz et al., 2019). Reclamation activities on 25 well sites on cultivated lands near Calgary had long-term effects that impacted soil capability and quality, with significantly elevated surface soil pH and decreased total organic carbon, due to subsoil admixing, an alkaline C horizon, and parent material high in inorganic carbon (Janz et al., 2019).

Detailed soil reclamation methods are not known for our reclamation sites and our soil results could be due to different properties in available soil building materials, specifically subsoil and overburden, and associated soil parent materials. Research at reclaimed sites established in 1980 at Battle River Coal Mine near Forestburg, compared six increasing subsoil thickness treatments to determine long-term effects of reclamation methods on soil chemistry, and found subsoil thickness influenced salt movement, transmission zone size, and volume of water supply in the soil profile (Olatuyi and Leskiw, 2014). The Aurora capping study, which aimed to determine appropriate soil cover system designs and capping depths for overburden in oil sands mining soil reclamation, found key mechanisms influencing tree growth included freeze-thaw cycles and water retention, while the influence of lower subsoil materials was unclear (Barber et al., 2015). Soil impacts endured for many decades post reclamation on well sites on cultivated lands in southern Alberta, regardless of certification data and yearly agricultural activities (Janz et al., 2019). Reclaimed soil properties can change over time due to physical and biological processes including freeze-thaw cycles, plant rooting, and settling of soil materials (Huang et al., 2015).

Dominant soil textures differed among our sites and previous research suggests this could explain some of the chemical differences we found. Aspen topsoil and subsoil were predominantly loam, while Spruce soils were predominately clay loam. Reference was a mixture of sandy loam and loam in topsoil, and silt loam, sandy loam, and loam in subsoil. Research quantifying soil water regimes on reclaimed upland slopes of various reclamation prescriptions using peat in the Athabasca Oil Sands Region found water holding capacity differences were likely due to clay, sand, and organic matter (Leatherdale et al., 2012). Soil texture and drainage were important for creating small scale mosaic ecosite conditions, while soil chemical properties did not contribute to differentiating reclaimed ecosites (Thiffault et al., 2017). Tree seedling growth in reclaimed forest sites with fine soils in northern Alberta was twice that of sites reclaimed with coarse soils, but nutrient supply rates and soil water potential had reduced availability in fine soils (Merlin et al., 2019). Over 10 years coarse textured reclaimed soils had 30 to 60 % maximum functional capacity, and fine had 40 to 60 % (Ojekanmi et al., 2020).

4.2. Vegetation Community Composition

Use of stockpiled versus direct placed topsoil is known to affect plant community development in reclaimed sites. This was evident with our Spruce site which received stockpiled soil, and had greater cover of noxious weeds, non-native grasses, and moss cover, while Aspen with directly placed soil had greater cover of native grasses, forbs, and shrubs, and highest overall vegetation cover. Our results echo other research showing direct soil placement is preferable over stockpiling to reduce negative effects on soil quality and seed bank. Stockpiling reduces seed viability and germination and rhizome emergence for many plant species in oil sands mining forest reclamation (Mackenzie and Naeth, 2019). At Coal Valley Mine south in Alberta, reclamation sites with stockpiled or directly placed cover soil had no significant associations between native species cover, proximity to native vegetation, proportion of exposed mineral matter, or agronomic plant cover (Strong, 2000). There was no significant difference between native species abundance and cover soil source (direct placement versus stockpiling), slope gradient, or stand age, but older stands with directly placed cover soil had more shrubs than stockpiled cover soil sites. Oil sands reclamation research in Alberta’s boreal forest found seed bank diversity was higher in stockpiled soils than reference mature forest seed banks, but non-native forb and grass species introduced during soil salvage or storage, contributed greatly to diversity (Buss et al., 2020). Sites with stockpiled soil had more grasses, non-native forbs, and lacked woody species. Researchers found reclaimed sites with stockpiled peat mineral mix soils had lower tree cover and higher graminoid cover relative to direct placement sites, and only two non-native grass species were found, *Agropyron* and *Bromus inermis* (Dhar et al., 2019). Directly placed cover soil increased cover, diversity, and species richness for all plants except graminoids (Dhar et al., 2019). Indicator species analysis found stockpiling led to communities dominated by annual forbs and grasses.

Indicator species selection of two shrubs and two grasses (native, non-native) for Aspen relative to two native forbs, three non-native grasses, one weed, and one noxious weed for Spruce suggests Aspen is at a later successional stage. Aspen had lower weed and noxious weed cover than Spruce; cover was significantly higher than Reference indicating weed and non-native species management is a site wide issue. Previous research at the Genesee mine study area investigating soil compaction and below ground competition, recorded *Bromus inermis* to a height of 1.3 m and complete ground cover after one growing season (Bockstette et al., 2017).

Similar to our study, *Taraxacum officinale*, dandelion, was the most common non-native species at 68 reclaimed mine sites at Coal Valley Mine aged 3 to 19 years and reclaimed with stockpiled or directly placed cover soil (Strong, 2000). Lupardus et al (2019) found dandelion was the strongest indicator species for 30 reclaimed well sites throughout the boreal forest in Alberta. Dandelion was strongly correlated with non-native grass *Phleum pratense*, non-native forb *Trifolium hybridum*, native grass *Agropyron scabra* and forb *Vicia americana*, and the noxious weed *Cirsium arvense* (Lupardus et al., 2019). Forest reference indicator species found by Lupardus et al (2019) included the native shrub *Cornus canadensis*, which was strongly correlated with four other shrubs, (*Lonicera involucrate*, *Viburnum edule*, *Rosa acicularis*, *Rubus pubescens*), and three native forbs (*Mitella nuda*, *Petasites palmatus*, *Aster ciliolatus*). Although our sites had different tree species, these were not strong indicators; four shrub and two native forb species were selected as indicators for Reference, indicating the importance of establishing understory vegetation communities, particularly with native shrubs in forested reclamation sites. Forest reclamation sites characterized by non-native grasses and forbs and weed species, as seen in Spruce, are found in a variety of reclamation methods, disturbance types, and locations throughout Alberta. Non-native species may be an indicator of ecological impairment in some systems (Baah-Acheamfour et al., 2022). Prioritizing direct placement over stockpiling cover soil can promote diversity and cover of native species, as seen in Aspen. While increased forest canopy cover is an important reclamation goal, establishing an understory community with native shrubs and forbs is important for successful forest reclamation.

Revegetation methods can affect reclamation. Spruce was actively revegetated with a grass and forb seed mix and *Picea glauca* seedlings, while Aspen revegetation was through direct soil placement, plant emergence from the seedbank and colonization of nearby species. Competition among plant species is expected in land reclamation sites; however, the type of competitive pressures can vary by site depending on reclamation methods and ongoing management. Reclaimed systems can be characterized and dominated by non-native species, as seen with a *Melilotus officinalis* vegetation community 26 to 34 years post-reclamation (Shaughnessy et al., 2022). Early colonization by weeds and other invasive species before tree establishment, can result in a leaf litter layer that suppresses soil evaporation in reclaimed sites (Huang et al., 2015).

Previous research at Genesee found that restricted rooting space in reclaimed sites caused by soil compaction and below ground competition with herbaceous species affected aspen seedling growth (Bockstette et al., 2017). Herbaceous ground cover and root activity partially mitigated soil compaction, but competitive vegetation, such as grasses, negatively affected tree establishment and growth which altered reclamation trajectories and functional forest development. Competing vegetation was the dominant driver of aspen seedling performance. Smaller pine seedlings were associated with higher colonizing vegetation cover, and higher available soil water, likely because colonizing vegetation has lower water demand than planted seedlings (Merlin et al., 2019). Stand growth parameters in reclaimed trembling aspen (*Populus tremuloides*) sites were correlated with soil pH and CEC, and in *Picea glauca* stands were correlated with forest litter carbon and bulk density, and mineral soil bulk density (Ojekanmi et al., 2020). Site specific differences between Aspen and Spruce, such as revegetation technique, topsoil quantity and quality, subsoil, soil amendments, surrounding vegetation, and proximity to disturbance, surface water bodies, and paved and gravel roads, could account for the largest amount of variance among sites.

4.3. Soil Invertebrates

Our NMDS results of soil invertebrate taxa reveal clear site separation, indicating dissimilarity between Reference and both reclamation sites. Crucially, these indicators not only differentiate between reference and reclaimed sites but between the two reclaimed sites, Aspen and Spruce, reflecting the expected differences in reclamation methods. Soil invertebrate assemblages in Aspen and Spruce are distinct from each other, with Aspen more closely resembling Reference, aligning with reclamation expectations. While vegetation communities failed to provide sufficient separation between Aspen and Spruce, and soil properties did not reflect the closer similarity of Aspen to Reference, soil invertebrate assemblages successfully captured these site specific differences, demonstrating sensitivity and value as ecological indicators.

Our db-RDA results had low explanatory power and showed weak support for specific soil invertebrate groups as indicators differentiating reclaimed and undisturbed forest sites. Constrained ordination analyzing vegetation community composition and soil invertebrate taxa abundance singled out oribatid mites and springtails in topsoil (0 to 15 cm) as soil invertebrate taxa associated with Reference. Springtails in pitfall traps were the only significant metric in the final constrained ordination analysis of vegetation community composition with a combined dataset of soil properties and soil invertebrate taxa abundance. However, none of these taxa explained a large amount of variation in the other datasets. Mite and springtail populations are primarily regulated by climate and food availability, and to a lesser extent, predation (Ferguson and Joly, 2002). Small scale variation in soil properties strongly influence spatial distribution of soil mite and microbial species within habitats (Nielsen et al., 2012). Meta-analysis compared natural variation in oribatid mite community structure with deviations associated with three human disturbance types (agricultural and forest management), and concluded oribatid mite assemblages are effective community level indicators when comparing disturbed and undisturbed sites since natural community variation was lower than changes after human disturbance (Gergócs and Hufnagel, 2017). Our study provides only weak support for the effectiveness of oribatid mite community structure as an indicator of reclamation success.

4.4. Reclamation Indicators

Our third objective was to determine which reclamation indicators, vegetation, soil, or soil invertebrates, best captured variability and could be used in reclamation monitoring. Vegetation composition and soil properties are known to influence below ground mite and microbial communities, although the relative importance of each factor depends on habitat and organism type (Nielsen et al., 2012). We found vegetation community composition and soil properties separately accounted for only 3.3 % and 8.3% of the variation in soil invertebrate taxa abundance respectively, but together accounted for 26.8 %. Soil invertebrates did not account for a large amount of variation in vegetation community composition or in soil properties. Our Mantel correlation results showed that when controlling for the effects of soil invertebrates, the correlational strength between vegetation cover and soil properties decreased. The correlational strength between vegetation cover and soil invertebrates also decreased when controlling for the effects of soil properties, but not as significantly. Adding soil invertebrates into reclamation monitoring can enhance our understanding of disturbance recovery and succession. Our results suggest vegetation assessments based on species cover are sufficient to assess site and soil property variances.

Our results are site specific and small scale, so more research is needed to understand the soil invertebrate role in reclaimed systems. A national scale soil mesofauna assessment was efficient, cost effective, did not add considerably to sampling effort of a monitoring program, and showed simple and standardized sampling methods of soil mesofauna can provide strong conclusions (George et al., 2017). Some hurdles when assessing arthropods as ecological indicators are difficult species identification, inherent bias in sampling methods, seasonal population variation, and spatial robustness, all needing consideration when developing scientific and ecological basis for indicators in reclamation monitoring (Langor and Spence, 2006). Relationships between soil and forest vegetation in the boreal forest are site specific, tree species dependent, and need to account for effects of time and other stand influencing factors (Ojekanmi et al., 2020). Economic considerations are important when choosing appropriate indicators, but evidence based ecological indicators reflecting ecosystem health, function, and resilience need to be included.

Assessing vegetation, soil, and soil invertebrate parameters as suitable indicators for reclamation monitoring showed there were not a few select parameters that can be easily used to explain reclamation or reference sites. This was apparent when assessing the variance that individual parameters explained in our distance based redundancy analysis; the first one or two parameters explained a majority of variance, while remaining parameters explained small percentages. No single metric accounted for the majority of variation in our data; however, several metrics explained significant variation, were highly correlated with axes distinguishing between sites, and were good proxies for site state. We recommend a set of indicators that effectively capture the most variation across sites and reflect shifts that may be more challenging to measure. Key indicators include non-native plant cover, shrub cover, soil pH, cover of weedy species, and abundance of beetles and ants (reclaimed invertebrate indicators), as well as oribatid mites and springtails (reference invertebrate indicators). Ecosystems are complicated and many components come together to influence them. Measuring ecosystem recovery on industrially disturbed sites is difficult, and in cultivated reclamation sites change is slow and activities may reset invertebrate community succession, thus researchers found no single approach was feasible to assess soil quality (Lupardus et al., 2021). Ojekanmi et al. (2020) suggest a soil quality index should include all biological, chemical, and physical indicators of critical process and functions to best represent reclaimed forest stand productivity.

Determining which soil or vegetation parameters were most important to be monitored was difficult. Sometimes total organic carbon was important; total nitrogen was important for all soil (0 to 30 cm); and electrical conductivity was important for soil invertebrates but not vegetation. Reclamation methods have varying effects on soil properties, vegetation composition, and soil invertebrate taxa abundances, as reflected in our indicator species analysis since no indicator was selected for both Aspen and Spruce. Even broad functional vegetation categories of non-native forbs and grasses were not selected for both reclamation sites. Thus when selecting indicators, forest reference sites can act as benchmarks for parameters we want to measure and through reclamation activities eventually return to equivalent capability and function. A major challenge for managing reclaimed environments is ensuring integrity of essential landforms and ecological functioning in reclaimed ecosystems (Audet et al., 2015). Maintaining function and predicting response in boreal forest requires greater understanding of properties and mechanisms responsible for changes in soil over time with and without disturbance (Kishchuk et al., 2014).

Given the successional trajectory of most boreal forests, soil conditions most impacted are related to soil organic matter, nitrogen, pH, cation exchange capacity, and exchangeable cations (Kishchuk et al., 2014). Ojekanmi et al. (2020) found appropriate soil quality indicators to adequately predict forest productivity in reclaimed sites were soil organic carbon and nitrogen to best represent nutrient cycling and transformation processes; soil pH and cation exchange capacity reflect biomass productivity and plant nutrition; and soil texture and bulk density control water retention, transmission, and indirect flow of resources between soil and vegetation.

Olatuyi and Leskiw (2014) studying long-term effects of reclamation prescriptions on soil chemistry at Battle River Coal Mine near Forestburg Alberta, found topsoil (0 to 15 cm) pH was 6.8, and increased with depth to 7.3 in upper subsoil (15 to 40 cm) and 7.7 after 40 cm. Soil chemical properties were correlated with tree cover, and indicators based on vegetation composition could be sufficient to account for biogeochemical cycling and could act as a proxy for soil chemical properties in reclaimed sites (Thiffault et al., 2017). Organic matter content and cation exchange capacity (or base cations) have been identified as the best indicators of mixedwood forest response to variable retention harvesting (Kishchuk et al., 2014). There was no significant difference in total organic carbon between undisturbed forest soils and reclaimed Anthroposols amended with peat, thus using soil organic matter as an indicator of ecosystem recovery would not be appropriate for these reclaimed ecosystems (Norris et al., 2013).

Based on previous research and our results, some ecological parameters explain more variation in reclamation status than others, and these need to guide best management practices and inform reclamation monitoring, regulations, and associated legislation. Our results show it is important to sample topsoil and subsoil separately. Other significant parameters shared between reclamation sites include pH, grass and forb cover, native versus non-native species, and weed cover. Some of our indicators overlap with those supported by previous research. Nine indicators were found useful in assessing ecological recovery, soil bulk density and pH, introduced plant species richness, grass cover, live tree basal area, noxious weeds, canopy cover, downed wood cover, and LFH depth (Lupardus et al., 2019). Long-term assessments of reclamation sites found arrested succession and novel ecosystems, which focused reclamation research on native plant species, increasing shrub diversity and density in the understory, and managing weed and nuisance vegetation (Baah-Acheamfour et al., 2022; Dhar et al., 2020; Hawkes and Gerwing, 2019; Lupardus et al., 2019; Norris et al., 2013).

Residuals in our models were high; our db-RDA results did not explain as much of the variance as expected, which could be because a critical component of this equation has not been included, bacteria and fungi. Including these parameters in the model would likely increase the variation explained and help with understanding soil invertebrates and their function in ecosystem recovery. Vegetation removal associated with surface mining disturbs the rhizosphere; however, resistant propagules can survive in salvaged soils depending on soil handling techniques and time since disturbance (Hankin et al., 2015). To restore pre-disturbance soil conditions, abiotic factors such as pH and to a lesser extent types of soil building materials, are more important to returning soil microflora than spatial structuring (Dimitriu et al., 2010). Differences in soil microbial community composition among reclamation sites is due to amendment type and established vegetation communities (Hahn and Quideau, 2013). Soil microbial community function and structure differed in reclamation sites and undisturbed forest references, with gram negative bacteria dominating reclaimed sites and fungi more abundant in undisturbed sites (Dimitriu et al., 2010). Six, ten, and twelve years after reclamation reconstructed soils only supported 20 % of the total soil microbial biomass in natural forest reference soils, as soil microbial communities may be more resilient to change in natural soils than in reclaimed soils (Hahn and Quideau, 2013).

Our fourth and final objective was to determine if the addition of soil invertebrate abundance, at a broad taxonomic level, to soil and vegetation metrics, changed our results and how reclamation sites were interpreted. When analyzing vegetation community composition, soil properties, and soil invertebrate abundance, we found our results were consistent between constrained and unconstrained ordination, with sites distinctly different from each other. This suggests soil invertebrate taxa abundance was correlated with soil properties and vegetation community composition of reclaimed and forest reference sites. In contrast, reclaimed cultivated land research found soil invertebrate densities and abundances did not accurately indicate soil quality in cultivated reclaimed well sites and reference sites of various ages in southwest Alberta; researchers concluded higher ranking invertebrate taxa were not appropriate for estimating soil biological quality (Lupardus et al., 2021). Use of higher taxonomic levels of mesofauna was informative of relationships in local soil data; further identification could be informative as coarse taxonomic identification of mesofauna is an important complement to national monitoring programs and assessments of soil properties and biodiversity (George et al., 2017). Areas requiring further research include relationships between mesofauna abundances and soil type, and the interaction of mesofauna and soil properties on larger scales (George et al., 2017). Although some of our findings indicate soil invertebrate abundance aligns with soil and vegetation, we still lack understanding of how this correlation relates to soil and vegetation, and whether the low variance explained by factors explored in this study is explainable, or due to stochastic factors unrelated to reclamation practices (e.g., weather).

When considering soil invertebrates as indicators in province wide reclamation monitoring, more research is required. It is important to integrate ecological processes at various temporal and spatial scales in heterogeneous landscapes when assessing ecosystem recovery (Merlin et al., 2019). Research is needed to ensure common monitoring pitfalls such as user error in sample collection, sorting, and coarse taxonomic identification, and cost of additional sampling and identification, can be mitigated and meaningful data collected. Canadian soil and litter associated mite fauna are substantially more diverse than expected and linked to high levels of spatial structure, suggesting environmental differences at regional scales are important in shaping soil mite diversity and future research should examine effects of soil type, land use, and climate on community composition (Young et al., 2019). A study on areas for improvement in long-term environmental monitoring assessing impacts of offshore petroleum industry in Norway on physical and chemical properties of sediments and benthic macroinvertebrates found sources of error: taxonomic resolution, changes in procedures without calibration, use of different laboratories, and outliers and missing values (Ellingsen et al., 2017). Further work on quality control to ensure data collection is consistent and data are comparable is required before oribatid mites and collembolans could be considered indicators in the next iteration of reclamation criteria in Alberta.

5. CONCLUSIONS

Soil properties and soil invertebrate taxa provided good site discrimination, with soil properties explaining the highest amount of variability in our data set. The combined use of soil properties and invertebrate taxa could enhance our ability to discriminate between reclamation sites, providing a more comprehensive understanding of reclamation outcomes. Soil properties offer a broad overview of site physical and chemical conditions, while soil invertebrate taxa reflect biological interactions and ecological health. This dual approach allows for a more nuanced assessment of reclamation success, where both abiotic and biotic components are considered.

Selecting suitable indicators for reclamation monitoring remains challenging due to the complexity of ecosystem recovery after disturbance. Our research found that current metrics of soil and vegetation accounted for more variation, and provided similar multivariate separation of our sites as compared to invertebrate metrics. However, the multivariate separation of our sites using soil invertebrate was the only metric to accurately reflect soil reclamation methods, suggesting soil invertebrate metrics would be more sensitive.

Addition of soil invertebrate taxa abundance to reclamation site assessments using vegetation cover and soil properties, did not change the overall interpretation of our results. Soil invertebrate data reinforced our findings and enhanced our understanding, specifically of vegetation species cover. However, in our young reclamation sites, incorporating soil invertebrate indicators into assessments does not provide an additional benefit, and vegetation assessments of cover by species appear to be sufficient. But in particular sensitive reclamation areas, or end land use objectives that prioritize restoration, soil invertebrate metrics should be considered as our results show they are more sensitive than soil properties and vegetation community composition.

We found the soil invertebrate groups contributing most to differentiation between reclaimed and undisturbed forest reference were related to their size and mobility. Reclaimed sites had a higher abundance of macrofauna collected with pitfall traps such as beetles, ants, and true bugs, whereas Reference was characterized by a higher abundance of mesofauna groups, including mites and springtails, particularly oribatid mites. This reflects the slower recovery of soil dwelling mites and springtails in areas with severe soil disturbance, aligning with expectations based on their ecological roles and habitat requirements.

Direct soil placement in Aspen resulted in higher cover of shrubs and native forb species and lower cover of non-native grasses than Spruce with stockpiled soil. This supports the critical role of the seed bank and native propagules in forest floor litter and topsoil. However, the overall cover of non-native grasses and weed species was similar between Aspen and Spruce, indicating effective management of non-native species is essential to fully benefit from directly placed soil. Vegetation health and ground cover showed no significant differences between reclamation sites. Soil properties, including elevated calcium and magnesium, were similar between reclamation methods, suggesting that direct placement and stockpiling both lead to soil horizon admixing.

6. TABLES AND FIGURES

Table 2.1. Measured soil parameters and associated analytical methods. Saturation extract soluble ions were converted from mg/L to mg/kg: mg/kg = mg/L * (% Saturation / 100 %).

Soil Parameter	Detection Limit	Analysis
Total Nitrogen (%)	0.02	Combustion method, 22.4 (Carter and Gregorich, 2007)
Total Carbon (%)	0.05	Dry combustion, 21.2 (Carter and Gregorich, 2007)
Total Organic Carbon (%)	0.05	
Total Inorganic Carbon (%)	0.05	Empirical standard curve, 20.2 (Carter and Gregorich, 2007)
Inorganic Carbon (carbonates)	0.4	Calculation
Exchangeable Calcium, Magnesium, Potassium, Sodium (meq/100 g)	0.5	Ammonium acetate extraction, 19.4 (Carter and Gregorich, 2007)
Cation Exchange Capacity (meq/100 g)	0.8	Barium chloride method, 18.4 (Carter and Gregorich, 2007)
Exchangeable Sodium Percentage (%)	0.1	Calculation
Saturation Percentage (%)	1	15.2.1, (Carter and Gregorich, 2007)
Soluble Ions (mg/kg)	5	Saturated extraction, 15.2.1 (Carter and Gregorich, 2007)
Calcium		ICP-OES 3120B (APHA, 1998a)
Magnesium		ICP-OES 3120B
Potassium		ICP-OES 3120B
Sodium		ICP-OES 3120B
Chloride		Colourimetry 4500-Cl E (APHA, 1998b)
Sulfur (as SO_4^{-2})		ICP-OES 3120B
Electrical Conductivity (dS/m)	0.1	Saturated extraction, meter, 15.2.2, 15.3.1 (Carter and Gregorich, 2007)
Hydrogen Ion Concentration (pH)	0.1	1:2 Calcium chloride, 3.11 (Carter and Gregorich, 2007)
Sodium Adsorption Ratio	0.1	Calculation, 15.4.4 (Carter and Gregorich, 2007)
Theoretical Gypsum Requirement (t/ha)	0.1	(Ashworth et al., 1999)

Table 2.2. Soil chemical properties of topsoil (0 to 15 cm) and subsoil (15 to 30 cm) collected via Dutch auger. 50 samples collected, per depth, per site: 5 cores per plot across 10 plots. Values are means with standard deviation and standard error in parenthesis.

	Aspen		Spruce		Reference	
	Topsoil	Subsoil	Topsoil	Subsoil	Topsoil	Subsoil
Total Nitrogen	0.36 (0.16, 0.02)	0.18 (0.09, 0.01)	0.35 (0.11, 0.01)	0.15 (0.08, 0.01)	0.52 (0.23, 0.03)	0.28 (0.11, 0.02)
Total Carbon	5.77 (2.87, 0.41)	3.13 (1.36, 0.19)	5.39 (1.67, 0.24)	2.55 (1.03, 0.15)	7.86 (3.74, 0.53)	3.76 (1.46, 0.21)
Total Inorganic Carbon	0.25 (0.10, 0.01)	0.27 (0.10, 0.01)	0.19 (0.03, 0.00)	0.24 (0.04, 0.01)	0.11 (0.05, 0.01)	0.06 (0.03, 0.00)
Total Organic Carbon	5.52 (2.81, 0.40)	2.86 (1.39, 0.20)	5.20 (1.67, 0.24)	2.31 (1.04, 0.15)	7.75 (3.70, 0.52)	3.71 (1.43, 0.20)
Calcium Carbonate	2.05 (0.81, 0.11)	2.22 (0.85, 0.12)	1.61 (0.24, 0.03)	2.01 (0.35, 0.05)	0.95 (0.43, 0.06)	0.54 (0.26, 0.04)
Exchangeable Calcium	28.48 (6.30, 0.89)	23.81 (4.13, 0.58)	29.01 (3.66, 0.52)	24.83 (2.86, 0.40)	19.82 (9.23, 1.31)	12.55 (4.72, 0.67)
Exchangeable Magnesium	6.14 (1.10, 0.16)	5.49 (0.72, 0.10)	8.05 (1.11, 0.16)	6.89 (0.78, 0.11)	6.95 (2.68, 0.38)	4.99 (1.66, 0.23)
Exchangeable Potassium	0.86 (0.22, 0.03)	0.57 (0.21, 0.03)	1.33 (0.38, 0.05)	0.59 (0.26, 0.04)	1.59 (0.85, 0.12)	1.10 (0.60, 0.09)
Exchangeable Sodium	< 0.50	< 0.50	< 0.50	< 0.50	1.47 (0.145)	0.933 (0.058)
Cation Exchange Capacity	33.52 (9.36, 1.32)	24.57 (5.36, 0.76)	35.19 (5.90, 0.83)	25.07 (4.80, 0.68)	40.53 (17.88, 2.53)	25.51 (9.12, 1.29)
Exchangeable Sodium %	-	-	-	-	2.91 (0.209)	3.56 (0.216)
Saturation	83.92 (21.63, 3.06)	61.18 (9.86, 1.39)	85.24 (9.84, 1.39)	66.28 (10.21, 1.44)	96.90 (32.08, 4.54)	60.10 (10.82, 1.53)
Soluble Calcium	132.86 (77.91, 11.02)	80.88 (36.24, 5.13)	102.81 (26.36, 3.73)	43.41 (16.67, 2.36)	73.67 (55.98, 7.92)	24.71 (16.13, 2.28)
Soluble Magnesium	30.81 (17.93, 2.54)	19.44 (8.44, 1.19)	29.02 (7.74, 1.09)	11.93 (4.19, 0.59)	26.14 (18.53, 2.62)	8.76 (5.37, 0.76)
Soluble Potassium	18.63 (12.99, 1.84)	8.15 (4.25, 0.60)	20.23 (10.00, 1.41)	5.03 (2.66, 0.38)	22.89 (19.57, 2.77)	7.11 (5.54, 0.78)
Soluble Sodium	22.41 (15.93, 2.25)	24.38 (10.09, 1.43)	17.58 (7.93, 1.12)	20.20 (8.34, 1.18)	49.55 (56.30, 7.96)	29.66 (21.79, 3.08)
Soluble Chloride	12.40 (5.82, 0.82)	6.09 (2.37, 0.34)	12.09 (2.87, 0.41)	6.36 (1.85, 0.26)	19.11 (7.97, 1.13)	10.15 (3.07, 0.43)
Soluble Sulphate	136.82 (45.78, 20.62)	132.79 (85.89, 12.15)	29.60 (12.60, 1.78)	20.49 (12.71, 1.80)	59.67 (30.10, 4.26)	39.84 (22.01, 3.11)
Electrical Conductivity	0.95 (0.29, 0.04)	0.90 (0.27, 0.04)	0.86 (0.15, 0.02)	0.53 (0.11, 0.02)	0.63 (0.36, 0.05)	0.45 (0.23, 0.03)
Hydrogen Ion Concentration (pH)	6.60 (0.27, 0.04)	6.95 (0.23, 0.03)	7.01 (0.20, 0.03)	7.32 (0.16, 0.02)	5.28 (0.65, 0.09)	5.01 (0.64, 0.09)
Sodium Adsorption Ratio	0.48 (0.18, 0.03)	0.81 (0.26, 0.04)	0.44 (0.21, 0.03)	0.91 (0.40, 0.06)	1.24 (0.91, 0.13)	1.73 (1.00, 0.14)
Theoretical Gypsum	< 0.10	< 0.10	< 0.10	< 0.10	< 0.10	< 0.10

Table 2.3. Ground cover and vegetation health assessed annually, in mid July, from 2017 to 2019. Five vegetation assessments were completed at each plot, with 10 plots per site, for a total of 50 vegetation assessments per site, per sampling year. Values are means with standard error of the mean in parenthesis.

	2017		2018		2019	
	Ground Cover	Vegetation Health	Ground Cover	Vegetation Health	Ground Cover	Vegetation Health
Aspen	0.99 (0.01)	1.26 (0.06)	0.96 (0.01)	1.15 (0.04)	0.96 (0.01)	1.16 (0.05)
Spruce	0.92 (0.01)	1.09 (0.04)	0.88 (0.02)	1.13 (0.04)	0.89 (0.02)	1.21 (0.04)
Reference	0.83 (0.02)	1.33 (0.06)	0.83 (0.02)	1.17 (0.04)	0.86 (0.02)	1.14 (0.05)

Table 2.4. Indicator species analysis using the point-biserial correlation coefficient to examine topsoil (0 to 15 cm) and subsoil (15 to 30 cm) properties. Values are soil properties significant at $\alpha = 0.05$, R = Pearson correlation between soil properties and site, ** $p < 0.001$, * $p < 0.05$. Bolded values indicate a strong positive association, correlation > 0.6.

Site	Soil	Parameter	R	P Value
Aspen	Topsoil	Soluble sulphate	0.450	0.001**
		Soluble calcium	0.340	0.001**
	Subsoil	**Soluble sulphate**	**0.683**	**0.001****
		Electrical conductivity	**0.662**	**0.001****
		Soluble calcium	**0.651**	**0.001****
		Soluble magnesium	0.561	0.001**
Spruce	Topsoil	Exchangeable magnesium	0.366	0.001**
	Subsoil	Exchangeable magnesium	0.563	0.001**
		Saturation	0.252	0.006*
Reference	Topsoil	Sodium adsorption ratio	0.563	0.001**
		Soluble chloride	0.482	0.001**
		Total nitrogen	0.420	0.001**
		Soluble sodium	0.381	0.001**
		Total organic carbon	0.370	0.001**
		Total carbon	0.351	0.001**
		Saturation	0.247	0.008*
		Cation exchange capacity	0.235	0.013*
	Subsoil	**Soluble chloride**	**0.601**	**0.001****
		Sodium adsorption ratio	0.545	0.001**
		Exchangeable potassium	0.531	0.001**
		Total nitrogen	0.518	0.001**
		Total organic carbon	0.377	0.001**
		Total carbon	0.317	0.001**
		Soluble sodium	0.231	0.014*
Aspen + Spruce	Topsoil	**pH**	**0.848**	**0.001****
		Total inorganic carbon	0.590	0.001**
		Exchangeable calcium	0.530	0.001**
		Electrical conductivity	0.425	0.001**
	Subsoil	**pH**	**0.920**	**0.001****
		Exchangeable calcium	**0.814**	**0.001****
		Total inorganic carbon	**0.802**	**0.001****
Reference + Aspen	Subsoil	Soluble potassium	0.275	0.002*
Reference + Spruce	Topsoil	Exchangeable potassium	0.451	0.001**

Table 2.5. Indicator species analysis examining plant communities. Values are plant species significant at α = 0.05, listed in order of vegetation functional group and by descending aspecificity (A). Aspecificity is the probability that the surveyed site is part of the target site group, given the plant's presence; positive predictive value. Sensitivity (B) is the probability of finding the plant in sites belonging to the site group. Bolded values indicate a strong positive association, correlation > 0.6. R = Pearson correlation between plant species and site, ** p < 0.001, * p < 0.05.

Site	Vegetation Functional Group	Plant Name	A	B	R	P Value
Aspen	Shrub	*Ribes oxyacanthoides*	0.772	0.160	0.352	0.001**
		Ribes americanum	**0.735**	**0.693**	**0.714**	**0.001****
		Rubus idaeus	**0.504**	**0.807**	**0.638**	**0.001****
	Native forb	*Smilacina stellata*	0.932	0.220	0.453	0.001**
		Heracleum maximum	0.883	0.060	0.230	0.001**
		Viola canadensis	0.768	0.400	0.554	0.001**
		Equisetum arvense	0.727	0.400	0.539	0.001**
		Lathyrus ochroleucus	0.627	0.313	0.443	0.001**
		Aster ciliolatus	0.431	0.267	0.339	0.039*
	Non native forb	*Stellaria media*	1.00	0.073	0.271	0.001**
	Native grass	***Calamagrostis canadensis***	**0.604**	**0.793**	**0.692**	**0.001****
	Noxious weed	*Bromus japonicas*	1.00	0.033	0.183	0.014*
Spruce	Tree	*Picea glauca*	1.00	0.227	0.476	0.001**
		Salix spp.	0.785	0.173	0.369	0.001**
	Native forb	*Aster laevis*	1.00	0.067	0.258	0.001**
		Potentilla norvegica	1.00	0.033	0.183	0.014*
		Achillea millefolium	0.937	0.193	0.426	0.001**
		Geum aleppicum	0.770	0.213	0.405	0.001**
		Vicia americana	**0.765**	**0.587**	**0.670**	**0.001****
		Mentha arvensis	0.722	0.060	0.208	0.009*
	Non native forb	*Thlaspi arvense*	1.00	0.040	0.200	0.004*
		Trifolium pratense	1.00	0.033	0.183	0.007*
	Native grass	*Bromus pumpellianus*	1.00	0.087	0.294	0.001**
		Poa palustris	0.524	0.127	0.258	0.023*
	Non native grass	*Poa compressa*	1.00	0.027	0.163	0.039*
		Bromus inermis	**0.849**	**0.947**	**0.896**	**0.001****
		Elytrigia repens	**0.712**	**0.727**	**0.720**	**0.001****
	Noxious weed	*Silene latifolia*	1.00	0.080	0.283	0.001**
		Alliaria petiolate	1.00	0.067	0.258	0.001**
Reference	Shrub	***Amelanchier alnifolia***	**0.996**	**0.647**	**0.803**	**0.001****
		Viburnum edule	0.975	0.300	0.541	0.001**
		Rubus pubescens	**0.880**	**0.540**	**0.690**	**0.001****
		Lonicera involucrate	0.793	0.173	0.371	0.001**
		Lonicera dioica	0.756	0.427	0.568	0.001**
		Symphoricarpos albus	**0.585**	**0.900**	**0.726**	**0.001****
		Rosa acicularis	**0.471**	**0.987**	**0.682**	**0.001****
	Native forb	***Cornus canadensis***	**1.00**	**0.647**	**0.804**	**0.001****
		Maianthemum canadense	1.00	0.267	0.516	0.001**
		Sanicula marilandica	1.00	0.167	0.408	0.001**
		Mitella nuda	0.994	0.293	0.540	0.001**
		Pyrola asarifolia	0.964	0.340	0.573	0.001**

		Agrimonia striata	0.941	0.100	0.307	0.001**
		Geum triflorum	0.914	0.047	0.207	0.004*
		Moehringia lateriflora	0.903	0.100	0.300	0.001**
		***Thalictrum* spp.**	**0.869**	**0.507**	**0.664**	**0.001****
		Petasites frigidus	0.685	0.287	0.443	0.001**
		Mertensia paniculate	0.655	0.153	0.317	0.001**
		Galium boreale	0.558	0.587	0.572	0.001**
	Native grass	*Bromus carinatus*	1.00	0.200	0.447	0.001**
		Schizachne purpurascens	1.00	0.073	0.271	0.001**
		Bromus ciliates	0.873	0.280	0.494	0.001**
		Carex spp.	0.617	0.133	0.287	0.002*
Aspen + Spruce	Native forb	***Solidago canadensis***	**1.00**	**0.630**	**0.794**	**0.001****
		Fragaria virginiana	**0.880**	**0.683**	**0.775**	**0.001****
		Aster conspicuous	0.955	0.147	0.374	0.001**
		Epilobium spp.	0.981	0.240	0.485	0.001**
	Non native forb	*Melilotus officinalis*	1.00	0.263	0.513	0.001**
		Trifolium hybridum	**0.972**	**0.557**	**0.736**	**0.001****
	Non native grass	***Poa pratensis***	**0.918**	**0.907**	**0.912**	**0.001****
		Phleum pratense	**0.986**	**0.393**	**0.623**	**0.001****
	Weed	***Taraxacum officinale***	**0.858**	**0.750**	**0.802**	**0.001****
		Galeopsis tetrahit	0.899	0.230	0.455	0.001**
	Noxious weed	***Cirsium arvense***	**1.00**	**0.643**	**0.802**	**0.001****
		Sonchus arvensis	**1.00**	**0.557**	**0.746**	**0.001****
Reference + Aspen	Tree	*Populus balsamifera*	1.00	0.110	0.332	0.001**
		Populus tremuloides	0.966	0.187	0.425	0.001**
	Shrub	*Cornus stolonifera*	0.983	0.257	0.502	0.001**
Reference + Spruce	Native grass	*Carex* spp.	0.925	0.090	0.289	0.003*

Table 2.6. Indicator species analysis examining general vegetation parameters. Values are vegetation parameters significant at α = 0.05 and listed by descending aspecificity (A). Aspecificity is the probability that the surveyed site is part of the target site group, given the plant's presence; positive predictive value. Sensitivity (B) is the probability of finding the plant in sites belonging to the site group, ** p < 0.001. All parameters have a strong positive association, correlation > 0.6, therefore no values are bolded.

	Cover Parameter	A	B	R	P Value
Aspen + Spruce	Non native forbs	0.996	0.707	0.839	0.001**
	Non native grasses	0.985	0.993	0.989	0.001**
	Weeds	0.955	0.947	0.951	0.001**
	Noxious weeds	1.00	0.827	0.909	0.001**

Table 2.7. Indicator species analysis using the point-biserial correlation coefficient to examine soil invertebrate assemblages. Values are soil invertebrate taxa significant at α = 0.05, R = Pearson correlation between soil invertebrate taxa and site, ** $p < 0.001$, * $p < 0.05$. Bolded values indicate a strong positive association, correlation > 0.6.

Site	Taxa	R	P Value
Spruce	**Ants (Formicidae)**	**0.670**	**0.001****
	True bugs (Hemiptera)	**0.566**	**0.001****
Reference	**Oribatid mites (Oribatida)**	**0.741**	**0.001****
	Prostigmatid mites (Prostigmata)	0.487	0.001**
	Springtails (Collembola)	0.439	0.001**
Aspen + Spruce	Beetles (Coleoptera)	0.543	0.001**
	Earthworms (Lumbricidae)	0.285	0.014*
Reference + Spruce	Bees and wasps (Apoisidae)	0.317	0.008*
	True flies (Diptera)	0.261	0.037*

Table 2.8. Distance based redundancy analysis (db-RDA) for plant communities. The trace value (sum of all the canonical eigenvalues) and eigenvalues and their contributions to the squared Bray distance for the first 4 axes are given for soil properties and major soil invertebrate taxa abundance; ** axis significant at $p < 0.001$, * $p < 0.05$.

	Axis 1	Axis 2	Axis 3	Axis 4
Physical Properties				
Trace: 21.886				
Eigenvalue**	11.776	2.678	2.005	1.133
Proportion Explained	27.33 %	6.22 %	4.65 %	2.63 %
Cumulative proportion	27.33 %	33.55 %	38.20 %	40.83 %
Soil Invertebrate Taxa Abundance				
Trace: 14.479				
Eigenvalue**	10.037	1.697	1.057	
Proportion Explained	23.29 %	3.94 %	2.45 %	
Cumulative proportion	23.29 %	27.23 %	29.69 %	

Table 2.9. ANOVA testing of proposed db-RDA model of vegetation community composition explained by A) soil properties and B) major soil invertebrate taxa abundance. Presented are degrees of freedom (df) and sum of squares (SS) for the model and residual, and p values indicating significance; ** p < 0.001, * p < 0.05.

A)	df	SS	P Value	B)	df	SS	P Value
Residual	128	21.203		Residual	139	28.610	
Model	21	21.886	0.001**	Model	10	14.479	0.001**
Subsoil pH		10.14	0.001**	Ants		6.018	0.001**
Topsoil soluble sodium		2.025	0.001**	Oribatid mites in topsoil		1.987	0.001**
Subsoil soluble sulphate		1.927	0.001**	Springtails in pitfall traps		1.325	0.001**
Topsoil soluble potassium		0.992	0.001**	Springtails in topsoil		1.225	0.001**
Topsoil exchangeable magnesium		0.894	0.001**	True bugs		1.146	0.001**
Topsoil exchangeable potassium		0.624	0.002*	True flies		0.757	0.002*
Topsoil exchangeable sulphate		0.499	0.002*	Beetles		0.621	0.005*
Topsoil exchangeable calcium		0.488	0.007*	Oribatid mites in litter		0.563	0.007*
Subsoil exchangeable magnesium		0.458	0.010*	Prostigmatid mites in topsoil		0.509	0.019*
Topsoil soluble sodium		0.429	0.012*	Springtails in topsoil		0.328	0.092
Subsoil cation exchange capacity		0.333	0.032*				
Subsoil total nitrogen		0.333	0.048*				
Subsoil total inorganic carbon		0.319	0.043*				
Subsoil exchangeable potassium		0.316	0.040*				
Topsoil pH		0.313	0.051				
Topsoil total carbon		0.308	0.045*				
Topsoil cation exchange capacity		0.298	0.064				
Subsoil total carbon		0.289	0.061				
Topsoil soluble chloride		0.277	0.081				
Subsoil saturation		0.277	0.078				
Subsoil soluble potassium		0.275	0.089				

Table 2.10. Variance partitioning explaining variation of vegetation community composition, soil properties, and soil invertebrate taxa assemblage and abundance. Presented are degrees of freedom (df), coefficient of determination (R^2), and p values indicating significance; ** $p < 0.001$, * $p < 0.05$.

	df	Adjusted R^2	Variance	p value
Vegetation Community Composition				
Total	52	0.443	0.5271	
Soil without controlling for invertebrates **X1**	38	0.426	0.3129	0.001**
Invertebrates without controlling for soil **X2**	14	0.247	0.2123	0.001**
Soil alone **X1|X2**	38	0.197	0.1439	0.001**
Invertebrates alone **X2|X1**	14	0.017	0.0432	0.003*
Residuals		0.557		
Soil Invertebrate Assemblages				
Total	115	0.384	5056.5	
Soil without controlling for vegetation **X1**	38	0.351	2606.7	0.001**
Vegetation without controlling for soil **X2**	77	0.301	3286.3	0.001**
Soil alone **X1|X2**	38	0.083	948.9	0.026*
Vegetation alone **X2|X1**	77	0.033	1628.5	0.043*
Residuals		0.616		
Soil Chemical Properties				
Total	91	0.593	38.00	
Vegetation without controlling for invertebrates **X1**	77	0.541	28.87	0.001**
Invertebrates without controlling for vegetation **X2**	14	0.182	15.21	0.001**
Vegetation alone **X1|X2**	77	0.411	15.98	0.001**
Invertebrates alone **X2|X1**	14	0.051	2.32	0.277
Residuals		0.407		

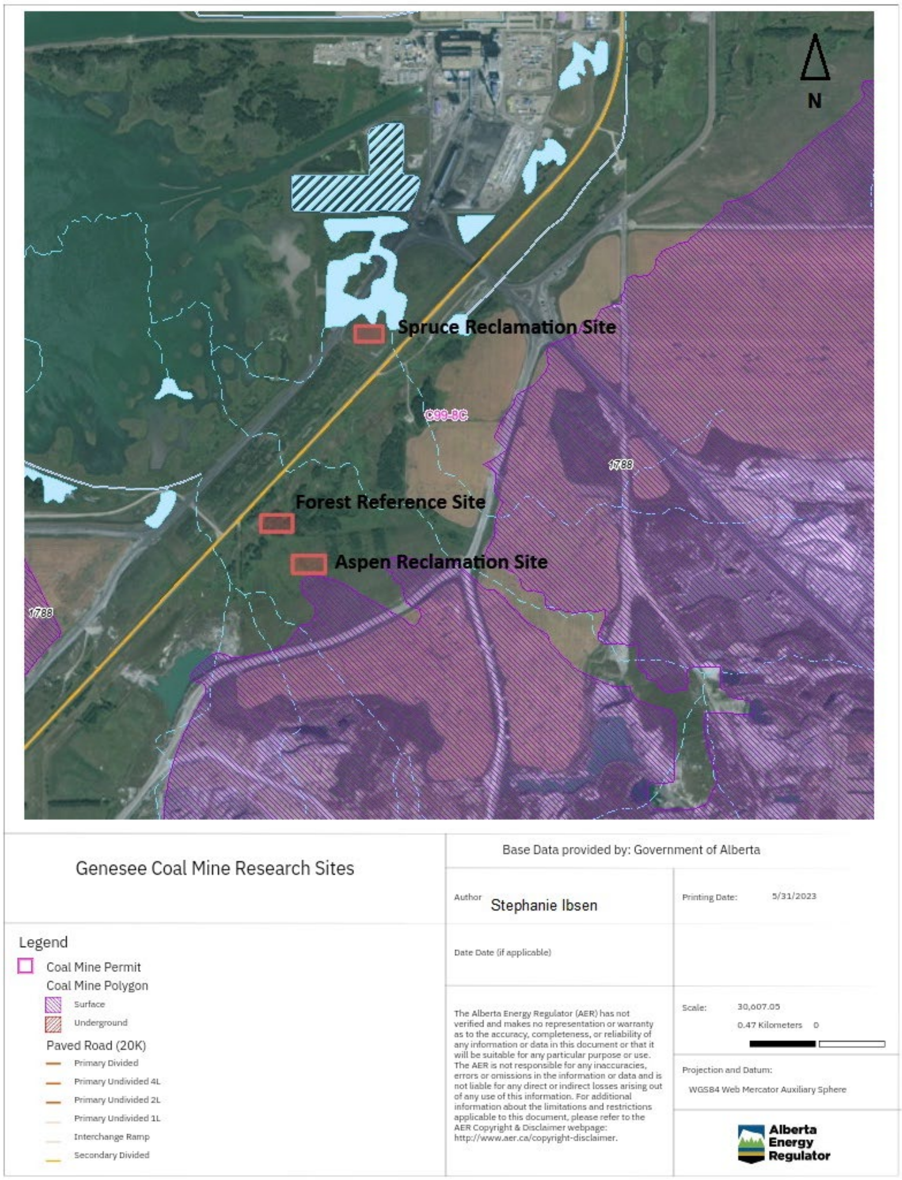


Figure 2.1. Research area at Genesee Coal Mine in western Alberta with overlain research sites, hydrology, and surface mined areas. The spruce reclamation site (Spruce) is on the west side of Highway 770 near the Genesee Generating Station and across from the cooling pond. The other two research sites are located east of the highway, with the aspen reclamation site (Aspen) nested between the forested reference (Reference) and other active reclamation areas. Areas being actively mined and associated haul roads can be seen further south. (Government of Alberta; https://extmapviewer.aer.ca/AERCoalMine/Index.html).

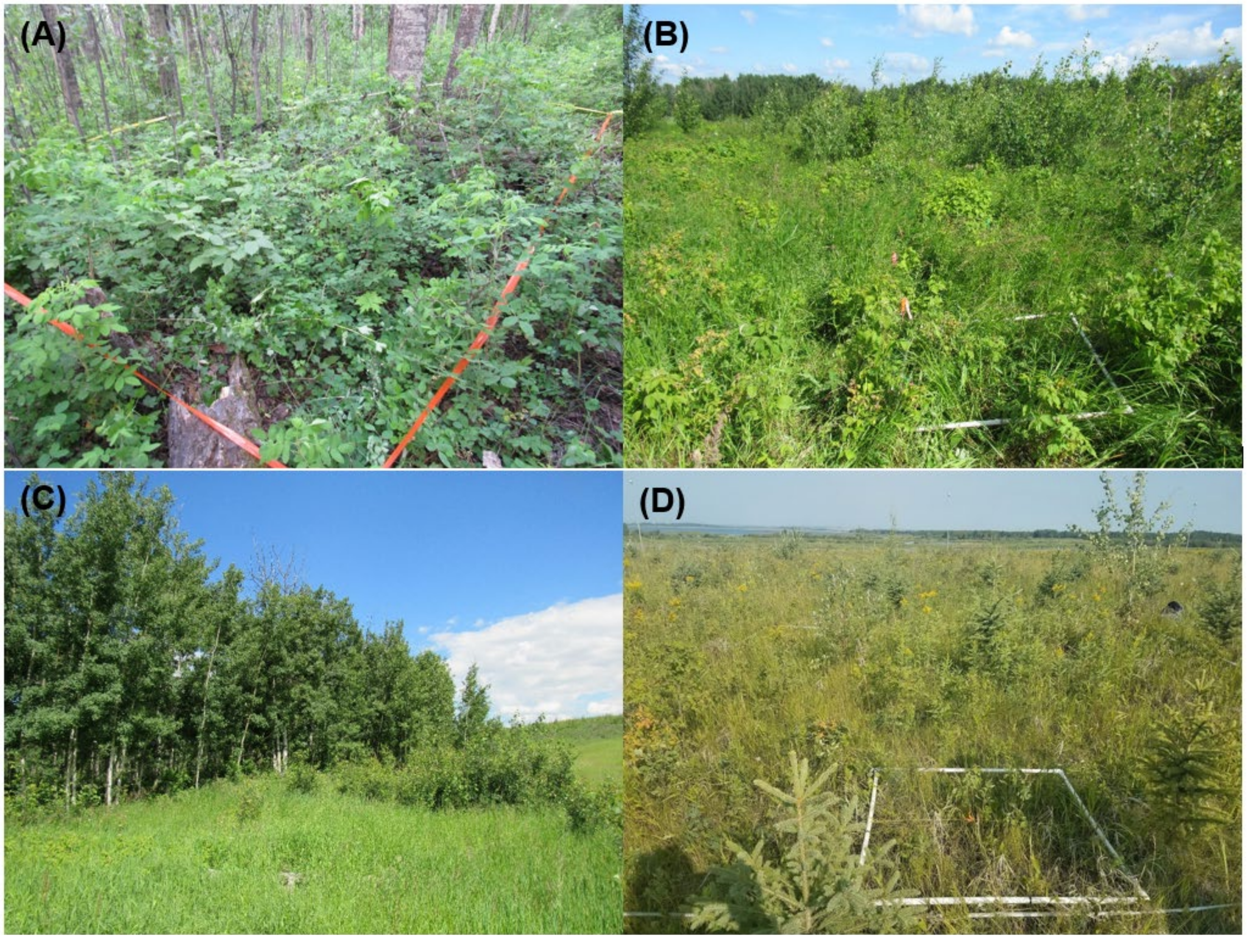
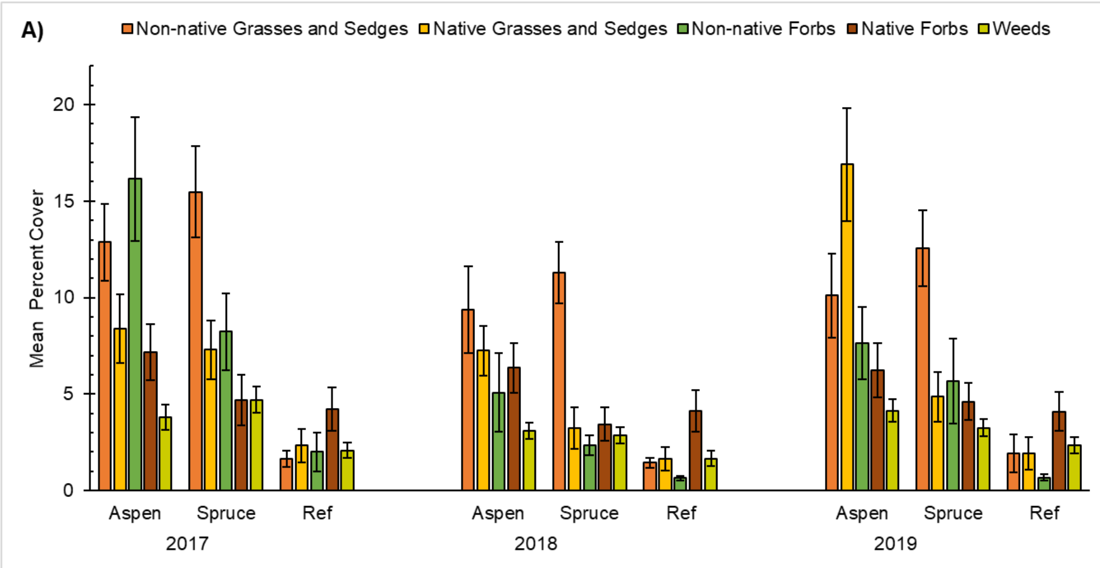


Figure 2.2. Photos showing general vegetation cover, composition, growth stage, and health of (A) Reference, (B) Aspen, (C) transition between Reference and Aspen, and (D) Spruce. Photos taken by Stephanie Ibsen on July 12, 2018.



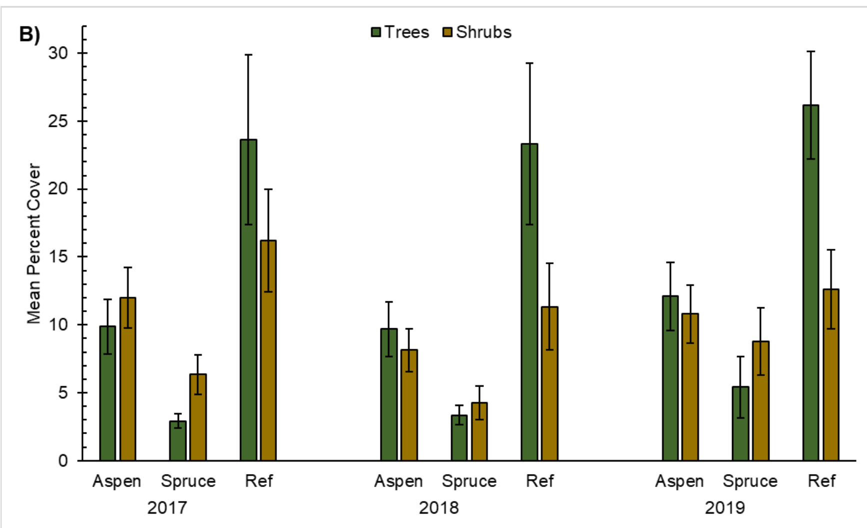


Figure 2.3. Mean percent cover of vegetation functional groups assessed during annual sampling from 2017 to 2019. Displayed are A) herbaceous and grassy vegetation, and B) woody vegetation. Error bars = standard error.

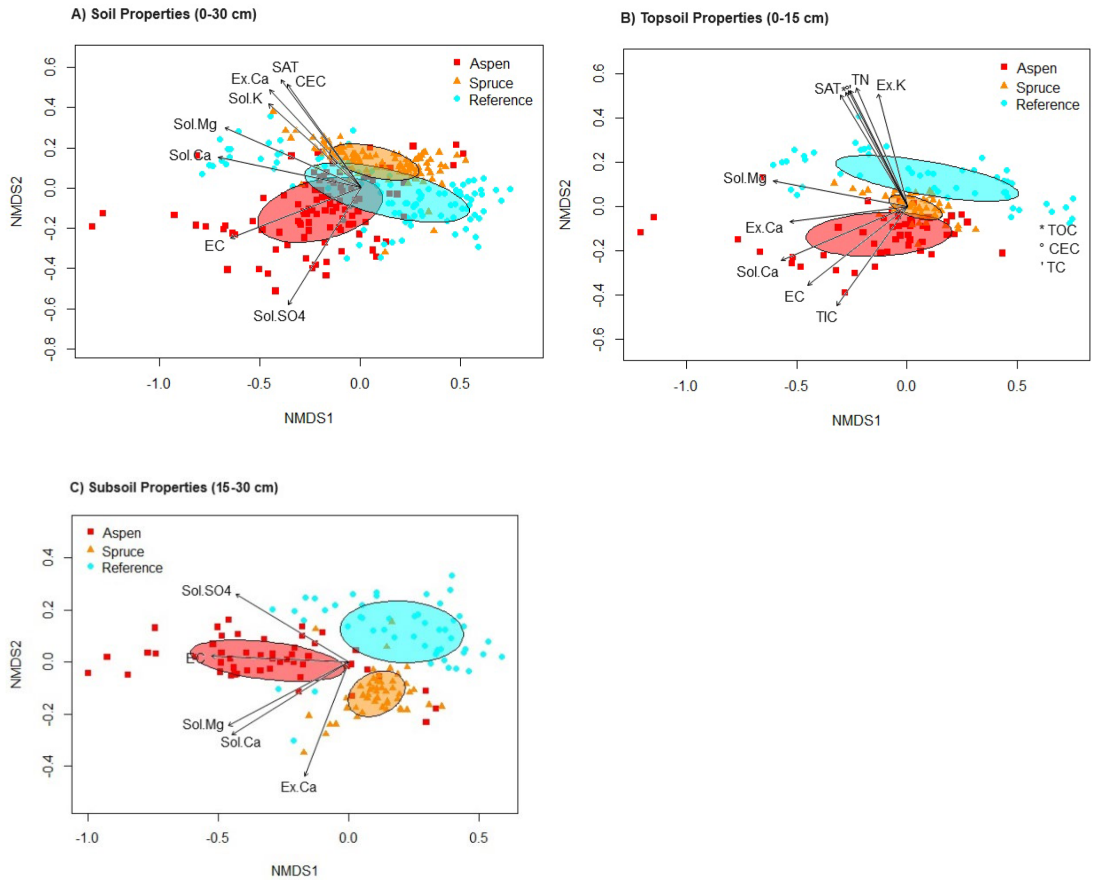


Figure 2.4. Non metric multidimensional scaling (NMDS) ordination with a Bray-Curtis distance measure of A) soil properties (0 to 30 cm), B) topsoil properties (0 to 15 cm), and C) subsoil properties (15 to 30 cm). Ellipses are 95 % confidence intervals and the angle and length of the vectors indicate direction and strength of association with the ordination axis. Soil property vectors with Pearson R > 0.6 and p = 0.001 are displayed.

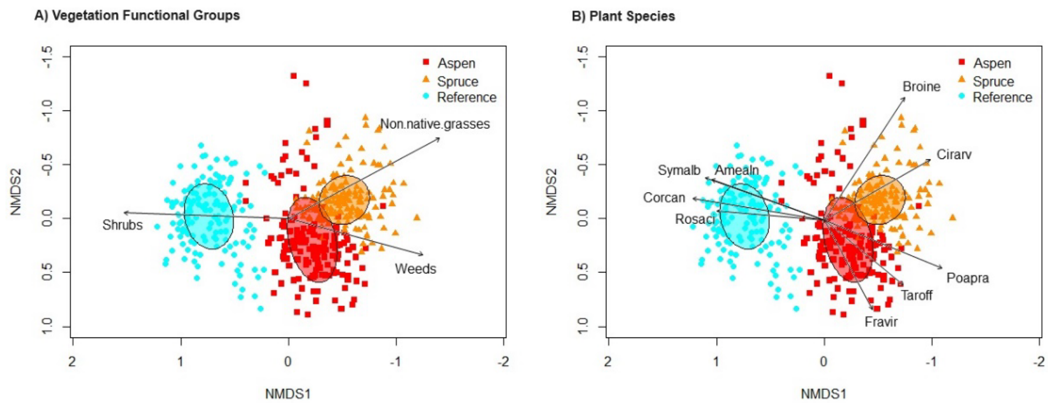


Figure 2.5. Non metric multidimensional scaling (NMDS) ordination with a Bray-Curtis distance measure of vegetation community composition. Ellipses are 95 % confidence intervals and the angle and length of the vectors indicate direction and strength of association with the ordination axis. Displayed are the A) vegetation functional groups and B) plant species; vectors with Pearson R > 0.3 and p = 0.001 are included.

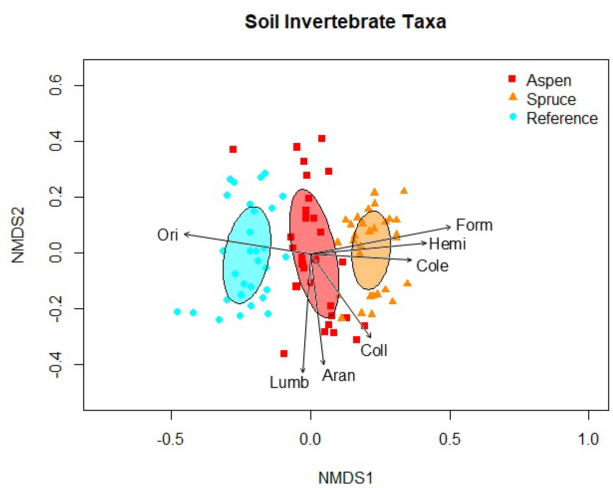


Figure 2.6. Non metric multidimensional scaling (NMDS) ordination with a Bray-Curtis distance measure of soil invertebrate assemblages. Ellipses are 95 % confidence intervals and the angle and length of the vectors indicate direction and strength of association with the ordination axis. Soil invertebrate taxa vectors with Pearson R > 0.3 and p = 0.001 are displayed. Soil invertebrate taxa abbreviations are: oribatid mites (Ori), earthworms (Lumb), spiders and harvestmen (Aran), springtails in pitfall traps (Coll), beetles (Cole), true bugs (Hemi), and ants (Form).

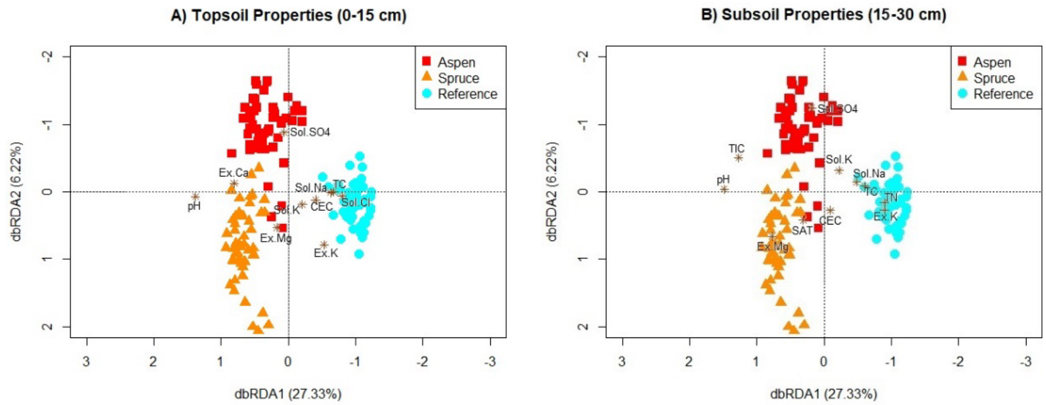


Figure 2.7. Distance based redundancy analysis (db-RDA) of vegetation community composition delineated by research site type with type 2 scaling. Variation explained by each axis is included in parentheses. Key driving factors of the vegetation community composition for A) topsoil properties and B) subsoil properties. All scores were scaled by 2.6 to improve readability.

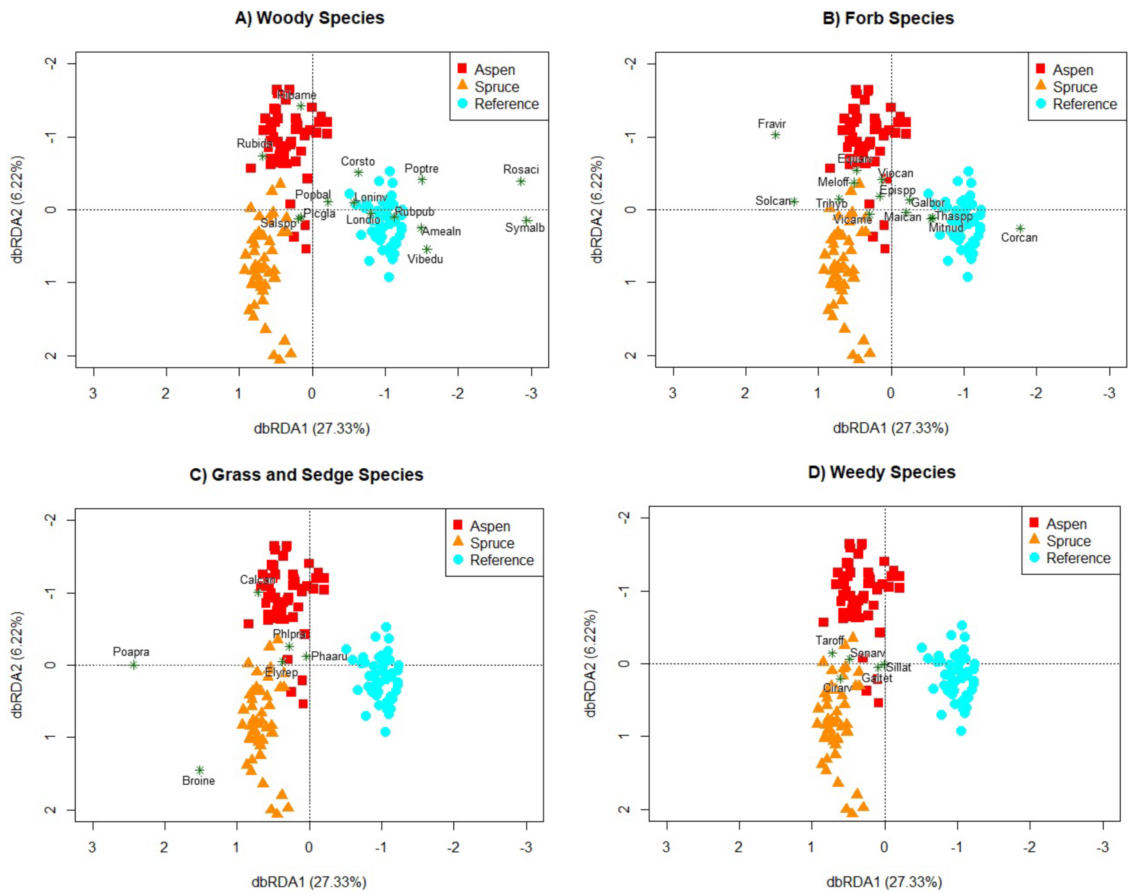


Figure 2.8. Distance based redundancy analysis (db-RDA) of vegetation community composition delineated by research site type and type 2 scaling. Vegetation species (Codes in Table A.4) were separated into major functional groups: A) woody species, B) forbs, C) grasses, D) weedy species. All species scores, except for grasses, were scaled by 1.6 to improve readability.

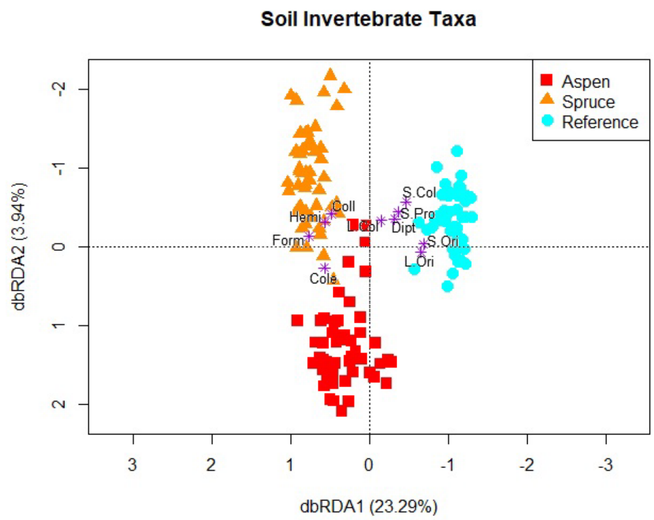


Figure 2.9. Distance based redundancy analysis (db-RDA) of vegetation community composition and major soil invertebrate taxa abundance delineated by research site type and type 2 scaling. Variation explained by each axis is included in parentheses. All taxa scores were scaled by 2.6 to improve readability. Samples were collected from litter (L), soil (S), or using pitfall traps. Soil invertebrate taxa abbreviations are: ants (Form), true bugs (Hemi), true flies (Dipt), beetles (Cole), springtails in pitfall traps (Coll), oribatid mites in topsoil (S.Ori), springtails in topsoil (S.Col), prostigmatid mites in topsoil (S.Pro), oribatid mites in litter (L.Ori) and springtails in litter (L.Col).

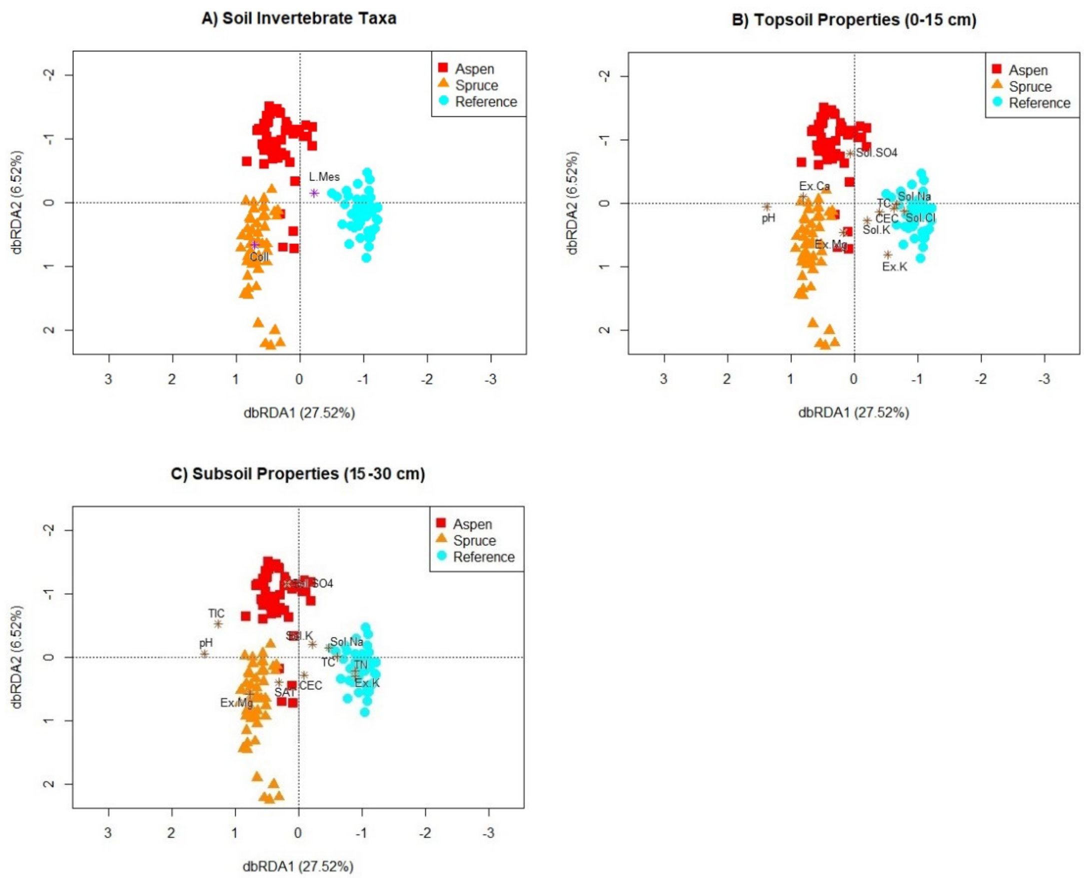


Figure 2.10. Distance based redundancy analysis (db-RDA) of vegetation community composition, major soil invertebrate taxa abundance, and soil properties delineated by research site type and type 2 scaling. Variation explained by each axis is included in parentheses. Key driving factors of vegetation community composition for A) soil invertebrate taxa, B) topsoil properties, and C) subsoil properties. All scores were scaled by 2.6 for readability. Soil invertebrate taxa abbreviations are: springtails in pitfall traps (Coll) and mesostigmatid mites in litter samples (L.Mes).

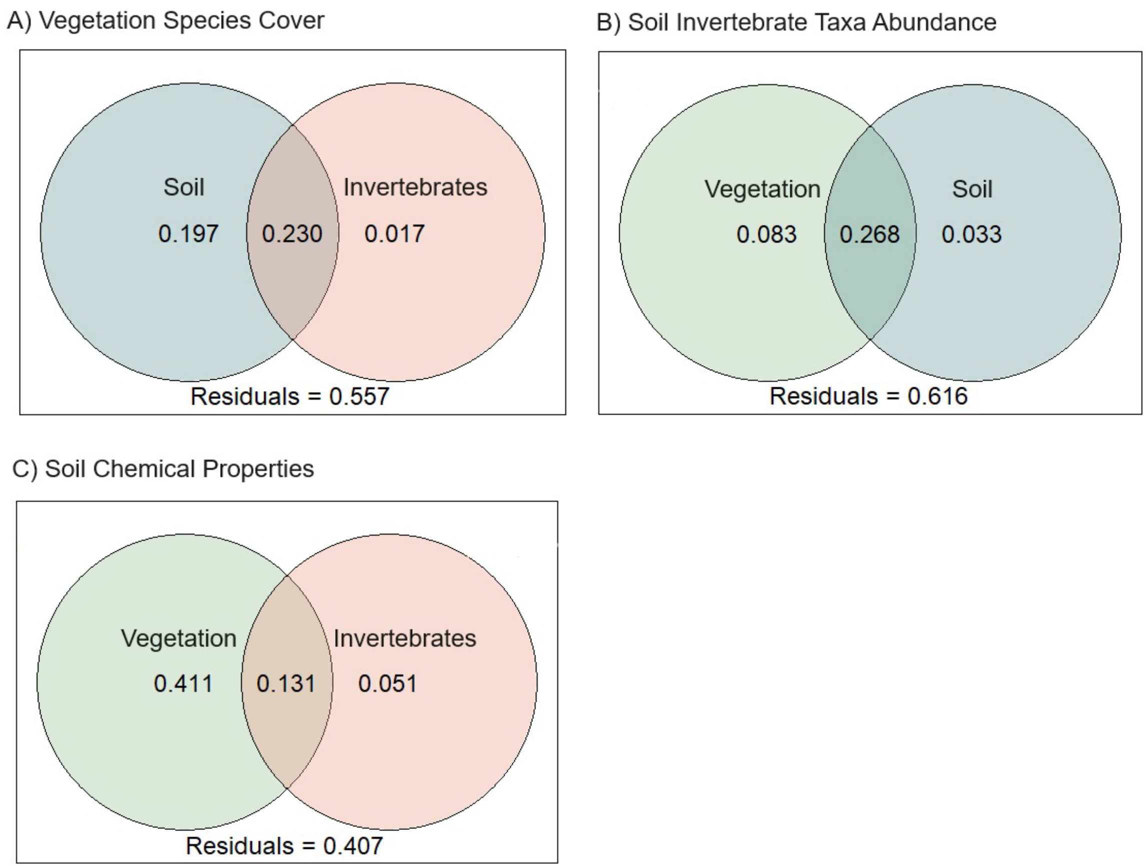


Figure 2.11. Variance partitioning between A) vegetation species composition, B) major soil invertebrate taxa abundance, and C) soil chemical properties. Displayed are adjusted R^2 values which indicate the total variance explained by one variable or a set of variables. Note circles sizes are not correlated with variance explained.

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CHAPTER III. AN ASSESSMENT OF SOIL INVERTEBRATES (MACRO AND MESO FAUNA) AS INDICATORS OF FOREST RECLAMATION

1. INTRODUCTION

Soils provide essential ecosystem services including carbon sequestration, nutrient cycling, climate regulation, and primary production; and diverse soil organisms play a large role in delivering those ecosystem services (Lavelle et al., 2006). Soil fauna can alter soil properties and plant communities, impacting microbial activity and decomposition (Frouz, 2018). Several studies found soil invertebrate diversity and composition are directly linked to ecosystem biodiversity, function, and stability (Majer, 1983; Majer, 1990; Ferguson and Berube, 2004). Disturbances that alter soil invertebrate activity can modify soil function (Blanchart et al., 1997; Barros et al., 2001). Changes in soil invertebrate diversity and species composition can correspond to ecosystem changes and provide information on ecosystem health, complexity, function, and stability; thus, soil invertebrates could be good candidates for assessing impacts of anthropogenic activities (Majer, 1983; Majer, 1990; Ferguson and Berube, 2004). Soil invertebrate taxa have been used as effective, robust, and sensitive indicators of forest management in China (Zhao et al., 2013), restoration after bauxite mining in Australia (Orabi et al., 2010), and arable site management in Argentina (Bedano et al., 2011). Gaps in ecological and taxonomic knowledge can limit soil invertebrate assessment tools (Hawksworth and Ritchie, 1993). Coarser taxonomic identification can overcome difficulties, if it meets the required monitoring objectives (Bedano et al., 2011).

Mites (Acari) and springtails (Collembola) dominate mesofauna in soil and litter and represent up to 95 % of arthropods in samples (Battigelli, 2000; Behan-Pelletier, 2003). Environmental and spatial variables, species interactions, weather, and community structure seasonality influence their distribution patterns (Dwyer et al., 1998; Ferguson and Joly, 2002; Minor et al., 2004; Nielsen et al., 2012; Maaß et al., 2015; Meehan et al., 2018; Wehner et al., 2018). Seasonality of soil invertebrate groups varies with ecology and life strategies (Schenker, 1984; Stamou and Sgardelis, 1989). Reproductive cycles operate on short time scales for some taxa (Ferguson and Joly, 2002); others including many oribatid mites (Oribatida) can take months to years to mature (Norton, 1994). Improving knowledge of spatiotemporal dynamics and soil invertebrate seasonality is important to interpret ecosystem function and processes (Wu and Wang, 2019), especially after mining, where reclamation methods and monitoring are expected to evolve.

Surface mined reclamation sites in the boreal forest of Alberta, Canada, are in a state of arrested succession, recovering more slowly and on different trajectories than forests impacted by fire or logging (Dhar et al. 2018, Hawkes and Gerwing 2019, Lupardus et al. 2019). This is likely due to extensive soil disturbance associated with surface mining and soil reclamation with various soil building materials to create an Anthroposol (Naeth et al., 2012; Hawkes and Gerwing, 2019; Naeth et al., 2023). Factors such as soil cover type, soil stockpiling, grass and canopy cover, hydrologic regimes, soil pH, and litter accumulation have been linked to recovery of forest reclamation sites, while the impact of time since reclamation is unclear (Rowland et al., 2009; Sorenson et al., 2011; Thiffault et al., 2017; Dhar et al., 2018; Lupardus et al., 2019). An ecological recovery assessment of five to twenty-year-old reclaimed well sites in Alberta's boreal forest, revealed that the 2010 reclamation criteria, which incorporated indicators of soil quality, woody stem requirements, and native plant cover, promoted more effective ecosystem recovery compared to the 1995 criteria (Environmental Protection, 1995; Environment and Sustainable Resource Development, 2013; Baah-Acheamfour et al., 2022). These updated standards facilitated better reclamation outcomes on reclaimed forest sites. Updating reclamation criteria with appropriate ecological indicators can more accurately reflect reclamation site recovery; however, more research is needed to determine reclamation indicators that reflect ecosystem function and can be assessed efficiently and effectively (Lupardus et al., 2019).

Soil and litter invertebrate assemblages and dynamics have been studied for various Canadian forestry (Langor and Spence, 2006; Pohl et al., 2007) and agricultural practices (Price and Gordon, 1998; Osler et al., 2008), and more recently in land reclamation research (Battigelli, 2011; Hammond et al., 2018; McAdams et al., 2018; Lupardus et al., 2021; Hammond et al., 2022). For example, an assessment of soil mesofauna in reclaimed and undisturbed forest sites in northern Alberta, found that soil mesofauna were useful for assessing disturbance and management practices, but lacked a single feasible approach due to insufficient baseline data from natural and reclaimed soils of various ages and ecosites (Battigelli, 2011).

To assess soil arthropod communities in reclaimed soil and determine success of reclamation methods, a master list of species and associated environmental variables must be developed (Battigelli, 2011). We have baseline data on response of ground beetles (Carabidae), rove beetles (Staphylinidae), and spiders (Araneae) to soil reclamation methods used in oil sands mine and borrow pit forest reclamation sites in northern Alberta (Hammond et al., 2018, 2022). We also have baseline data on response of springtails (Collembola) and mites (Acari) to soil reclamation methods and time since soil reconstruction in forested oil sands mining reclamation sites (McAdams et al., 2018; Hook, 2019). There are baseline data on all soil invertebrates smaller than 2 mm, at reclaimed well sites on cultivated lands in southern Alberta; however, researchers concluded soil invertebrate densities and abundance, identified at higher taxonomic levels (order, suborder, family) were not appropriate indicators for soil biological quality (Lupardus et al., 2021).

Reclamation of plains coal mines involve removing vegetation one to two years before mining, stripping and salvaging approximately 20 cm of topsoil, remaining subsoil, and overburden (Navus Environmental Inc., 2012). Topsoil and subsoil can be used directly in active reclamation or stockpiled. Subsoil is generally placed to 1 m depth at the final elevation, with large rocks removed and compaction alleviated by deep tillage, creating a level surface for uniform topsoil placement of approximately 18 to 20 cm (McQueen et al., 1991). Forest end land use revegetation methods include to seed with native and or non native grasses, to prevent soil erosion and improve soil conditions, followed by tree seedling planting (Rowland et al., 2009). Environmental variables in monitoring to obtain forest reclamation certification include topography, geotechnical stability, surface and ground water quantity and quality, replaced soil quantity and quality, vegetation composition, wildlife use, and contaminant or waste presence (Alberta Energy Regulator, 2019).

To determine whether soil invertebrates may be appropriate reclamation indicators we assessed above ground (epigaeic), litter dwelling, and below ground soil invertebrates at two forest reclamation sites and an undisturbed forest reference site at a coal mine in central Alberta. Our objectives were to (i) assess soil and litter invertebrate assemblages and abundance; (ii) determine which invertebrate taxa can be used as indicators of undisturbed forest in the research area; and (iii) determine optimal sampling month and methods for soil and litter invertebrates. The ideal invertebrate indicator taxon would consistently differ in reclaimed and reference sites regardless of sample year or month. We hypothesized (i) invertebrate abundance would be lower in reclamation sites and lower with salvaged stockpiled soil than directly placed soil; (ii) invertebrate assemblage would differ in reclamation sites due to soil and litter microhabitat and dominant vegetation; and (iii) invertebrate abundances would peak in September after reproducing in the growing season. These data were used to assess invertebrate taxa most likely to be able to differentiate reclaimed and reference sites, and to provide guidance to reclamation practitioners on which could be pragmatically incorporated into regular reclamation monitoring.

2. MATERIALS AND METHODS

2.1. Research Area And Sites

The study area was located at the Genesee Mine, 70 km southwest of Edmonton, Alberta (53°54'N 113°49'W). The surface coal mine began operations in 1989, and is located in the Dry

Mixedwood Natural Subregion of the Boreal Forest Natural Region (Natural Regions Committee 2006), with fragmented aspen forest and peatlands. Land use in the surrounding area includes power generation, coal mining, and agriculture as pasture or cropland (Capital Power, 2013). Dominant soils are Orthic and Dark Gray Luvisols with Brunisols on sandy sites. Mean annual temperature is 2.0 °C, while mean annual precipitation is 536 mm (Alberta Parks, 2015). The Genesee mine has over 550 hectares of reclaimed land; most used for agriculture, with some wetlands, reforested areas, and reconstructed county roads (Renkema et al., 2023).

Three research sites were selected based on reclamation prescriptions, end land use, and proximity to undisturbed forest for a reference (Table 3.1). Sites had comparable slope, aspect, topography, and drainage. Reclamation sites were named after respective tree species. The aspen reclamation site (hereafter Aspen) had approximately 20 cm of surface soil spread in January 2009; with coarse woody debris and straw amendments, and direct placement forest floor salvage including the LFH and Ae soil horizon from nearby forested areas. All revegetation was due to seeds and vegetative propagules in the forest surface soil and non-native species from surrounding agro-ecosystems (Navus Environmental Inc., 2012). At the time of our study (2017 to 2019), the site had high cover of graminoids, strawberry (*Fragaria virginiana*), prickly rose (*Rosa acicularis*), and raspberry (*Rubus idaeus*). The spruce reclamation site (hereafter Spruce) had stockpiled soils placed to 15 cm depth in 2010, and *Picea glauca* seedlings planted in 2013. Vegetation cover had a high percentage of graminoids, goldenrod (*Solidago canadensis*), and prickly rose. The reference site (hereafter Reference) was dry mixedwood forest dominated by *Populus tremuloides* with an understory of rose *(Rosa woodsii*) and low bush cranberry (*Viburnum edule*). Soil was characterized as a Dark Gray Luvisol. In June 2017, ten 5 by 5 m plots were established at each study site. Plots had a minimum buffer zone between them of 5 m and were distributed throughout the reclamation area to capture site variability (Figure 3.1).

2.2. Field Sampling

Soil and litter dwelling invertebrates were collected using pitfall traps, soil litter (LFH), and soil cores. Samples were collected during the Alberta growing season from May to October in 2018 and 2019. Monthly sampling was from three randomly selected subplots per plot, for a total of 30 samples per site, per collection method, per month.

Pitfall traps were made of clear 16 oz plastic containers (height 7.62 cm, top diameter 11.75 cm) filled with 200 mL of propylene glycol, and with a styrofoam plate roof anchored with bamboo skewers < 3 cm above the trap. Traps were left in the field for one week, then opened and the soil and litter was sampled on the second Thursday of each collection month. Litter (LFH) was collected using a 10 by 10 cm quadrat. All litter and organic matter were collected until mineral soil was reached. Mineral soil was sampled after litter collection, to a depth of 10 cm with a 5 cm diameter metal core. Samples were put in brown paper bags then unsealed ziploc bags, and stored in a cooler with ice packs for 4 days at 4 to 8 °C. Sampling dates were May 10, June 7, July 12, August 9, September 6, and October 18 in 2018; and May 9, June 13, July 11, August 8, September 12, and October 10 in 2019.

2.3. Invertebrate Extraction And Identification

Pitfall trap specimens were sorted into their coarse taxonomic categories, mostly to order except for ants (Formicidae), using field guides for North American insects (Borror et al., 1998; Eaton and Kaufman, 2007). Harvestmen (Opiliones) were not differentiated from spiders (Araneae) and the members of these were grouped. Araneae/Opiliones and beetles were stored separately in containers with isopropyl alcohol for preservation and for later identification. Abundance is the number of individuals per taxon per trap.

Soil cores and litter were extracted within 7 days of their collection in the field (Alberta Biodiversity Monitoring Institute, 2009). Samples were brought to the laboratory and placed on modified Tullgren extractors for 7 days (Crossley and Blair, 1991). Extracted specimens were preserved in 95 % ethanol until further processing. Soil mesofauna were separated into Collembola and major mite groups including Astigmata, Oribatida other than Astigmata, Mesostigmata, and Prostigmata. Specimens were identified using a dissecting microscope and an unpublished key from the 2018 Summer Acarology Program at Ohio State University (Walter and Beaulieu, 2014). Abundance was the number of individuals per taxon per sample. Soil and litter samples were weighed after extraction and the weights were recorded (Table 3.2).

2.4. Statistical Analyses

Plots were treated as subsamples, and monthly samples were treated as repeated measures. Indicator variables were calculated using taxa abundances (Table 3.2). Total abundance was the sum of taxa abundances per sample, total mite abundance was all mites per sample (Oribatida + Mesostigmata + Prostigmata + Astigmata). Oribatid mite to collembolan ratio (Ori/Col) and OM/PA index ([Oribatida + Mesostigmata]/[Prostigmata + Astigmata]) were calculated per sample. OM/PA index was previously shown to differentiate benchmark and intensively managed sites and soil degradation level in intensively managed arable sites in Argentina (Bedano et al., 2011).

Multivariate analyses were performed with the R statistical package (version 4.1.3) (R Core Team, 2023). Soil invertebrate assemblage was visualized using nonmetric multidimensional scaling unconstrained ordination metaMDS function in vegan, and Bray-Curtis dissimilarity (Oksanen et al., 2015). Dimensions were determined using dimcheckMDS in goeveg (McCune and Grace, 2002). To visualize significant invertebrate taxa, environmental variables, and the direction in which they correlated with the ordination plot, vec and envfit functions in vegan (alpha = 0.001, permutations = 999) were used. Permutational multivariate analysis of variance (perMANOVA) adonis in vegan was used to determine whether the soil invertebrate assemblages were significantly different among sites (alpha = 0.001, permutations = 999). P values were corrected by Bonferroni for repeated tests using RVAideMemoire. Differences among sites in invertebrate assemblages and abundance were evaluated using homogeneity of dispersions tests (betadisper), followed by ANOVAs to compare mean distance-to-centroid among sites. Significant ANOVAs (alpha < 0.05) were followed by Tukey's HSD post-hoc test to identify pairwise differences in soil invertebrate assemblages and abundance between sites.

Indicator species analysis (ISA) determines significant indicators based on aspecificity (A), which is the probability that the surveyed study site is part of the target site group; and given a species presence and sensitivity (B), the probability of finding the species in sites belonging to the target site group. The indicator value index assesses the predictive value of a species as an indicator of a combination of site groups, but fails to account for species absences inside and outside the site group combination (De Cáceres et al., 2010). Correlation indices assess positive or negative preference of a species within the site group combination, compared to the remaining sites; negative index values suggest a species avoiding particular site groups (Chytrý et al., 2002; De Cáceres et al., 2010). ISA was used to determine indicator values and point-biseral correlation coefficients (r_{pb}), to be used with species abundance data (De Cáceres and Legendre, 2009).

In our study, r_{pb} is the Pearson correlation between a binary variable indicating whether the site belongs to the site group combination (reclaimed or reference), and a quantitative variable containing species abundance. Analysis was performed with multipatt function from indicspecies package (Dufrêne and Legendre, 1997; De Cáceres and Legendre, 2009) to determine which plant species, soil properties, and soil invertebrate taxa were indicators of Reference or reclamation sites (if certain plant species or invertebrate taxa were more abundant in undisturbed or reclaimed habitats). We used a quantitative or binary response with a randomization test (alpha = 0.001, permutations = 999). The point-biseral correlation coefficient (r_{pb}) was calculated when no appropriate indicators could be selected from aspecificity and sensitivity scores.

3. RESULTS

3.1. Relative Abundance Of Major Soil Invertebrate Taxa

Major epigaeic invertebrate taxa from pitfall traps varied with sites and months, but were generally dominated by Araneae/Opiliones, beetles, and ants (Figure 3.2A). Invertebrate groups with few individuals and medians of zero included mites, grasshoppers (Orthoptera), butterflies and moths (Lepidoptera), mayflies (Ephemeroptera), diplurans (Diplura), net winged insects (Neuroptera), and thrips (Thysanoptera). Relative abundance of major litter dwelling invertebrate taxa varied by site and month, and were dominated by collembolans, oribatid mites, and prostigmatid mites (Figure 3.2B). Major taxa in topsoil were similar to dominant groups in litter but relative abundance differed (Figure 3.2C). Taxa with few individuals and medians equal to zero included diplurans, pseudoscorpions (Pseudoscorpiones), spiders, and Coleoptera, Hymenoptera, and Diptera.

3.2. Mean Abundance Relative To Reference

Mean abundance of major soil invertebrate taxa in reclamation sites were plotted relative to Reference. Ants in pitfall traps were the only taxon consistently more abundant in reclamation sites, which went through annual cycles of higher spring abundance, declining over summer (Figure 3.3A). Spiders and beetles were more stochastic, varying relative to reference by month and year (Figure 3.3A). Beetle abundances in spring pitfall traps were similar relative to reference, and increased until fall in both years. Mean abundance of mesostigmatid mites in litter and soil were similar to Reference, and varied little over time. Almost all other taxa in litter and soil, across most months, were less abundant than Reference (Figure 3.3B, C). In reclaimed litter, mean prostigmatid mite abundance was often lower than Reference early in the growing season, and higher in September and October (Figure 3.3B). This was not seen in reclaimed topsoil (Figure 3.3C). Collembolans and oribatid mite abundances were consistently lower in reclaimed than Reference, particularly in soil. Mean abundance of collembolans in reclaimed soil was lower than Reference, and differences were greater than in litter (Figure 3.3C). Abundance of oribatid mites in reclaimed litter and topsoil was lower relative to Reference, with greater differences in topsoil.

3.3. Mean Monthly Abundance Of Major Soil Invertebrate Taxa

Abundance of epigaeic soil invertebrates over the growing season varied significantly by taxon and site. Only Formicidae showed strong patterns that were consistent over most sample periods;

ants were virtually absent at the reference site and relatively abundant at both reclaimed sites. Araneae/Opiliones abundance was generally highest in July of both years; except for a large increase in October 2018 (Figure 3.4A). Reclamation sites had similar trends in Araneae/Opiliones throughout the growing season; means were generally higher in Aspen. Araneae/Opiliones abundance was similar among sites in May, and lower in Reference than reclamation sites in June. Beetle abundance was highest across sites in July and August in both years, and lowest in September and October (Figure 3.4B). Beetle abundance in Reference and reclamation sites were similar in May, June, and July; and lower in Reference in August, September, and October. In both years, beetle abundance in Aspen was highest in May and July, and in Spruce was highest in July and August. Ant abundance was higher in reclamation sites, and higher in Spruce than Aspen, specifically May through August (Figure 3.4C). Seasonal trends in reclamation sites were similar except May to June in both years, when abundance increased in Spruce and decreased in Aspen. Dipteran abundance was similar throughout the growing season, sites and years. Reference had a large peak in July of both years (Figure 3.4D).

In litter, only Oribatida showed consistent trends, typically with >10 individuals per sample in Reference and ≤10 individuals per month in reclaimed sites. Monthly collembolan abundance was similar among sites, with Reference generally higher (Figure 3.5A). Spruce had higher abundance than Reference in July and September 2019. May 2019 had significantly higher means in Reference. Monthly oribatid mite abundance in litter was highest in Reference, followed by Aspen (Figure 3.5B). Abundance in Reference was generally higher May to July and lower August to October; reclamation sites had no obvious trends. Oribatid mite abundance in September and October (2019) in Spruce was higher than in Aspen and similar to Reference. Prostigmatid mite abundance varied with site, month, and year (Figure 3.5C). Spruce had an eight fold increase in Prostigmata from September to October 2019, not observed in other sites or previous years. Mesostigmatid mite abundance in litter varied slightly by month, site, and year (Figure 3.5D). Reference generally had higher abundance than reclamation sites, although trends are weak given abundance of mesostigmatid mites collected per month was generally below five.

In soil, both Collembola and Oribatida were consistently higher in Reference. Mean monthly collembolan abundance in soil differed greatly from litter (Figure 3.6A). Reference generally had the highest abundance and Aspen lowest. The high abundance of springtails in Reference and Spruce from August to October 2019 was striking. Oribatid mite abundance exhibited the clearest distinction between Reference and reclamation sites. Abundance was relatively stable, with a general increase from July to October that was highest in Reference, then Aspen (Figure 3.6B).

Reclamation site abundance was more similar to each other than to Reference, which was not seen in other invertebrate groups. Abundance in Aspen was higher than in Spruce in 2018, and similar in 2019. Reclamation site abundance was generally below 20, half of Reference. Prostigmatid mite abundance in soil was more stable over the growing season than in litter. Abundance in Reference was highest, followed by Spruce (Figure 3.6C). Mesostigmatid mite abundance was similar in soil and litter, with mite numbers often less than five (Figure 3.6D).

3.4. Multivariate Results

3.4.1. Major soil invertebrate taxa and monthly sampling

The NMDS two-dimensional solution had a final stress of 0.145, the best solution repeated 4 times in 20 tries (Bray-Curtis) (non-metric fit, R^2 = 0.979; linear fit, R^2 = 0.898). Soil invertebrate assemblage and abundance in reclamation sites differed significantly from Reference, with non-overlapping 95 % confidence ellipses (Figure 3.7). Reclamation sites were similar to each other with larger ellipses at Spruce indicating higher variation. Reference was significantly ($p < 0.001$) associated with total collembolan (Pearson R = 0.84), oribatid (Pearson R = 0.78), and prostigmatid mite abundances (Pearson R = 0.68), while reclamation sites were significantly associated with beetle (Pearson R = 0.80), ant (Pearson R = 0.63), and true bug (Pearson R = 0.28) abundances (Figure 3.7A). Total mite abundance was significantly and strongly associated with Reference (Pearson R = 0.91), followed by total invertebrate abundance (Pearson R = 0.65), and litter weight (Pearson R = 0.50) (Figure 3.7B).

There were significant site (PERMANOVA, df = 2, F = 21.4, R^2 = 0.42, $p < 0.001$) and month effects (PERMANOVA, df = 5, F = 7.11, R^2 = 0.34, $p < 0.001$); however, the interaction between these two factors was not significant (PERMANOVA, df = 10, F = 0.69, R^2 = 0.07, p = 0.863). Post-hoc comparisons showed Reference significantly different from reclamation sites (p = 0.003), which were not significantly different from each other (p = 0.067). There were no significant differences between sampling months. Dispersion of soil invertebrate assemblages differed by site (betadisper, ANOVA, df = 2, F = 5.74, p = 0.007), being more variable in Spruce than in Reference (Tukey's HSD, 95 % CI = 0.022 to 0.148, p = 0.006), with no differences in Reference-Aspen (p = 0.555) and Spruce-Aspen (p = 0.074).

Indicator species analysis (ISA) was used to determine significant soil invertebrate taxa for Reference and reclamation sites. The point-biserial correlation coefficient (r_{pb}) was used to calculate a correlation index to determine soil invertebrate taxa indicators, as no appropriate indicators were selected based on aspecificity and sensitivity values (Table 3.3). Strongest and most significant ($p = 0.001$) indicators were oribatid mites and springtails in Reference; prostigmatid mites were also indicators ($p = 0.013$). The only significant indicator taxa for both of the reclamation sites, was for ants ($p = 0.006$), while predatory mesostigmatid mites were an indicator for Reference + Spruce ($p = 0.016$).

3.4.2. Major soil invertebrate taxa and sampling methods

Epigaeic invertebrates from pitfall traps had an NMDS two-dimensional solution (Bray-Curtis) with a final stress of 0.132 (non-metric fit, $R^2 = 0.983$; linear fit, $R^2 = 0.918$). Abundance and community assemblage were similar, with high overlap in 95 % confidence ellipses (Figure 3.8A). Significant associations ($p < 0.001$) of soil invertebrate taxa vectors were seen for spiders and harvestmen (Pearson R = 0.88), beetles (Pearson R = 0.76), and ants (Pearson R = 0.52). Site (PERMANOVA, df = 2, F = 4.69, $R^2 = 0.12$, $p < 0.001$) and month (PERMANOVA, df = 5, F = 7.80, $R^2 = 0.50$, $p < 0.001$) were significant, but their interaction was not (PERMANOVA, df = 10, F = 1.21, $R^2 = 0.15$, $p = 0.220$). Post-hoc comparisons showed no significant differences between reclamation sites ($p = 1.00$), or Aspen and Reference (0.192), but slight differences for Spruce and Reference ($p = 0.033$). There were significant differences by month; July, May, and June ($p = 0.045$), and July and September ($p = 0.030$). Dispersion of epigaeic invertebrates did not differ by site (betadisper, ANOVA, df = 2, F = 0.581, $p = 0.565$). Indicator species analysis with the correlation index using the point-biserial correlation coefficient (r_{pb}), found that ants were the only significant indicator, and this was assigned to Spruce and Aspen (Pearson R = 0.520, $p = 0.004$).

Extracted litter dwelling invertebrates had an NMDS two-dimensional solution (Bray-Curtis) with a final stress of 0.089 (non-metric fit, $R^2 = 0.992$; linear fit, $R^2 = 0.962$). Soil invertebrate assemblage and abundance in litter was similar between reclamation sites, with nearly full overlap between 95 % confidence ellipses (Figure 3.8B). There was no overlap between Aspen and Reference, but some with Spruce and Reference. Prostigmatid mites had the strongest association with our ordination (Pearson R = 0.82, $p < 0.001$), but without clear site preferences. Oribatid mites (Pearson R = 0.76), collembolans (Pearson R = 0.73), and mesostigmatid mites (Pearson R = 0.55) were significantly ($p < 0.001$) associated with Reference, with the strongest association in OM/PA index (Pearson R = 0.80). There were significant site (PERMANOVA, df = 2, F = 7.68, $R^2 = 0.25$, $p < 0.001$), and month effects (PERMANOVA, df = 5, F = 3.16, $R^2 = 0.25$, $p = 0.002$), but no interaction effects (PERMANOVA, df = 10, F = 1.34, $R^2 = 0.21$, $p = 0.178$). Post-hoc comparisons showed no significant differences between reclamation sites ($p = 0.840$), and significant differences between Reference and Spruce ($p= 0.015$) and Reference and Aspen (p = 0.003). Significant differences between months occurred for May to October (p = 0.06). Dispersion of soil invertebrate assemblages in litter differed by site (betadisper, ANOVA, df = 2, F = 6.07, p = 0.006), with litter dwelling assemblages more variable in Spruce than Aspen (Tukey's HSD, 95% CI = 0.030 to 0.239, p = 0.009), and to a lesser extent than Reference (Tukey's HSD, 95% CI = 0.017 to 0.226, p = 0.020); differences were not significant in Reference Aspen (p = 0.951). Indicator species analysis (ISA) with the correlation index using the point-biserial correlation coefficient (r_{pb}), determined two significant indicators, oribatid mites (Pearson R = 0.689, p = 0.001) and collembolans (Pearson R = 0.450, p = 0.012); both were assigned to Reference. There were no indicator taxa for individual reclamation sites or both reclamation sites.

Soil dwelling invertebrate NMDS analysis (Bray-Curtis) provided a two-dimensional solution with a final stress of 0.064 (non-metric fit, $R^2 = 0.996$; linear fit, $R^2 = 0.983$). Soil invertebrates in topsoil were significantly different among sites, with clearly separated 95 % confidence ellipses with no overlap; reclamation sites were more similar to each other than Reference (Figure 3.8C). Vectors for oribatid and prostigmatid mites showed similar significant association (Oribatida: Pearson R = 0.84, Prostigmata: Pearson R = 0.76; p < 0.001), along with Ori/Col ratio (Pearson R = 0.86, p < 0.001). An opposing significant association occurred in collembolans (Pearson R = 0.91, p < 0.001). Total abundance had a significant positive association with Reference (Pearson R = 0.93, p < 0.001). There were significant site (PERMANOVA, df = 2, F = 34.26, $R^2 = 0.64$, p < 0.001), and month effects (PERMANOVA, df = 5, F = 3.34, $R^2 = 0.16$, p = 0.008), but no interactions (PERMANOVA, df = 10, F = 0.44, $R^2 = 0.04$, p = 0.961). Post-hoc comparisons showed significant differences between reclamation sites (p = 0.003), Reference and Spruce (p= 0.003), and Reference and Aspen (p = 0.003). Post-hoc comparisons found no significant differences among months. Dispersion of soil invertebrate assemblages in soil did not differ by site (betadisper, ANOVA, df = 2, F = 0.72, p = 0.495). Indicator species analysis with the correlation index using the point-biserial correlation coefficient (r_{pb}), determined three significant indicators for Reference, oribatid mites (Pearson R = 0.889, p = 0.001), collembolans (Pearson R = 0.702, p = 0.001), and prostigmatid mites (Pearson R = 0.637, p = 0.001). An additional indicator was selected for Reference and Spruce, predatory mesostigmatid mites (Pearson R = 0.514, p = 0.007).

4. DISCUSSION

Here we provide one of the few datasets examining temporal and annual trends in a diversity of invertebrate groups at reclaimed sites. Taxa that showed promise as reclamation indicators were epigaeic ants, which were more abundant at reclamation sites, and soil and litter dwelling oribatid mites and springtails, which were more abundant in Reference. Trends for these taxa were largely consistent across sampling months and years. We recommend avoiding sampling in the fall, as relative abundances in September and October varied from trends observed in May through August. Pitfall traps required two site visits but could be processed relatively quickly in the lab, while soil samples required specialized equipment and extra laboratory time for extraction. For that reason, epigaeic ants may be the most efficient indicator of reclaimed sites in our study.

Contrary to our hypothesis, invertebrate abundance was not lower in reclamation sites, nor was it the lowest in Spruce. Certain groups, such as oribatid mites, exhibited decreased abundance in reclamation sites compared to Reference and were less abundant in Spruce than in Aspen. In contrast, other groups, including mesostigmatid mites, ants, and beetles, showed higher abundance in reclamation sites than Reference. As hypothesized, most invertebrate groups did differ in reclamation sites compared to Reference, and our Reference site was well separated in multivariate space from the reclaimed sites. However, few groups peaked in abundance in September as predicted. Monthly trends varied between years for some taxa, such as Prostigmata, but many taxa including ants, true flies, and beetles were at their lowest abundance in the fall. We further explore these results to understand the variability in soil invertebrate responses to reclamation practices.

4.1. Soil Invertebrate Assemblages And Abundance

Significant differences in soil invertebrate assemblages and abundances in Aspen and Spruce relative to Reference was expected as reclaimed sites were young. We expected Spruce with stockpiled topsoil to differ significantly from Aspen with direct placed forest floor and soil materials. The similar changes to soil invertebrate assemblages and abundance throughout the growing season between reclamation sites, and similarities in our ordination plots, suggest regional factors such as climate had a stronger effect on monthly invertebrate abundance and assemblage than soil reclamation methods. This supports the work of Déchêne and Buddle (2009) and Erdmann et al. (2012) who found regional factors had a greater impact on oribatid mite community structure than reduced intensity forest harvesting methods and forest type.

Examining differences in epigaeic, litter dwelling, and soil dwelling invertebrate assemblages and abundance showed interesting trends of soil invertebrate recovery in reclaimed systems. Epigaeic soil invertebrates were the most homogenous between sites, with only Spruce differing significantly from Reference. This suggests these taxa are quicker to recover in reclamation sites.

Recovery of invertebrate assemblages at land reclamation sites can be informed by research in fire disturbed forest stands, as in both disturbances most vegetation is absent and litter and some topsoil have been lost. Soil invertebrate recovery after forest fires has been extensively studied and occurs through immigration from undisturbed sites, survival in deep soil layers and stockpiles, and amendments or soil building materials (Gongalsky and Persson, 2013; Malmström et al., 2009; Zaitsev et al., 2014). Recovery is sometimes slow; five years after a prescribed burn in a Swedish mixedwood forest, soil invertebrate taxa with strong dispersal abilities including true flies, beetles, and spiders had not recovered (Malmström et al., 2009). Six years after forest wildfires in central Sweden and northwestern Russia some soil surface dwelling invertebrate taxa with low dispersal abilities immigrated from an unburnt area, although most recovered through survival in deep soil layers (Gongalsky and Persson, 2013). Above ground soil macro and meso fauna were more vulnerable to forest fire than below ground taxa (Zaitsev et al., 2014); however, in surface mining soil disturbance makes below ground taxa very vulnerable.

Abundance and assemblages of epigaeic soil invertebrate taxa collected by pitfall trapping appear to be recovering, and are recovering more compared to soil invertebrate groups in litter and topsoil. This may be explained by the fact that almost all above ground soil invertebrate taxa collected through pitfall trapping were relatively mobile with strong dispersal capabilities. Spruce is a further distance from Reference than Aspen, and has more dispersal limitations. Beetle and spider species at the borrow pit forest reclamation sites in northern Alberta were good dispersers and tolerant of a variety of environmental conditions (Hammond et al., 2022). Biotic and structural properties of reclamation sites impacted soil invertebrate response more than abiotic factors did' however, significant variation in assemblages suggest site location, colonization rates, and/or disturbance history could play a significant role in soil invertebrate community structure recovery after disturbance (Hammond et al., 2022).

Differences in abundance and assemblages of litter and soil dwelling invertebrate taxa in Aspen and Spruce indicate slower recovery to Reference. These differences in Aspen and Spruce relative to Reference are likely due to litter amount and composition in grass dominated reclamation sites, relative to forest LFH. Simple (mono-species) and mixed litters differ in microhabitat heterogeneity, and litter with greater variety in substrate lability increases microhabitat variety more quickly (Hansen and Coleman, 1998). Oribatid mite abundance, richness, and diversity in forest soils is strongly related to litter, as noted by their declines in litter of reduced heterogeneity in a deciduous forest research site in North Carolina where simple litter types lost structure which reduced available habitat and humus retention (Hansen, 2000).

Our young reclamation sites lack a forest canopy and have lower tree cover than other vegetation groups; combined with a thin litter layer, this creates a transient, exposed, and unfavourable habitat that cannot retain moisture as effectively as forest litter. Oribatid mite richness, abundance, and diversity in forest reclamation sites in northern Alberta was affected by accumulation and formation of a new forest litter layer (McAdams et al., 2018). Litter mass and soil carbon were positively correlated with oribatid mite densities in forest management research sites in Germany, while soil pH was negatively correlated (Erdmann et al., 2012). Interestingly, litter dwelling soil invertebrate assemblages were more similar between Spruce and Reference, than Aspen and Reference, likely related to site water. Depth, heterogeneity, and water holding capacity of forest floor soils increased oribatid mite richness in a temperate rainforest on Vancouver Island (Lindo and Winchester, 2008). We found no significant differences in litter dwelling invertebrate assemblages and abundance between our reclamation sites. We expect Aspen and Spruce to diverge at some point during ecological succession, due to dominant tree species, and because reclamation sites with direct placement or stockpiled soil had significantly different plant community composition 18 to 24 years post-reclamation (Dhar et al., 2019).

Assemblages and abundance of soil invertebrate taxa in topsoil showed the most differentiation among sites, suggesting slower recovery than litter and increased sensitivity to reclamation methods. Analyzing our invertebrate collection methods, significant differences between sites were observed only in soil. This was expected given notable differences in reclamation methods, and common management issues including compaction, adverse soil chemical properties including pH, and microbial communities. Research on increasing biomass removal and soil disturbance in jack pine forests in northern Ontario found collembolans needed high forest floor moisture, and oribatid mites were influenced by nutrient richness and pH (Rousseau et al., 2018).

Soil compaction can occur in reclamation sites from heavy soil handling machinery. Soil and site conditions, such as texture and drainage can contribute to compaction severity (Ampoorter et al., 2012; Startsev and McNabb, 2009). Soil compaction associated with timber harvesting reduced total mite and mesofauna abundance at conifer and deciduous dominant stands in northern Alberta, and decreased soil mesofauna densities by 50 % in British Columbia (Battigelli et al., 2004; Lindo and Visser, 2003). Lower collembolan abundance was associated with removal of harvest debris and with severe soil compaction from mechanical felling (Bird et al., 2004). Treatment options for compacted areas can include deep tillage or ripping and soil amendments to alter structure and permeability (Bateman and Chanasyk, 2001). Mite recovery in clear cut sites at a pine plantation in eastern Texas was quicker with more intensive harvesting and site preparation treatments including bedding (elevated rows with coarse woody debris), herbicide treatments, and fertilizer treatments (Bird et al., 2004).

Our relatively young reclamation sites, with limited soil development and absence of litter and moss layers, are undergoing soil formation and developing soil bacterial and fungal communities. Key environmental conditions that contributed to soil invertebrate community changes twenty years after biomass removal in jack pine forests, were organic soil horizon development, accumulation of fallen woody and coarse woody debris, and development of continuous moss layer (Rousseau et al. 2018a). Reclaimed forest litter layers lack evidence of meso faunal and fungal activity, fine roots, and humic layer development (Sorenson et al., 2011). Reclaimed forest sites in northern Alberta are bacteria dominant while undisturbed boreal forest has higher fungal abundance, due to soil abiotic properties and indirectly by reclamation effects on plant growth (Dimitriu et al., 2010). Total mite and collembolan abundance had a positive correlation with microbial and fine root biomass in conifer and deciduous stands (Lindo and Visser, 2003).

Oribatid mites were strongly correlated with Reference, with slow recovery in litter and topsoil of reclamation sites relative to collembolans and prostigmatid mites, suggesting oribatid mites are more sensitive to soil reclamation methods. Battigelli (2011) also found lower oribatid mite abundance compared to a reference, at forest reclamation in northern Alberta. Oribatid mite relative abundance was lower, prostigmatid mite abundance was higher, and collembolans and mesostigmatid mites in reclaimed sites were similar to forest references (Battigelli, 2011). Oribatid mites were more impacted than prostigmatid mites after burning and clear cutting in a mixedwood forest in Sweden (Malmström et al., 2009), and after stem only harvesting in interior British Columbia (Battigelli et al., 2004). Greater impacts to oribatid mites than collembolans was recorded two years post harvest in a mixed conifer forest (Bird and Chatarpaul, 1986) and two years post disturbance in jack pine stands in Ontario (Rousseau et al., 2019).

We saw an increased abundance of springtails and mesostigmatid and prostigmatid mites at our reclamation sites, which could be associated with dispersal capabilities and plant community composition of early forest successional stages. Lindo and Visser (2003) suggested mesofauna, mite, and collembolan abundance in clearcut treatments were not significantly different from uncut references because of tree, shrub and grass regeneration. Another reason for increased abundance of these groups in reclamation sites, is some groups, such as prostigmatid mites, like disturbance. Research on long term agricultural management found oribatid mites with longer life cycles can be eliminated as they are susceptible to habitat disturbance, while encouraging multiplication in prostigmatid mites (Behan-Pelletier, 2003). Increased management of young reclamation sites could be responsible for lower oribatid mite abundance and higher prostigmatid mite abundance in Aspen and Spruce. Mesostigmatid mite abundances in Aspen and Spruce were similar to Reference, which indicates these mobile and often predatory soil mites may have recovered. Particularly in Spruce where mesostigmatid mites were an indicator for Reference + Spruce. Mesostigmatid mites restored in abundance seven years post wildfire, regardless of distance from an unburned reference pine forest in Sweden (Zaitsev et al., 2014).

4.2. Soil Invertebrate Taxa As Indicators

Appropriate indicators were determined based on correlation indices of indicator taxa, consistent relationships relative to reference regardless of sampling month or year, and ability to reflect soil reclamation methods between Aspen and Spruce with Reference > Aspen > Spruce. Our assessment of soil invertebrate indicator taxa, at coarse taxonomic resolution, found ants (Formicidae) an appropriate indicator for reclamation sites. While ant abundance did not always distinguish Aspen and Spruce, it was the strongest indicator taxa for reclamation sites and showed reliable trends over the growing season and sampling years. The consistent presence and behavior of ants in these disturbed ecosystems highlight their potential to serve as a practical and informative indicator in reclamation monitoring, providing valuable insights into the ecological health and progression of reclaimed sites. Ants are easier to count and categorize than many other soil invertebrate taxa, which makes them a convenient choice for monitoring. A downside to using ants as indicators is that collecting in pitfall traps can be inconsistent. Occasionally, our traps placed near an ant hill would yield samples with disproportionately high abundance and relative abundance, which may skew data. Despite this variability, ants remain a strong candidate for use in reclamation assessments due to their ecological significance and ease of identification.

Significant indicators for Reference included oribatid mites, springtails, and prostigmatid mites. Prostigmatid mites did not have a consistent relationship relative to reference and did not reflect differences between Aspen and Spruce. Oribatid mites had higher correlation and positive predictive value than springtails. At the coarse level of taxonomic resolution applied, oribatid mites were the strongest indicator, sensitive to accurately reflect differences in reclamation methods at Aspen and Spruce, but not so sensitive that abundances fluctuated wildly month to month, or between years. Oribatid mites in Aspen and Spruce were consistent relative to Reference.

Our results are echoed by recent research from Lumley et al (2023), who found that oribatid mite assemblages in northern Alberta's oil sands region demonstrated a distinct landscape level response to natural land cover, anthropogenic disturbance, space, and climate, suggesting their potential as robust indicators for assessing soil condition in the area. Oribatid mite abundance remained consistent across various natural land covers and most human footprints, except for mines, well sites, and cultivation, which significantly reduced abundance. Broader spatial factors such as forest type, understory plant composition, litter composition, and abiotic soil properties had limited impact on total oribatid mite abundance (Lumley et al., 2023). Meta-analysis comparing natural variation in oribatid mite community structure with deviations associated with three anthropogenic disturbance types, found oribatid mite assemblages effective community level indicators when comparing disturbed to undisturbed reference sites; disturbances may have greater effects on oribatid mite abundance than individual species (Gergócs and Hufnagel, 2017). According to Gergócs et al (2012), oribatid mites are recommended as indicators in climate change research because samples can be collected from most substrates and habitat types. Collection is fast and effective, oribatid mites have seasonal stability and reflect ecological characteristics of their habitats, overriding geographical patterns (Gergócs et al., 2012).

Not all researchers agree oribatid mites are effective and appropriate indicators. Lupardus et al. (2021) assessed reclamation and soil quality of cultivated lands in southern Alberta disturbed by oil and gas activities, and found indicator taxa were Oribatida, Prostigmata, and Collembola. Broad taxonomic soil invertebrate identification and Acari:Collembola (A:C) ratio were not appropriate indicators for soil biological quality of reclaimed and cultivated lands as soil property differences were not reflected in densities or assemblages. Strip-cut partial harvesting and clear cutting in northern Alberta aspen stands changed abundance of oribatid mites but not diversity and community composition, limiting their use as a biological indicator (Lindo and Visser, 2003). Other researchers say it is important to include soil invertebrate indicators when assessing oribatid mite response. Oribatid community response to biomass removal showed stronger functional homogenization than Reference, and contrasted with collembolan response, making both taxa strong potential biological indicators for forest management (Rousseau et al., 2019). In our study, collembolans were significant indicators for Reference, ranked after oribatids, but did not reflect reclamation methods and individual site trends that fluctuate throughout the growing season. As an indicator, collembolan abundance lacked sufficient sensitivity and consistency.

Our results suggest oribatid mite abundance may be an appropriate indicator in monitoring reclamation; however, incorporating soil invertebrates into current reclamation criteria requires low cost and time efficient sampling that can be easily taught to reclamation practitioners, making detailed taxonomic identification infeasible. Impacts to oribatid mites can be observed at taxonomic levels coarser than species, including total abundances and shifts in community composition with morphotyped species, making their use as indicators of soil disturbance and recovery in reclamation sites feasible for reclamation practitioners without specialized expertise (Behan-Pelletier and Lindo, 2022). When examining high level taxonomy and its effectiveness in indicating disturbance in boreal forest, researchers recommended family level identification of oribatid and mesostigmatid mites when taxonomic expertise is not available (Meehan et al., 2019). Researchers in a national monitoring program for COUNTRY encourage high level taxonomic identification of mesofauna to understand ecological contexts of soils (George et al., 2017).

While our results showed that oribatid mites are the strongest soil invertebrate indicator for our sites, more research is needed to fully understand the impact of reclamation methods on oribatid mite community composition and function. Some oribatid mites are associated with disturbed soils, such as *Tectocepheus* spp., recording high abundance or high relative abundance (Behan-Pelletier and Lindo, 2022). Many of these species are small, parthenogenetic, and are considered ecosystem succession pioneer species (Behan-Pelletier and Lindo, 2022). Colonization and development of oribatid mite communities in forest reclaimed limestone dumps in northern Poland occurred in a five year old reclaimed site due to *Tectocepheus* spp. (Mozos, 2012). While not recorded, I observed high *Tectocepheus* spp. abundance in Aspen and Spruce; the relative abundance of this genus relative to others should be further assessed.

Our understanding of mite species in Canada is still evolving. Diversity in soil and litter is shaped by environmental differences at regional scales and is potentially more diverse than the most diverse insect orders (Young et al., 2019). Based on extrapolation researchers, Canada is estimated to have 10,000 to 15,000 mite species, of which 70 % are not recorded or described by taxonomists; arboreal litter and deep soils are expected to have high mite diversity (Beaulieu et al., 2019). Systems of classifying oribatid mites to genus or morphological groups is part of a standardized method to compare disturbed and transformed habitats (Gergócs et al., 2012). Our future research includes identification of oribatid mites to family and genus, assessment of functional traits outlined by Rousseau et al. (2019), and developing a rudimentary morphospecies photo sorting algorithm for dominant groups in reclaimed and reference sites.

The only significant soil invertebrate taxa indicator for reclamation sites was ants. Paired with the types of plants that were indicator species for both reclamation sites, it is likely ant abundance is related to high cover of grassy, herbaceous, and weedy vegetation and thrives in disturbed sites. Oil and gas reclamation sites less than 10 years post-reclamation on cultivated lands in Alberta, were more susceptible to crop pests and plant feeding groups of barklice and true bugs (Lupardus et al., 2021). Plant feeding groups of thrips and true bugs had higher abundance than an undisturbed reference 12 years after a prescribed burn (Gongalsky and Persson, 2013). Numerous anthills and higher aphid and seed bug (Hemiptera) abundance was observed at Spruce. There is value in reclamation practitioners to be aware of these soil invertebrate taxa, similar to management of weed and non-native plant species that can establish and thrive in forest reclamation sites (Native Plant Working Group, 2000; Small et al., 2018; Trepanier et al., 2021).

4.3. Appropriate Sampling Methods And Timing

Ants were only collected through pitfall traps, but our results suggest sampling in September and October should be avoided. Installing pitfall traps in reclamation sites for monitoring should occur early in the growing season (April or May) in order to capture peak abundance (June to August). Avoiding nearby ant hills should be noted in sampling protocol.

We recommend assessing and collecting oribatid mites with soil cores from the upper 15 cm of topsoil. This method was consistent between sampling months and years. Soil core samples were sensitive enough to show significant differences between reclamation sites, while litter samples were not as sensitive. We hypothesize this is related to site age, as planted trees did not contribute greatly to litter; dead grasses were the dominant component, and litter could not be classified as novel forest floor. When time and budget permit, we recommend including independent litter samples with soil cores when assessing oribatid mites in reclamation sites. Soil mite abundance and recovery is impacted by dominant tree species and forest floor characteristics (Díaz-Aguilar et al., 2013; McAdams et al., 2018; Sylvain and Buddle, 2010). Reclamation sites have lower litter decomposition rates than undisturbed sites, with site age, canopy cover, and shrub cover being key factors to forest floor litter development (Rowland et al., 2009; Sorenson et al., 2011). Future research should determine the time frame when litter dwelling oribatid mites in aspen dominant reclamation sites diverge from spruce dominant reclamation sites.

We recommend collecting samples for oribatid mite reclamation monitoring early (May, June) or later in the growing season (September, October), while avoiding peak growing season (July and August). While oribatid mite abundance was consistently high in Reference, regardless of sampling month, reclamation sites had slightly lower abundance in July and August.

5. CONCLUSIONS

Our results have important and practical implications for land reclamation monitoring and for use of soil invertebrate taxa as indicators of ecosystem recovery. We found that reclamation methods impacted soil invertebrate taxa abundance differently, likely due to their varied mobility, life cycles, habitat requirements, and overall sensitivity to disturbance. Ants and oribatid mites in soil emerged as robust indicators of reclamation success in our study. Identification training for reclamation practitioners using simplified taxonomic keys or photo reference libraries is potentially feasible. The prevalence of both indicators, and the sensitivity or oribatid mites in topsoil, underscore their significance for reclamation success monitoring.

Oribatid mites in the topsoil emerged as the most reliable indicator taxon, consistently showing significant differences between Spruce (with stockpiled topsoil and planted *Picea glauca*), Aspen (with direct placement of forest floor and topsoil), and Reference. Their abundance was highly sensitive to soil reclamation methods, remaining consistent across the different months and years of the study, and making them a strong candidate for inclusion in reclamation criteria and reclamation success monitoring. However, the practicality of using oribatid mites as indicators comes with several challenges. Assessing oribatid mites requires more time, specialized equipment, and specialized expertise compared to ants, which increases monitoring costs. Given these constraints, it is important to strike a balance between more specialized groups like oribatid mites and other indicators, like ants, which are easier to count and categorize and less expensive to monitor. While oribatid mites can provide detailed and sensitive information about soil health and the effectiveness of reclamation methods, ants offer a more accessible and cost effective alternative, even with sampling variability.

This balance between reclamation and reference indicators is essential for developing informed reclamation methods and reclamation policies. By integrating both detailed, sensitive indicators such as oribatid mites, and more practical, widespread indicators such as ants, we can create a comprehensive monitoring strategy that is both scientifically rigorous and feasible for long-term application. This approach will ensure that reclamation efforts are guided by reliable data while still remaining within practical and financial constraints.

Further research is needed to determine the extent to which oribatid mite abundance as an indicator of ecosystem recovery in forest reclamation sites can be extrapolated, including to other forest reclamation sites and other ecozones throughout the boreal forest. We need to assess the impact of reclamation methods including soil stockpiling and storage, soil building materials and soil amendments, and planting approaches including use of native plant species and weed management. A stronger understanding of oribatid mite recovery rates and abundances in newly reclaimed sites, older reclaimed sites, and along a chronosequence will help establish general thresholds for reclamation practitioners to aim for.

Results for soil samples from our reclamation sites showed significant differences in subsoil (15 to 30 cm), indicating the necessity for future research on oribatid mite abundance below 10 cm. Depending on outcomes, vertical oribatid mite abundance patterns could serve as a valuable tool for monitoring, potentially identifying soil layers that may hinder vegetation growth or water infiltration. Understanding changes to oribatid mite community composition in response to soil reclamation methods, and tracking abundance of disturbance associated species, is critical to determining if oribatid mite abundance is an appropriate indicator for forest reclamation monitoring. Ensuring the continued ecosystem recovery and long-term resiliency and success of forest reclamation sites in Alberta is paramount to ecological integrity and function and mandatory for maintaining ecosystem services and health.

6. TABLES AND FIGURES

Table 3.1. Research site locations and ecosite descriptions of two reclaimed sites (Aspen, Spruce) and proximate undisturbed forest site (Reference). Age is time since reclamation.

Site	Location (°N, °W)	Age (years)	Reclamation Prescription Soil Type	Dominant Tree Species	Dominant Shrub Species	Dominant Herbaceous Species
Aspen	53.3252, -114.3142	12	~ 20 cm topsoil, direct placed LFH and Ae, coarse woody material, straw	Aspen	Prickly rose, raspberry	Grasses, wild strawberry, clover
Spruce	53.3340, -114.3105	8	~ 15 cm salvaged stockpiled topsoil, straw	White spruce	Prickly rose	Grasses, goldenrod
Reference	53.3265, -114.3154		Orthic Gray Luvisol	Aspen	Rose, low bush cranberry	Bunchberry, bedstraw, bishop’s cap

Table 3.2. Sampling, environmental, and indicator variables used in NMDS analysis.

Variable Type	Variable	Description
Sampling	Month	Monthly collection May to October (2018, 2019), n=30 per research site and sampling method
Environmental	Litter weight (g)	Weight of dried dead litter and organic matter from 100 cm^2 area to mineral soil surface, indicates amount of LFH
	Soil weight (g)	Weight of dried soil core (depth 10 cm, diameter 5 cm), indicates general bulk density of soil
Indicator	Total Abundance	Univariate indicator of all soil invertebrates collected per sample
	Total Mite Abundance	Univariate indicator of all soil mite (Acari) collected per sample
	Ori/Col ratio Oribatida/Collembola ratio	Multivariate indicator of two most abundant soil invertebrate taxa, oribatids are k selected and collembolans are r selected, natural conditions ratio > 1 (Menta, 2012)
	OM/PA index (Oribatida + Mesostigmata)/ (Prostigmata + Astigmata)	Multivariate indicator to assess soil biological degradation based on differential responses and sensitivity to agricultural practices (Bedano et al., 2011)

Table 3.3. Indicator species analysis using the point-biserial correlation coefficient to examine soil invertebrate assemblages. Values are soil invertebrate taxa significant at α = 0.05, R = Pearson correlation between soil invertebrate taxa and site, ** $p < 0.001$, * $p < 0.05$. Bolded values indicate a strong positive association, correlation > 0.6.

Site	Soil invertebrate taxa	R	p value
Spruce	True bugs (Hemiptera)	0.312	0.072
Reference	**Oribatid mites (Oribatida)**	**0.934**	**0.001****
	Springtails (Collembola)	**0.757**	**0.001****
	Prostigmatid mites (Prostigmata)	0.471	0.013*
	True flies (Diptera)	0.321	0.120
Aspen + Spruce	Ants (Formicidae)	0.521	0.006*
	Beetles (Coleoptera)	0.232	0.369
Reference + Aspen	Earthworms (Lumbricidae)	0.341	0.102
	Spiders and harvestmen (Araneae/Opiliones)	0.135	0.720
Reference + Spruce	Mesostigmatid mites (Mesostigmata)	0.441	0.016*
	Bees and wasps (Apoidea)	0.410	0.034*

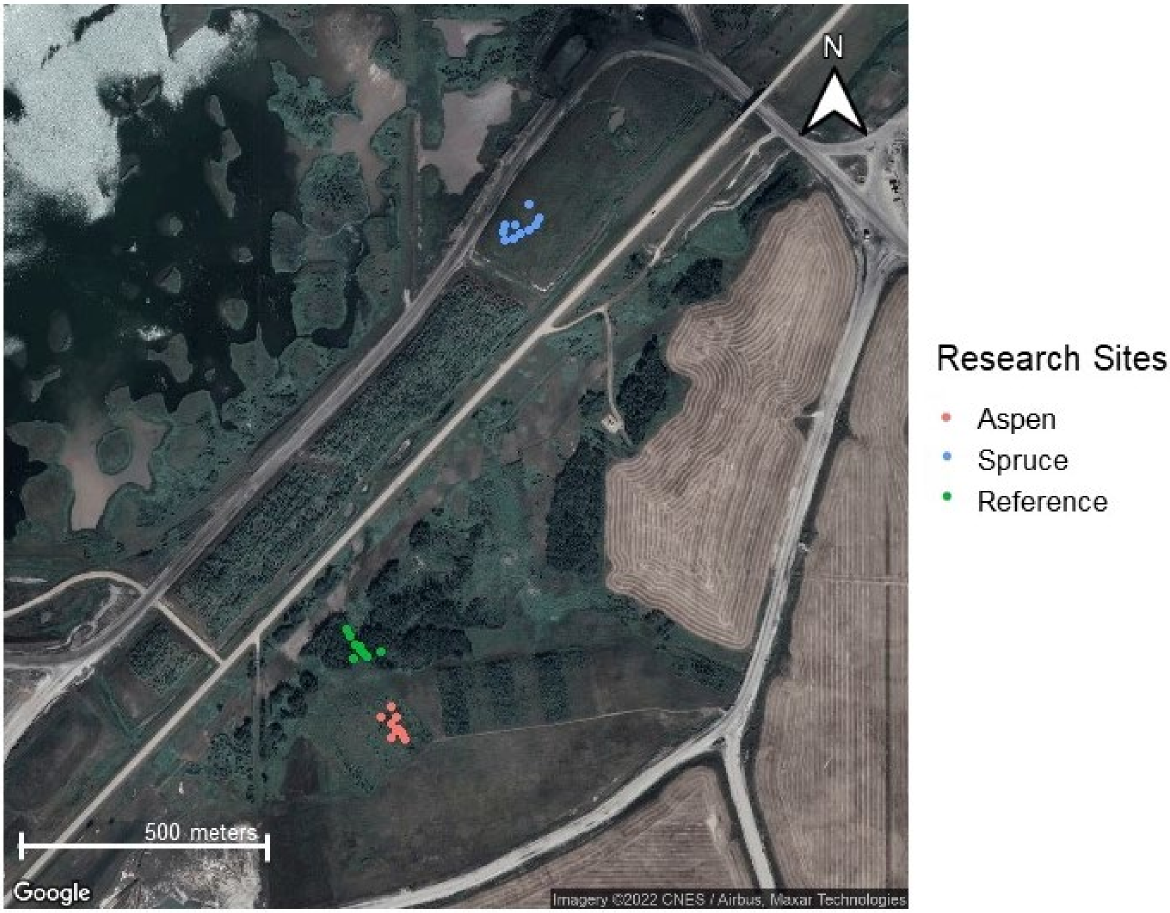

Figure 3.1. Research sites located at the Genesee Coal Mine. Highway 770 separates the spruce reclamation site (Spruce) from the forest reference (Reference) and aspen reclamation site (Aspen), all other roadways are haul roads, settling pond (NW corner), and reclaimed agricultural field (SE corner).

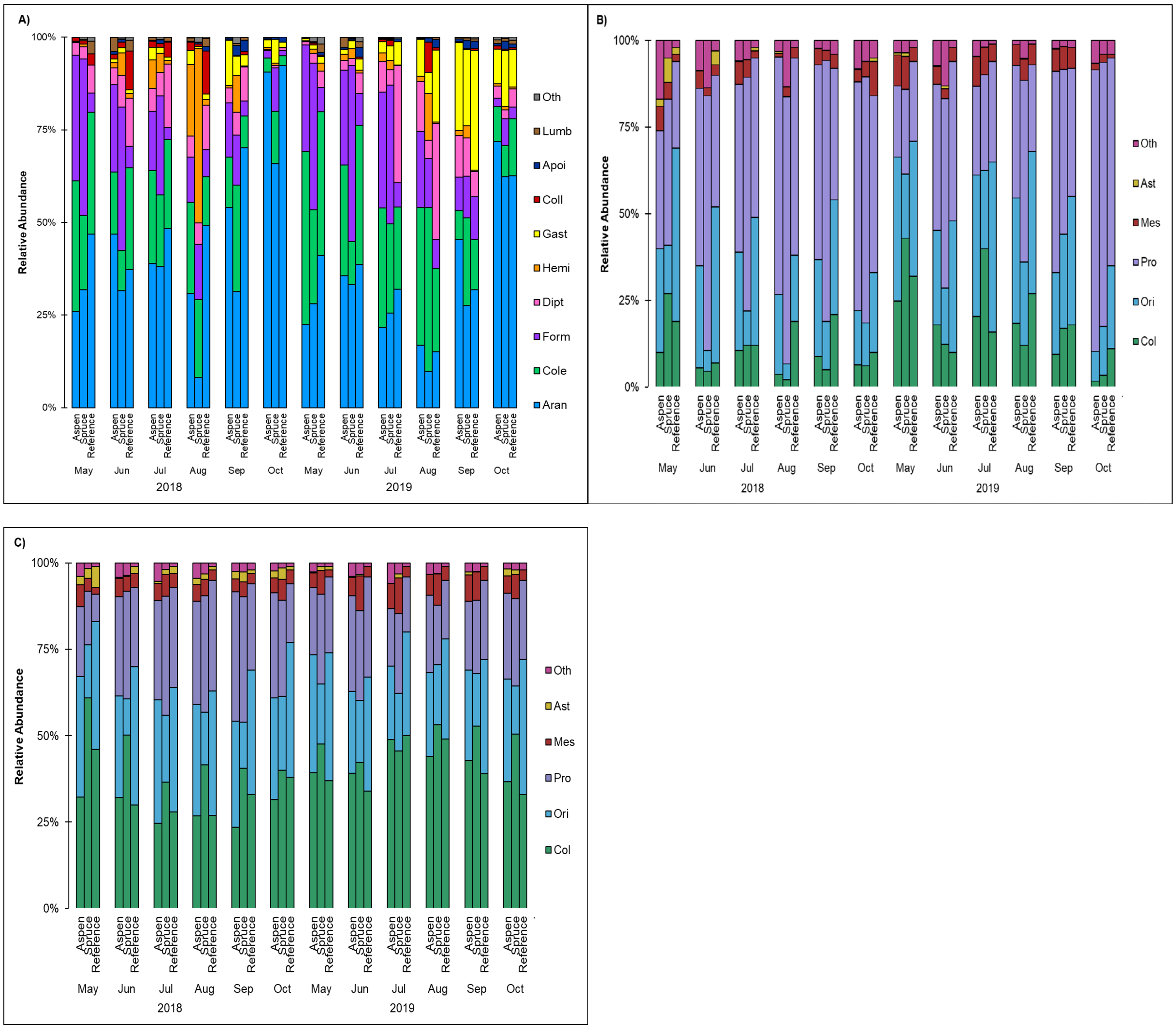

Figure 3.2. Relative abundance of (A) above ground, (B) litter dwelling, and (C) soil dwelling invertebrate taxa. Soil invertebrates in other (Oth) included groups with too few individuals (median equal to 0). Taxon abbreviations are as follows: spiders and harvestmen (Aran), beetles (Cole), ants (Form), true flies (Dipt), true bugs (Hemi), slugs and snails (Gast), springtails (Coll), bees and wasps (Apoi), earthworms (Lumb), oribatid mites (Ori), prostigmatid mites (Pro), mesostigmatid mites (Mes), and astigmatid mites (Ast).

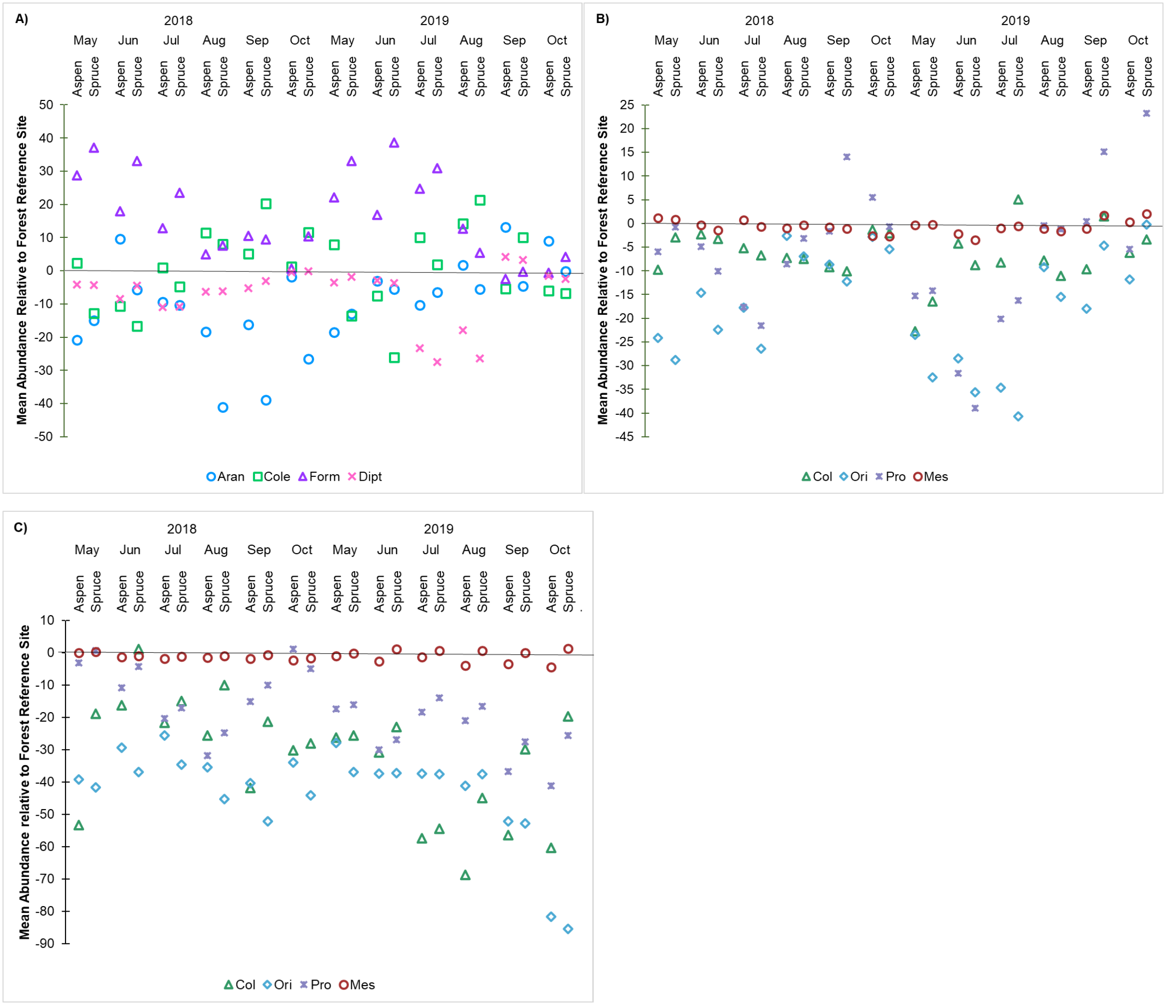

Figure 3.3. Mean abundance of soil invertebrate taxa in Aspen and Spruce relative to Reference. Soil invertebrates were collected monthly from May to October 2018 and 2019. Values were calculated as Reclaimed minus Reference. Soil invertebrate groups from (A) above ground pitfall traps, (B) litter dwelling, and (C) below ground. Taxon abbreviations are as follows: spiders and harvestmen (Aran), beetles (Cole), ants (Form), true flies (Dipt), springtails (Col), oribatid mites (Ori), prostigmatid mites (Pro), and mesostigmatid mites (Mes).

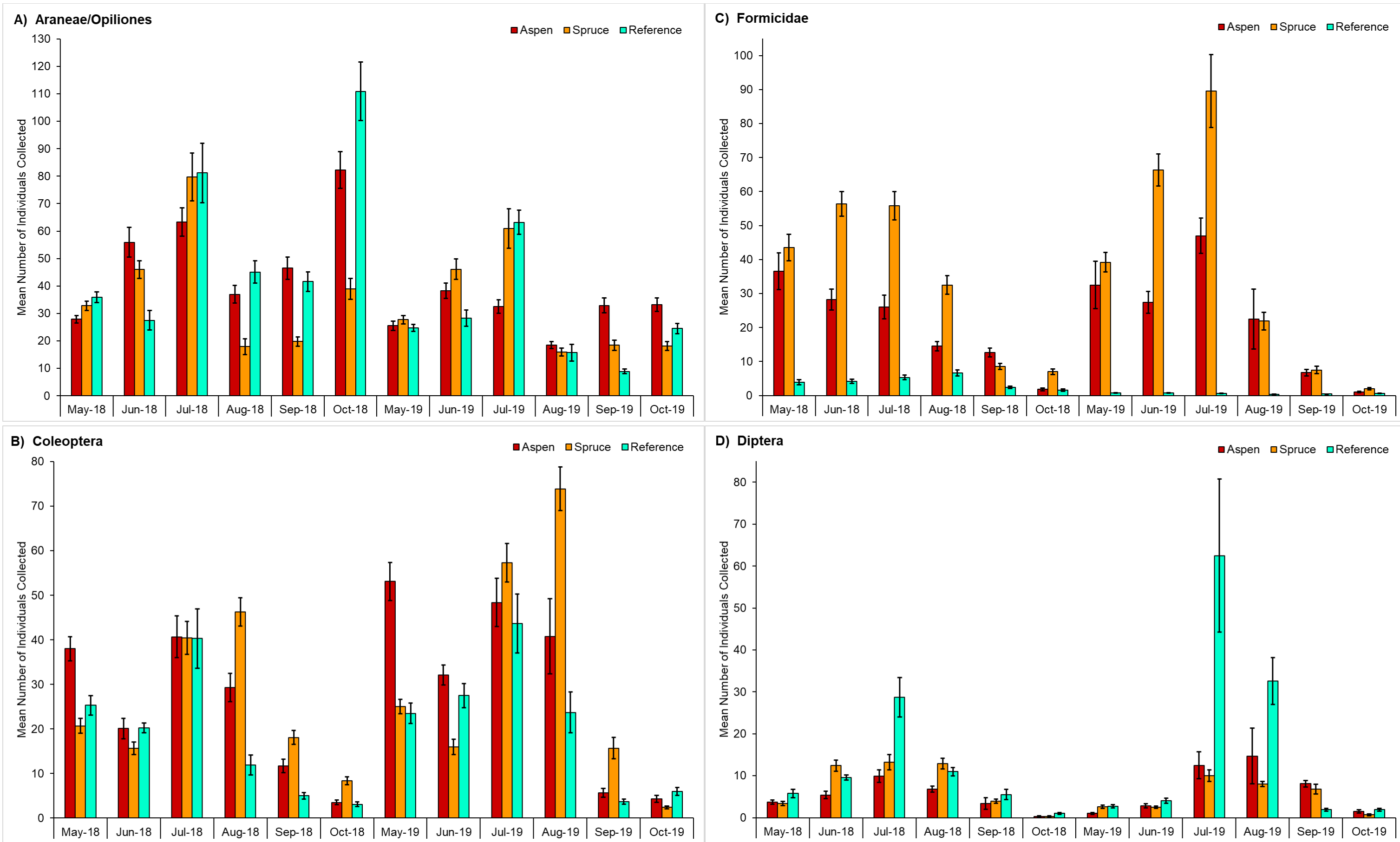

Figure 3.4. Mean number of A) Araneae/Opiliones, B) Coleoptera, C) Formicidae, and D) Diptera per pitfall trap collected monthly. Error bars = standard error.

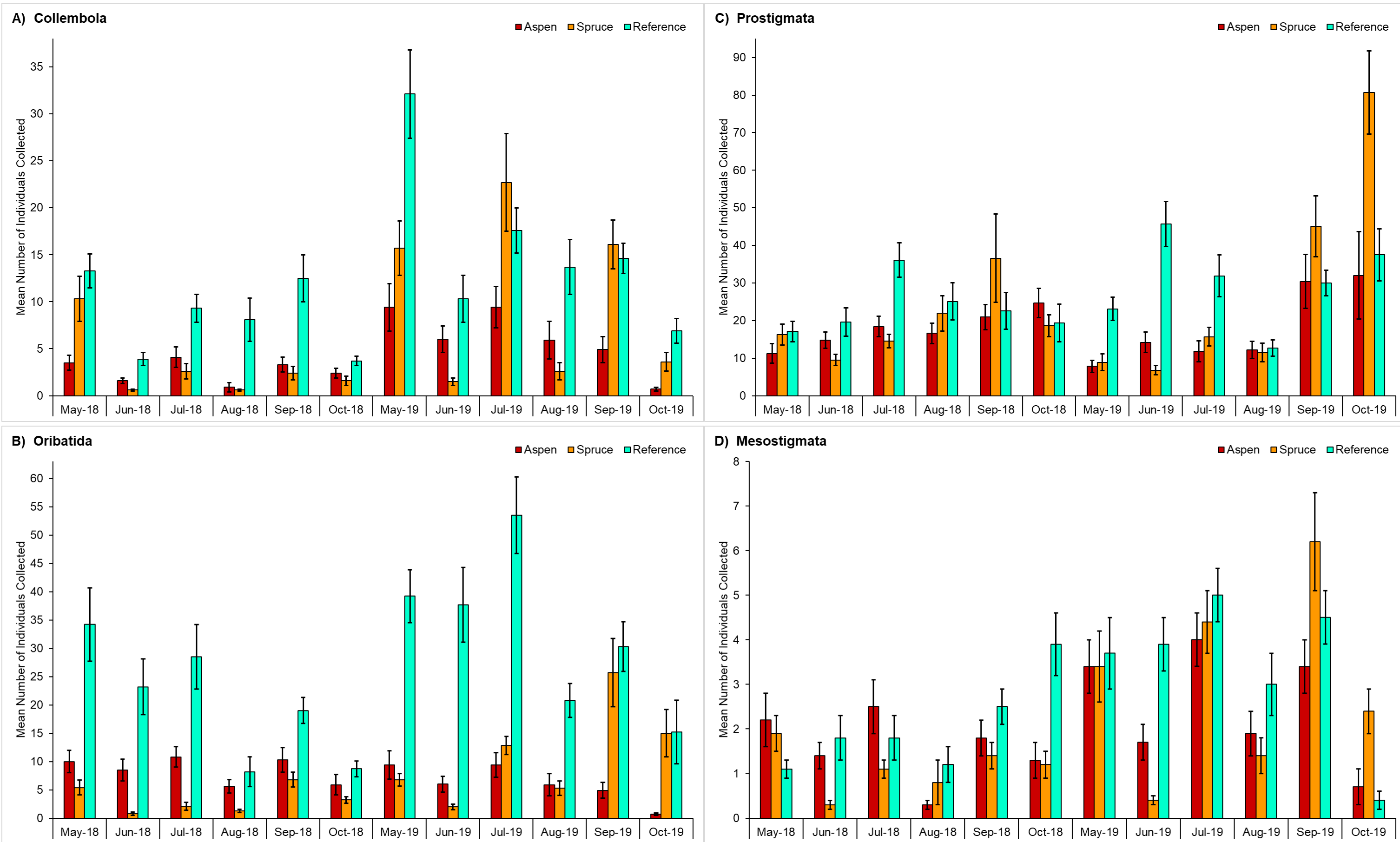

Figure 3.5. Mean number of A) Collembola, B) Oribatida, C) Prostigmata, and D) Mesostigmata per litter sample collected monthly. Error bars = standard error.

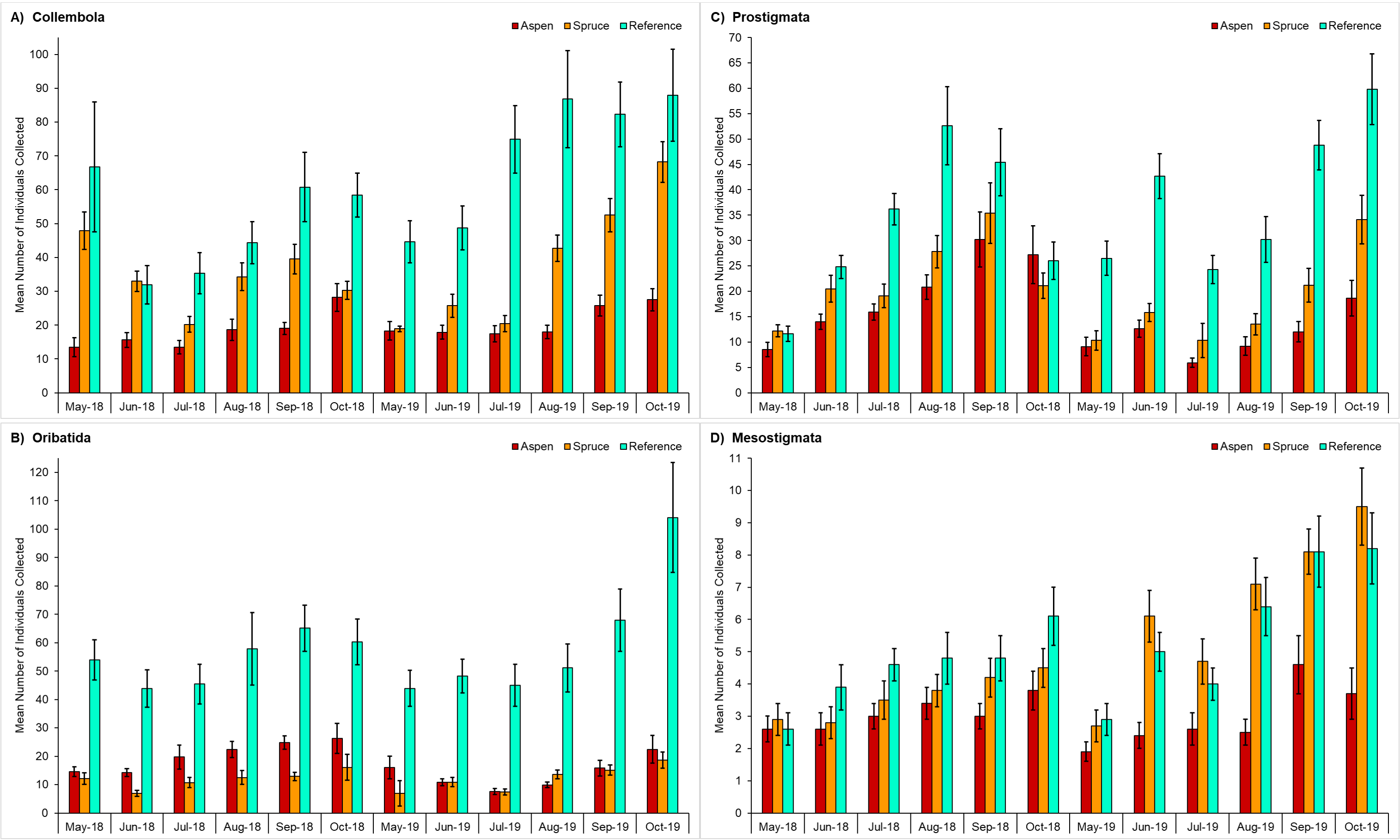

Figure 3.6. Mean number of A) Collembola, B) Oribatida, C) Prostigmata, and D) Mesostigmata per soil core sample collected monthly. Error bars = standard error.

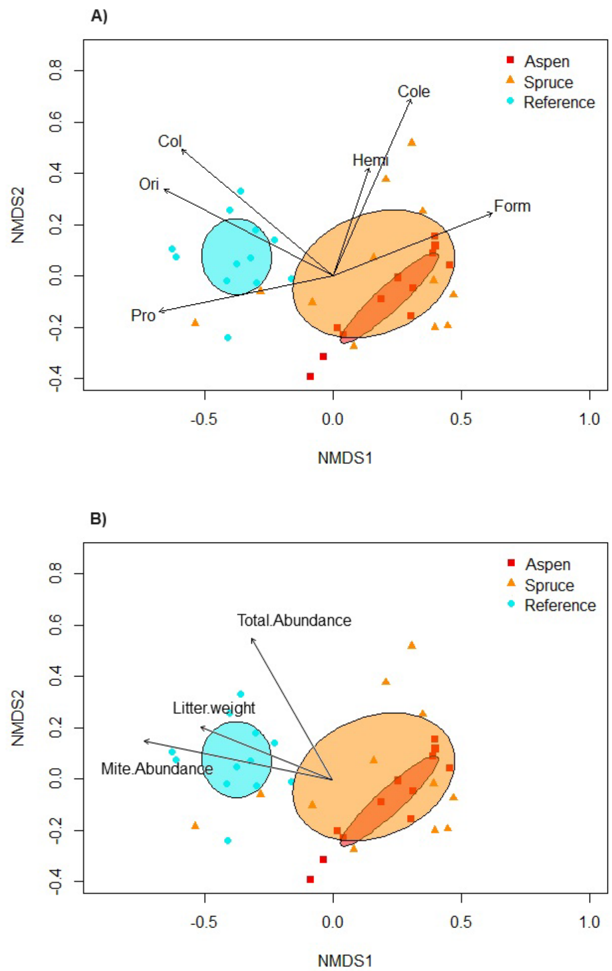

Figure 3.7. Non metric multidimensional scaling (NMDS) ordination with a Bray-Curtis distance measure of soil invertebrate abundance and assemblages. Samples collected monthly from May to October 2018 and 2019, each data point for each sampling date = 30 samples per method (pitfall, litter, soil), for a total of 90 samples per point. Ellipses are 95 % confidence intervals and the angle and length of the vectors indicate direction and strength of association with the ordination axis for A) taxa indicators, and B) environmental indicators. Taxon abbreviations are: beetles (Cole), ants (Form), true bugs (Hemi), springtails (Col), oribatid mites (Ori), and prostigmatid mites (Pro).

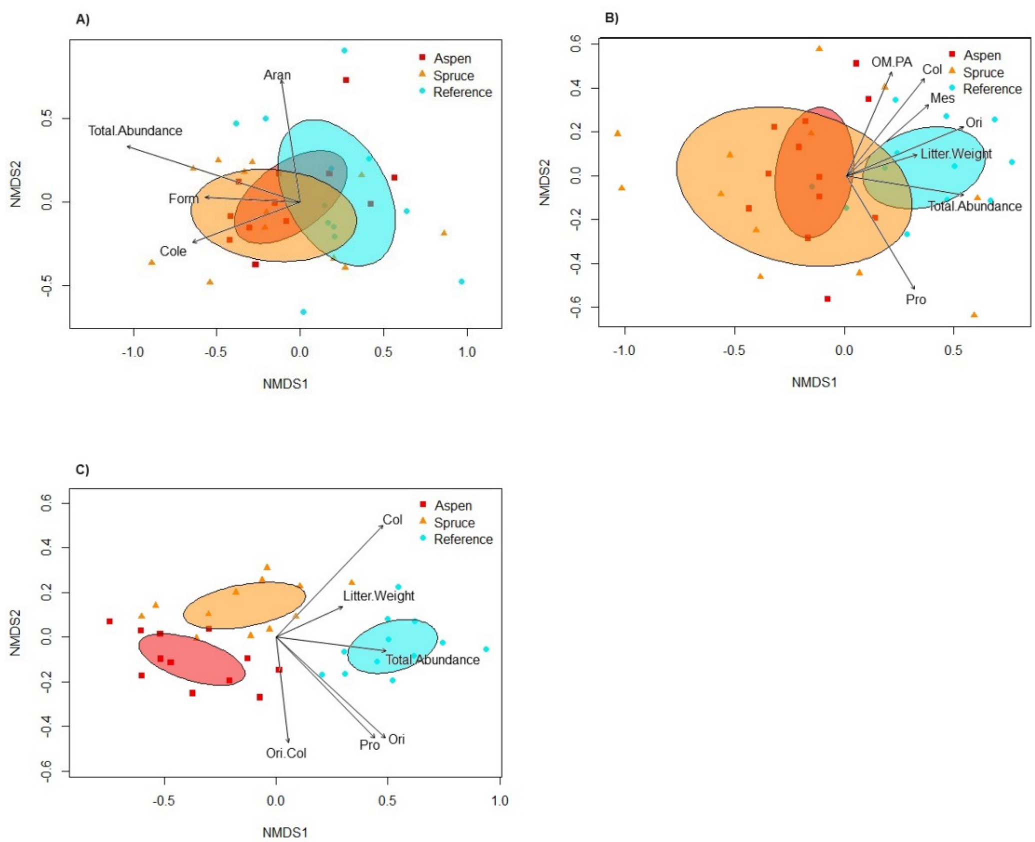

Figure 3.8. Non metric multidimensional scaling (NMDS) ordination with a Bray-Curtis distance measure of monthly abundance and assemblages of (A) above ground invertebrate taxa, (B) litter dwelling invertebrate taxa, and (C) soil dwelling invertebrate taxa. Samples collected monthly from May to October 2018 and 2019, each data point for each sampling date = 30 samples. Ellipses are 95 % confidence intervals and the angle and length of the vectors indicate direction and strength of association with the ordination axis. Abbreviations are as follows: spiders and harvestmen (Aran), beetles (Cole), ants (Form), springtails (Col), mesostigmatid mites (Mes), oribatid mites (Ori), prostigmatid mites (Pro), oribatids + mesostigmatids/ prostigmatids + astigmatids (OM.PA), and oribatid mite to springtail ratio (Ori.Col).

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CHAPTER IV: BEETLE (COLEOPTERA) RESPONSE TO FOREST RECLAMATION METHODS AFTER COAL MINING

1. INTRODUCTION

Alberta has extensive coal and oil sands reserves that are accessed through strip or open pit surface mining or in situ operations. While more than 75 % of land disturbed by coal mining has been reclaimed (Coal Association of Canada, 2018), most end land use is cropped agricultural lands, while reforestation is an ongoing challenge (MacKenzie et al., 2011). Numerous hectares of oil sands mining are to be reclaimed. Thus optimizing reclamation methods is important for reclaiming disturbed areas to create resilient and functioning ecosystems. In Alberta, land reclamation is required under the Environmental Enhancement and Protection Act (EPEA), with energy companies legally required to return disturbed land to equivalent land capability (Alberta Energy Regulator, 2020). Land reclamation should aim to restore biodiversity and function for sustainable land use. Criteria and indicators for land reclamation need to be ecologically robust, have quantitative and objective metrics, and facilitate evaluation of ecosystem recovery dynamics (Baah-Acheamfour et al., 2022; Lupardus et al., 2020, 2019; McIntosh et al., 2019).

Strong ecological indicator taxa are abundant, occur frequently, and are consistently associated with specific environmental conditions (Dufrêne and Legendre, 1997). Practical indicator species are reliably present in a specific habitat and are restricted to that habitat type, thus showing high fidelity and specificity (Pohl et al., 2007). Beetles (Coleoptera) have been extensively researched and discussed as indicators in relation to sustainable ecosystem management and habitat restoration (Borchard et al., 2014; Evans et al., 2019; Hodecek et al., 2016; Koivula, 2011; Makwela et al., 2023; Pearce and Venier, 2006). Researchers have explored beetle responses to disturbances such as clearcut forest harvesting (Niemela et al., 1993), variable retention forest harvesting (Lee et al., 2023; Thomas et al., 2009; Wu et al., 2020), and non-crop agriculture (Rischen et al., 2022). Beetles are one of the most well researched invertebrate taxa with well established taxonomic keys and identification resources. Although knowledge of Canadian beetle diversity has increased, significant contributions can still be made, as most biomes in Canada are only superficially sampled, especially those in central and western Canada. The total estimated undescribed and unreported beetle species is estimated at 1080 to 1280 (Brunke et al., 2019).

There is potential for beetles to be used as an indicator in land reclamation and environmental monitoring. Epigaeic beetle assemblages are predominantly composed of ground beetles

(Carabidae) and rove beetles (Staphylinidae). Carabid beetles are a well studied arthropod group, adults of all described species are well characterized, and they are abundant and diverse in agricultural and forest ecosystems. However, as accurate species level identification requires specialized training, other biodiversity measures such as ecological land classification are typically deemed more appropriate environmental indicators (Goulet, 2003; Langor and Spence, 2006). An assessment on the use of carabid beetles as indicators of forest management effects across Canada found carabid beetles better suited for fine scale local evaluations than for regional and national monitoring (Work et al., 2008). Carabid beetles were not good indicators of disturbance at larger landscape scales and showed little response to forest habitat fragmentation (Pearce and Venier, 2006). Staphylinid beetles are highly diverse, occupy numerous microhabitats, are sensitive to habitat change, and are affected by small and large scale disturbances (Klimaszewski et al., 2018). Staphylinid beetle assemblages are indicators for monitoring coal restoration sites in Western Siberia and have been used to inform forest management and reforestation practices in conifer forests in southwestern China (Luo et al., 2013; Luzyanin et al., 2023). In agroecosystems, staphylinid beetles are important biological control agents against pests, as generalist predators for widespread application, or highly specific parasitoids for dipteran hosts (Klimaszewski et al., 2018). Staphylinid beetles can enhance our understanding of biodiversity responses to cumulative disturbance effects, specifically wildfire and linear disturbances in boreal peatlands (Wu and Pinzon, 2022).

There is limited knowledge on beetle dispersal and succession in reclaimed systems and seasonal dynamics of beetle species. Our understanding of reclamation method effects on beetle abundance and assemblages is incomplete. Research examining effects of oil sands mine reclamation on terrestrial arthropod communities found ground and rove beetle assemblages in reclaimed sites dominated by small to medium sized, open habitat eurytopic species; burned and mature forest sites had larger forest species (Hammond et al., 2018). Rove beetles in reclaimed borrow pits appeared in arrested succession near Cold Lake and responded differently than ground beetles; rove beetle catches were lower, while species diversity was higher with intensity of soil disturbance (Hammond et al., 2022). There is potential for both carabid and staphylinid beetles to be used aa indicators in land reclamation and environmental monitoring in Alberta.

To build on previous beetle research in land reclamation in Alberta, our research objectives were: i) to evaluate effects of land reclamation methods (direct soil placement and natural revegetation versus stockpiled soil and planting) on beetle assemblages , diversity, and abundance; ii) to evaluate catch rates and patterns of abundant beetle species and beetle species of interest over the growing season (May to October); and iii) to assess beetle indicators and whether it is practical and feasible to incorporate Coleoptera into current reclamation monitoring criteria.

2. MATERIALS AND METHODS

2.1. Research Sites

This study was conducted at the Genesee coal mine and generating station, 80 km west of Edmonton, in Alberta's dry mixedwood natural subregion (Beckingham and Archibald, 1996). This area is characterized by mean annual temperature of 2.4 °C and annual precipitation of 434 mm (Government of Canada, 2023). Most vegetation at the Genesee site has been cleared for agricultural production; isolated pockets of aspen stands remain, and wetlands are located in the area but not hydrologically connected to the site (The Alberta Utilities Commission, 2014). Surface water runoff is contained and directed to an effluent settling pond then into a cooling pond.

Sites were selected based on reclamation prescriptions, end land use, and proximity to an undisturbed forest for a reference site. Reclamation sites were named after respective tree species, Aspen and Spruce. The proximate undisturbed reference site (hereafter Reference) was characterized as a d1 ecosite and deciduous community type, with a medium nutrient regime and mesic hydrologic regime, and a reference plant community that includes trembling aspen (*Populus tremuloides*) as the dominant tree species, low bush cranberry (*Viburnum edule*) and prickly rose (*Rosa acicularis*) in the understory, and greater proportion of low versus tall forb cover (Willoughby et al., 2021). Aspen and Spruce had different soil reconstruction and revegetation methods (Table 4.1); reclamation prescriptions were not replicated elsewhere on site.

2.2. Beetle Collection

Ten plots (5 x 5 m) were established at each site, ensuring similar drainage and slope, with at least 5 m buffer between plots and at least 10 m from the reclamation site boundary to limit edge effects. Plots were divided into 25 subplots. Vegetation assessments of cover by species and soil cores (0 to 15 cm, 16 to 30 cm) for soil chemical and physical analysis were conducted in July 2017 on five randomly located subplots at each site. Vegetation assessments were completed in 2018 and 2019. Key soil and vegetation results were summarized for each site (Table 4.1).

After soil core collection, the hole was used to install pitfall traps, clear 16 oz plastic containers (height 7.62 cm, top diameter 11.75 cm), with 200 mL of propylene glycol, and a styrofoam plate roof (diameter = 17.1 cm) anchored with bamboo skewers < than 3 cm above the trap. Pitfall traps were left in the field for one week and then collected. Monthly pitfall trap sampling started in May 2018 and was carried out until October, and repeated in 2019. Samples were collected from three randomly selected pitfall traps, for a total of 30 samples per site, per month. Pitfall traps were opened on the second Thursday of the month for both 2018 and 2019. Sampling dates in 2018 were May 10, June 7, July 12, August 9, September 6, and October 18. Sampling dates in 2019 were May 9, June 13, July 11, August 8, September 12, and October 10.

Individuals collected in pitfall traps were sorted into coarse taxonomic categories and beetles were stored in separate containers with 75 % isopropyl alcohol for preservation and further taxonomic identification. All identifications were completed by Gerald Hilchie and voucher specimens were deposited at the Strickland Entomological Museum, at the University of Alberta.

2.3. Statistical Analyses

Some pitfall traps were lost to disturbance; therefore traps were standardized to 25 per month. Means and standard error for monthly catches and species richness, evenness, and diversity were calculated. Specimens of the subfamily Aleocharinae (6.83 % of total catch, 57.99 % of Staphylinid catch) were included in abundance and diversity calculations, but were excluded from further analysis due to taxonomic impediments to species level determination (Lee et al., 2023).

2.3.1. Beetle alpha diversity

We calculated beetle alpha diversity indices including species richness, Pielou's evenness (J'), Shannon diversity index (H'), and Simpson diversity index (D) as follows.

$\mathrm{J}' = \frac{\mathrm{H}'}{\mathrm{H}'\mathrm{max}}$ and $\mathrm{H}'\mathrm{max} = \ln(\mathrm{S})$, where S is species richness.

$\mathrm{H}' = -\sum \mathrm{p} \ln(\mathrm{p})$, where p is the proportion of individuals of each species in a community.

$\mathrm{D} = 1 - \frac{\sum \mathrm{n}(\mathrm{n}-1)}{\mathrm{N}(\mathrm{N}-1)}$, where n is the number of individuals of each species and N is the total number of individuals of all species.

Shannon diversity index incorporates species richness and evenness. Evenness accounts for relative abundance of each species in a community. Low evenness suggests dominance by one or more species, while high evenness indicates a more balanced distribution where species occur in comparable numbers. Evenness and Simpson's diversity index are constrained between 0 and 1, where 0 means that there is no evenness and low diversity, and 1 means that all species occur in equal abundance or there is high diversity.

2.3.2. Rarefaction and total species richness

Further analyses were performed with the R statistical package (version 4.1.3) (R Core Team, 2023). Rarefaction curves calculate the expected number of species as a function of sampling, number of individuals or sampling units, allowing comparisons of species richness across sites (Gotelli and Colwell, 2001). Rescaling sample based rarefaction to individual based rarefaction will show if the species richness estimates are sensitive to the number of individual beetles collected. We used the rarefy function in the vegan package, multiples of 200, and total number of individuals for each site. We used the specpool function in the vegan package to calculate incidence based estimators of total species richness, Chao2 (S_2) and Jacknife (S_{jack}), for each of the sites as follows.

$S_2 = S_{obs} + \frac{(Q1)^2}{2(Q2)}$, where S_{obs} is the number of species in the sample, Q1 is the species occurring in only one sample (singletons) and Q2 is the number of species occurring in two samples (doubletons) (Chao, 1987; Colwell and Coddington, 1994).

$S_{jack} = S_{obs} + [\frac{Q1(2m-3)}{m} - \frac{Q2(m-2)^2}{m(m-1)}]$, where m represents the total number of samples (Smith and van Belle, 1984).

2.3.3. Multivariate analysis of beetle assemblages

Beetle assemblages were visualized using nonmetric multidimensional scaling (NMDS) unconstrained ordination with metaMDS function from the vegan package, and Bray-Curtis as the distance measure. Beetle data were Hellinger transformed with the decostand function in the vegan package, due to low counts and many zeros (Legendre and Gallagher, 2001). The number of dimensions were determined using the dimcheckMDS function in goeveg and evaluating scree plots, and reduction in stress with decreasing dimensionality (McCune and Grace, 2002). Significant correlated species were visualized on the ordination plot, using the vec and envfit functions in the vegan package (alpha = 0.001, permutations = 999). Species vectors were displayed when Pearson R was greater than 0.3.

Permutational Multivariate ANOVAs (perMANOVA) were performed with adonis in vegan, to determine significant differences of beetle assemblages among sites (alpha = 0.001, permutations = 999). P values were corrected by Bonferroni for repeated tests using the RVAideMemoire package. Differences among sites in assemblages were evaluated with homogeneity of dispersions tests (betadisper), followed by ANOVAs to compare mean distance-to-centroid assemblages among sites. Significant ANOVAs (alpha < 0.001) were followed by Tukey's HSD post-hoc test to identify pairwise differences in assemblages among sites.

2.3.4. Indicator species analysis

Indicator species analysis (ISA) determines significant indicators based on aspecificity (A), which is the probability that the surveyed site is part of the target site group; and given a species presence and sensitivity (B), probability of finding the species in sites belonging to the target site group. The indicator value index assesses the predictive value of a species as an indicator of a combination of site groups, but does not account for species absences inside and outside the site group combination (De Cáceres et al., 2010). ISA was performed using the multipatt function from the indicspecies package (Dufrêne and Legendre, 1997; De Cáceres and Legendre, 2009) to determine which beetle species were indicators of Reference or reclamation sites. We used a quantitative or binary response with a randomization test (alpha = 0.001, permutations = 999).

3. RESULTS

We collected 25,294 beetles, representing 157 species, with carabid and staphylinid beetles compromising 78.4 % and 11.3 % of the total catch, respectively. Staphylinid beetles were slightly more species rich (44) than carabid beetles (42) (Supplemental Table 4.1). The twelve most abundant species, representing five families, accounted for 88.2 % of the total catch, with *Pterostichus melanarius* (Illiger 1798) accounting for a majority of the total catch at 59.0 %. The next most abundant species, *Carabus granulatus (*Linneaeus 1758) accounted for 8.6 % of the total catch, followed by an *Aleocharinae* spp. accounting for 4.5 % of the total catch. The remaining species include *Catops americanus* (Hatch 1928) (3.2 %), *Lordithon fungicola* (Campbell 1982) (2.7 %), *Calanthus ingratus* (Dejean 1828) (2.6 %), *Hypnoidus bicolor* (Eschscholtz 1829) (1.7 %), *Pterostichus pensylvanicus* (LeConte 1873) (1.4 %), *Platynus decentis* (Say 1823) (1.2 %), *Sitona lineellus* (Bonsdorff 1785) (1.2 %), *Amara cupreolata* (Putzeys 1866) (1.1 %), and *Poecilus lucublandus* (Say 1823) (1.1 %).

Singletons species are collected once; doubletons are collected twice. We collected 33 singleton species (21 % of total) and 20 doubleton species (12.7 % of total) (Supplemental Table 4.1). Reference had 11 singletons and 4 doubletons, Aspen had 10 singletons and 3 doubletons, Spruce had 5 singletons and 4 doubletons. Reference and Aspen shared 3 doubletons, reclamation sites shared 4 doubletons.

Mean number of beetles collected varied by month and site, but generally followed a similar pattern over the two sampling years (Figure 4.1). Catch rates were high in May, decreased in June, peaked in July and August, and were lowest in September and October. Reclamation sites often had higher catch rates than Reference, especially in August. The highest mean number of beetles collected in Aspen was May 2019 and in Spruce was August 2019 (Figure 4.1).

3.1. Beetle Alpha Diversity

Mean species richness did not vary greatly throughout the growing season and was similar among sites, with an exception of higher mean species richness across sites in May and July 2019, and in June 2019 in Reference (Figure 4.2A). This is likely due to higher precipitation in June and July 2019 (Figure A.1). Variations in mean Shannon-Wiener diversity index over the growing season had interesting trends. Reference and Spruce had similar diversity indices in May, June, and July, and Aspen consistently had a lower diversity index (Figure 4.2B). Diversity indices were low for both reclamation sites in August of both years. In August, beetle diversity in Reference was significantly higher than in reclamation sites (Figure 4.2B). Species evenness was significantly different between reclamation sites and Reference in August (Figure 4.2C).

3.2. Rarefaction And Total Species Richness

Based on rarefaction curves and total species richness estimators, we underestimated total species richness in our sites (Figure 4.3, Table 4.2). Our results suggest that approximately 52 additional species are expected in Aspen, 33 in Spruce, and 41 in Reference (Table 4.2). These results are not unexpected as pitfall trapping is not an appropriate collection method for all beetle species. Total species richness estimates are lowest in Spruce and highest in Aspen.

3.3. Beetle Assemblages

Unconstrained ordination illustrated separation of beetle assemblages between reclamation sites and Reference. The NMDS two-dimensional solution had a final stress of 0.149 and the best solution was repeated after 20 tries (Bray-Curtis; non-metric fit, $R^2 = 0.978$; linear fit, $R^2 = 0.899$). The Reference 95% ellipse did not overlap with either reclamation site, while reclamation site ellipses had some overlap, indicating similarity in beetle assemblages in reclamation sites (Figure 4.4). Similar size ellipses indicate similar variation in beetle assemblages over a growing season.

Significant beetle species vectors ($p = 0.001$) were grouped by correlation strength (Pearson R); high correlation was greater than 0.6, while low correlation was less than 0.5. *Pterostichus pensylvanicus* and *Catops americanus* were highly correlated species vectors associated with Reference; *Dicheirotrichus cognatus* (Gyllenhall 1827) and *Pterostichus melanarius* were associated with reclamation sites (Figure 4.4A). *Dicheirotrichus cognatus* and *Pterostichus melanarius* vectors were mirror reflections. *Tachyporus inornatus* (Campbell 1979) was not strongly associated with any site (Figure 4.4A). Vectors with lower correlation included three species associated with Reference: *Hippuriphila mancula* (LeConte 1861), *Platynus decentis*, and *Tachinus fumipennis* (Say 1832) (Figure 4.4B). Four species vectors were associated with reclamation sites: *Pycnoglypta lurida* (Gyllenhall 1813), *Notiophilus aquaticus* (Linnaeus 1758), *Pterostichus femoralis* (Kirby 1837), and *Sitona lineellus* (Figure 4.4B). *Pterostichus melanarius* had the highest correlation (Pearson R = 0.894) followed by *Pterostichus pensylvanicus* (Pearson R = 0.707). An unknown *Aleocharinae* sp. was one of 12 abundant beetle species (Figure 4.4).

There were significant effects of site and month ($p = 0.001$), but not year (Table 4.3), and no significant interactions among year, month, and site. Post-hoc testing found beetle assemblage in Aspen and Spruce significantly different from Reference ($p = 0.003$); differences between reclamation sites were not significant (Table 4.3). Post-hoc testing showed beetle assemblages did not differ significantly among months (Table 4.3). While not significant, beetle assemblages in October were different from May, July, and August, likely due to lower temperatures. Dispersion of beetle assemblages did not differ by site (betadisper, ANOVA, df = 2, $F = 0.60$, $p = 0.555$).

3.4. Indicator Species Analysis

Indicator species analysis (ISA) considered 118 species; 20 species were selected (Table 4.4). Four species were significant indicators for both reclamation sites, including *Amara cupreolata, Poecilus lucublandus*, *Agonum cupreum*, and *Sitona lineellus*. Aspen had one indicator species, *Bembidion acutifrons,* Spruce had two, *Notaris puncticollis* and *Cymindis cribricollis* (Dejean 1831). Beetle indicator species for Reference had relatively high aspecificity, five species had scores of 1; sensitivity varied considerably. Five beetle species indicators for Reference with high indicator scores and strong significance ($p = 0.001$), included: *Tachinus fumipennis*, *Loricera pilicornis*, *Omosita colon*, *Quedius simulator*, and *Dorytomus parvicollis*. Interestingly, ISA found three species indicators for Reference + Aspen, which included *Catops amercianus*, *Atomaria ephippiata*, and *Bembidion quadrimaculatum*. There were no indicator species for Reference + Spruce. More interesting is that *Pterostichus melanarius* was not a significant indicator.

3.5. Relative Abundance Of Major Beetles Species

Examination of the relative abundance of the 12 most abundant species and remaining other beetle species over the growing season from May to October, showed a majority of the abundant species were primarily captured in Reference (Figure 4.5). Aspen and Spruce had consistent high relative abundance of *Pterostichus melanarius*, in most sampling months, especially July, August, and September. *Pterostichus melanarius* relative abundance was lowest in October, and the relative abundance of other beetle species outside of the top 12 was 41 % and 68 % in Aspen and Spruce sites, respectively (2018), and 90 % and 80 % in 2019 (Figure 4.5). Many beetles in the genera *Pterostichus* and *Poecilus* are forest or woodland species associated with dense vegetation, but the most abundant species also frequent cultivated fields (Holliday et al., 2014).

We collected 14,190 individual *Pterostichus melanarius*. Reference contributed 5.9 % to the total *Pterostichus melanarius*, while Aspen and Spruce accounted for 51.4 % and 42.7 %, respectively. This species was collected in all sites, but exhibited preference for reclamation sites, and showed similar trends over the growing season in both years. Mean number of individuals collected in Aspen was higher in May and June, followed by similar catch between reclamation sites in July (Figure 4.6). The highest mean number of individuals collected in each year occurred in August at Spruce. The abundance of *Pterostichus melanarius* populations in the reclamation sites peaked in August, rapidly declined in September, and were near zero in October (Figure 4.6).

Four species increased in abundance in reclamation sites in October once *Pterostichus melanarius* abundance decreased; *Tachyporus inornatus* (Campbell 1979), *Dicheirotrichus cognatus*, *Notiophilus aquaticus*, and *Bembidion acutifrons* (LeConte 1879). *Bembidion* and *Notiophilus* are designated as open habitat specialists (Niemela et al., 1993). The first three were selected as species vectors correlated with beetle assemblages, *Bembidion acutifrons* was an indicator species for Aspen. *Tachinus inornatus* increased in catch counts in October across sites, relatively few individuals were collected each month in each site, and there was a slight increase in catch totals in October for both years; especially in Spruce where 12 beetles were collected in October 2018 (6.8 % of total catch) and 11 beetles in 2019 (14.9 % of total catch) (Figure 4.7A). *Tachinus inornatus* did not show site preferences and recorded large catch rates in Reference in August 2019. *Dicheirotrichus cognatus* had relatively few individuals collected between May and September, and relatively high counts in October in Aspen and Spruce in 2018 and 2019 (Figure 4.7B). In Aspen, *Dicheirotrichus cognatus* made up 25.5 % and 28.3 % of October's total catch in 2018 and 2019, and 35.2 % and 24.3 % in Spruce. This species showed a strong preference for reclamation sites and was rarely captured in Reference. *Notiophilus aquaticus* was collected in all sites and from the 55 individuals collected, 18.2 % was from Reference, 45.5 % from Aspen, and 36.4 % from Spruce. Number of individuals collected monthly was low and no individuals were collected in June 2018 or September 2019 (Figure 4.7C). Number of individuals collected was generally higher in October for both years and reclamation sites. These trends were seen for *Bembidion acutifrons* (Figure 4.7D), which are associated with bare or sparsely vegetated areas near water; they are carnivorous predators on insects and pest insect eggs (Holliday et al., 2014).

Few individuals in the subfamily Aleocharinae were collected in 2018; Reference and Spruce had high collections in July and August 2019 (Figure 4.7B). *Dimetrota* sp. had 109 individuals, 65.1 % in Reference, 14.7 % in Aspen, and 20.2 % in Spruce. We collected 72 individuals of *Aleochara bilineata*, where 94.4 % of individuals were collected in Spruce.

Carabus granulatus, a non-native carabid beetle with significant southern geographic expansion in North America (Liebherr et al., 2023), displayed an interesting distribution among sites (Figure 4.8A). We collected 2,170 individuals, with 26 % from Aspen, 25 % from Spruce, and 49 %, from Reference. This species was almost always collected more in Reference than reclamation sites; however, in May 2018 all sites had similar catch. Catch was high in Reference and Spruce in the following month, while we saw a large decrease in the number of individuals collected in Aspen. In July 2018 we collected the highest number of individuals in Reference, while catch in reclamation sites continued to decrease. Numbers of individuals collected in August to October were relatively low. The number of *Carabus granulatus* collected in 2019 was much lower than 2018, and monthly results were more similar between sites, suggesting a sensitive population (Figure 4.8A). The species is active in the earlier months of the growing season, May to July, and does not exhibit a clear preference for undisturbed versus reclaimed, as observed in other non-native species such as *Pterostichus melanarius.* Competition from *Pterostichus melanarius* could explain lower catch rates in reclamation sites and higher catch rates in Reference.

Four major beetle species had a clear preference for Reference: *Catops americanus, Lordithon fungicola, Pterostichus pensylvanicus*, and *Platynus decentis*. *Pterostichus americanus* and *Lordithon fungicola* were predominately collected in Reference; we collected relatively high numbers in Aspen during some months, likely due to proximity to Reference. Catch rates of *Catops americanus* were higher earlier in the growing season and tapered off in September and October (Figure 4.8C). Large peaks in July 2018 and May 2019 were observed in Reference. catch rates for *Lordithon. fungicola* peaked in July, and similar to *Catops americanus*, had much higher catch rates in 2019 than 2018 (Figure 4.8D). *Pterostichus pensylvanicus* and *Platynus decentis* were rarely collected in reclamation sites. Catch rates of *Pterostichus pensylvanicus* were high in May and to a lesser extent June, collection in July was rare, and consistent collection in August, September, and October (Figure 4.8E). Catch rates of *Platynus decentis* were highest in May and June, individuals were rarely collected from July to October (Figure 4.8F).

Calathus ingratus exhibited no clear site preference. We collected 617 individuals, 25.1 % from Reference, 21.6 % from Aspen, and 53.3 % from Spruce. In some months the highest number of individuals was in Reference and in other months in Spruce. Large catch peaks were observed at all sites, being highest in Spruce in July of both years (Figure 4.8G). Another interesting trend is significantly higher collection numbers in August at Spruce than Reference and Aspen. It is likely that site water levels and precipitation events contributed to the number of *Calathus ingratus* beetles collected. *Calathus ingratus* is a generalist forest species and dominated beetle assemblages in logged spruce stands in Quebec (Saint-Germain et al., 2005).

Four major beetle species showed preference for the reclamation sites, particularly Spruce, including: *Hypnoidus bicolor*, *Sitona lineellus*, *Amara cupreolata*, and *Poecilus lucublandus*. *Hypnoidus bicolor* was primarily collected in Spruce early in the growing season, with none collected in September or October (Figure 4.8H). A large peak number from Spruce and Reference was in July 2019, likely related to higher precipitation. *Hypnoidus bicolor* is a common pest in the prairies (Drahun et al., 2021). *Sitona lineelus*, an alfalfa weevil, had unremarkable catch rates in 2018, with no clear differences among months and sites, although consistently collected in reclamation sites (Figure 4.8I). In May 2019 there was a peak in numbers collected, highest in Spruce. A small number of individuals were collected in June, none from July to September, and one in Spruce in October (Figure 4.8I). *Amara cupreolata*, a seed eating ground beetle, had only 3 individuals (out of 218) collected in Reference, while 30.3 % was collected from Aspen, and 68.3 % from Spruce. In 2018, catch rates in Spruce were high in May and June, decreasing for the remainder of the growing season (Figure 4.8J). Catch rates in May 2019 were high in reclamation sites, higher in Aspen, while the remainder of the growing season had low, but relatively consistent catch rates in Spruce and Aspen (Figure 4.8J). *Amara cupreolata* was only collected in Reference in July 2019. *Poecilus lucublandus* was not collected in Reference, Aspen represented 26.2 % of individuals collected, and Spruce 73.8 %. In some months catch was higher in Aspen than Spruce, but was generally higher in Spruce, and higher in 2019 (Figure 4.8K). Site preference and changes in catch over the growing season could be related to site water content and precipitation events. *Poecilus lucublandus* has diverse habitat associations and diet, adults feed on insects, plants, and fungi, while larvae are carnivorous (Holliday et al., 2014).

3.6. Other Beetle Species Of Interest

Other beetle species had most individuals in Reference. *Caenocelis parallela* (Casey 1900) had 12/13, *Hippuriphila mancula* 16/17, and *Tachinus fumipennis* 106/107 in Reference. This shows the importance of proximity to undisturbed sites for dispersal of forest beetle species which contributes to ecological succession and ecosystem recovery after disturbance.

Olibrus semistriatus (LeConte 1856) is an abundant pollen feeding beetle found in several Aster species (Majka et al. 2008). Only one individual was collected in Reference, 30 in Aspen, and 17 in Spruce, out of a total of 48. *Byrrhus americanus* (LeConte 1850), a pill beetle that prefers damp habitats, was not collected in Reference; two were collected in Aspen and 15 in Spruce.

Three beetle species were only collected in Reference: *Calosoma frigidum* (Kirby 1837) (40 individuals), non-native *Loricera pilicronis* (Fabricius 1775) (20 individuals), and sap beetle *Omosita colon* (Linnaeus 1758) (13 individuals). Two beetle species were only collected in Spruce: *Chlaenius purpuricollis* (Randall 1838) (7 individuals) and *Notaris puncticollis* (LeConte 1876) (9 individuals). Beetle species in the genus *Chlaenius* are associated with dense vegetation in agricultural and woodland habitats (Holliday et al., 2014).

4. DISCUSSION

Our study evaluated the effects of land reclamation methods, direct soil placement with natural revegetation versus stockpiled soil with planting, on beetle assemblages, diversity, and abundance. Results showed that beetle diversity and abundance were generally comparable between reclamation sites and Reference. However, beetle assemblages were significantly different between sites. Beetle assemblages in reclamation sites were homogenized and differed markedly from Reference. Differences between reclamation sites could be attributed to variations in reclamation methods, site soil water content, and proximity to reference areas, but our analysis could not determine which factor had the most significant impact. Catch rates and patterns of abundant beetle species varied over the growing season, with higher abundance and diversity earlier in the season and lower catch rates in September and October. Notably, catch rates in reclaimed sites were significantly higher in August, although species evenness and diversity indices were lower compared to Reference. These results suggest that using species-level taxonomy is more appropriate for beetle reclamation indicators than higher-level taxonomic groupings. However, the complexity and time required for species level identification present challenges for reclamation practitioners, limiting practicality of using beetle species as reliable indicators. A more practical approach to using beetle species as indicators in reclamation would involve identifying key species that are ecologically important, easy to catch in pitfall traps, easily identifiable, or known to be abundant in reclamation areas, similar to *Pterostichus melanarius*.

4.1. Beetle Response To Reclamation

Beetle assemblage differences in reclamation and Reference sites were expected, given site age, high graminoid cover, and early successional stages. Rove beetle response to variable retention harvesting at the Ecosystem Management Emulating Natural Disturbance (EMEND) experimental site, 1, 2, 11, and 16 years post-harvest in replicated stands of four cover types, showed community composition at year 16 converged towards a common composition and structure across harvest treatments and controls (Lee et al., 2023). Ground beetle assemblages were better retained and recovered more quickly with retention harvesting than clear cutting, and 15 years after harvest, assemblages were recovering towards pre-harvest conditions (Wu et al., 2020). Time since reclamation for Aspen and Spruce is 12 and 8 years, respectively; more time is apparently needed for beetle assemblages to recover. With tree growth, canopy closure, and a growing shrub layer, beetle assemblages in Aspen and Spruce should become more similar to Reference, as shown by (Belluz et al., 2022) that beetle assemblages in regenerating clearcut and wildfire lodgepole pine stands in western Alberta, stabilized near the time of canopy closure.

Similarity of beetle species richness and diversity among the research sites shows response to reclamation treatments and successional progression, which is reflected in other research outcomes. Open habitat and generalist ground and rove beetle species respond positively to forest harvest disturbances, followed by declines with site regeneration (Lee et al., 2023; Niemela et al., 1992; Wu et al., 2020). *Pterostichus melanarius*, which dominated reclamation sites, is a large non-native ground beetle and arthropod predator. Other beetles included common pest species, weed seed predators, general woodland species, and species associated with agricultural systems or vegetation. Ground and rove beetle assemblages in reclaimed sites were dominated by small to medium sized open habitat eurytopic species, while burned and mature forest sites had larger forest species (Hammond et al., 2018). The higher species richness estimates in Aspen could indicate our trap locations did not adequately convey available epigaeic beetle habitat and site variability. Aspen encompasses two ecosystems: a grassland with abundant low height vegetation and an emerging upper layer, along with areas of higher shrub cover forming an understory and the beginnings of tree cover. This diversity may attract open habitat beetle species and forest species, potentially contributing to the high species richness estimate. Low species richness estimates for Spruce align with our findings, as we observed high abundance of non-native grass species and extensive weed cover.

We found evidence that some species were dispersing from Reference to Aspen. We expected some similarities in beetle assemblages between Reference and Aspen given direct placement of forest soil and litter in Aspen, and its proximity to Reference for forest species dispersal. Mature forest reserves on post-harvest landscapes ensure beetle refuges to colonize regenerating forest stands (Wu et al., 2020). In western Canada, carabid beetle response to silvicultural and disturbance factors was consistent, no clear association between frequency-abundance and body size and/or dispersal ability was found (Work et al., 2008). There is little evidence to support or discount the importance of dispersal in maintaining carabid populations (Work et al., 2008). Unmeasured and uncontrolled factors that differ in Aspen and Spruce mean we cannot definitively determine why beetle assemblages are different between reclamation sites. The presence of Reference species in Aspen may stem from dispersal. Further investigation is warranted to elucidate the role of forested patches in facilitating soil invertebrate dispersal and subsequent colonization in reclamation areas within extensive mining and reclamation landscapes.

Differences in beetle composition between Aspen and Spruce were likely related to site hydrologic conditions. Spruce is a wetter site with more low lying areas and willow species, higher soil saturation percentage, and close proximity to a large cooling pond. The cooling pond is used as a source and receptor of cooling water for three power generating units, diverting water from the North Saskatchewan River to counter evaporative losses and improve overall water quality (Golder Associates, 2013). Higher soil water content likely explained the population explosion of fungi and associated beetles in coniferous foothills forested stands (Pohl et al., 2007). While our results and trends are interesting, presence and absence from pitfall traps, especially with a low number of individuals, does not always have site wide implications. However, there are some trends that are helpful in understanding beetle assemblages in reclaimed systems forward.

4.2. Beetle Species Of Interest

We determined abundant beetle species and activity patterns over the growing season, identifying 12 abundant species, and others with interesting trends and site preferences. Many beetle species of interest were singled out by other researchers when assessing ecosystem recovery after disturbance. *Lordithon fungicola*, the most abundant rove beetle at EMEND (Lee et al., 2023), was second most abundant in our study. *Lordithon fungicola* has been designated as a young forest specialist, regenerating after harvest and stand age (Pohl et al., 2007). It showed a clear Reference preference; our higher numbers in Aspen than Spruce indicates likely dispersing from Reference. At EMEND, *Tachinus fumipennis*, a mature forest species, decreased in captures immediately post-harvest, while captures increased with increasing retention in the first two years post harvest, potentially an indicator of post-disturbance recovery (Lee et al., 2023). In our research, *Tachinus fumipennis* was a significant Reference indicator and only one individual was collected outside Reference, in July 2019 in Aspen. Presence of a mature forest species suggests Aspen is in a later successional and recovery stage than Spruce.

Two most abundant carabid beetle species at EMEND, *Platynus decentis* and *Calathus ingratus* (Wu et al., 2020), were among our 12 most abundant species. *Calathus ingratus* is a forest generalist not significantly impacted by cutting and *Platynus decentis* can increase in cut areas (Niemela et al., 1992). *Calathus ingratus* is a young forest reclamation species and is not sensitive to soil disturbance, thus unsurprising we collected it in high numbers at Spruce, with strong catch in Aspen. Since *Platynus decentis* was almost exclusively found in Reference, it could be a good indicator species to differentiate young forest reclamation and proximate undisturbed sites.

One of the most common species in reclaimed oil sands sites relative to nearby burned and mature forests was *Poecilus lucublandus* (Hammond et al., 2018). It is a widespread species of woodland ground beetle that prefers open, moderately dry grass dominated areas, and has high capture rates in spring, low in summer months, and high in autumn (Roughley et al., 2010). In our sites *Poecilus lucublandus* is a reclamation associated species, and abundances will likely decrease as reclamation sites advance in forest successional stages, as evidenced in our collecting it only in reclamation sites and its selection as an indicator species for Spruce.

Aleocharinae taxonomic understanding in eastern Canada has increased, while western and northern Canada are still poorly studied (Klimaszewski et al., 2015). Taxonomic limitations in this group discount interesting trends that warrant further exploration. This staphylinid subfamily is an important natural enemy of root maggots (*Delia* spp.) and research on seasonal activity in canola fields in central Alberta found the number of individuals in pitfall traps peaked in July followed by a steep decline back to zero (Broatch et al., 2008). Density likely reflects proximity of canola fields, prey population, and competition within its species and among predators (Broatch et al., 2008).

4.3. Beetle Indicator Species

Determining indicator species for reclamation success was difficult as in our analyses different beetle species were selected for both Aspen and Spruce, and many were selected for Reference. Other studies have found that variation in reclamation methods can cause variation in beetle assemblages (Echiverri et al., 2023; Hammond et al., 2018, 2022); therefore identifying one species for reclamation sites with different methods is not very likely. *Pterostichus melanarius* was dominant in our reclamation sites, but was not considered a significant indicator species. A weak species indicator can be used to discriminate sites when used as part of a larger group of species (Pohl et al., 2007).

An evaluation of carabid beetles as forest change indicators in Canada found three species, *Agonum retractum*, *Calathus ingratus*, and *Platynus decentis*, primarily associated with western stands, and *Pterostichus pensylvanicus, Agonum rectractum* and *Platynus decentis*, were common species associated with deciduous stands (Work et al., 2008). *Pterostichus pensylvanicus* and *Platynus decentis* were significant Reference indicators in our study. Despite a relatively large proportion of carabid species with trans-Canadian distributions, regional and ecosystem differences in composition limit the applicability of individual species as indicators at a uniform national scale (Work et al., 2008). German researchers assessed arthropods across 93 temperate forest sites to determine their suitability as forest type indicators, concluding that while not reliable on a national scale, regional definitions may be appropriate (Gossner et al., 2014).

For beetles to be considered in reclamation monitoring or assessment criteria, sample collection must be straightforward, and any necessary identification should be cost effective and readily accessible. That is a massive hurdle to include any arthropod in reclamation monitoring in Alberta. Instead of conducting a comprehensive identification of all beetles captured in the pitfall trap, we recommend focusing on select species. By refining the focus to select species we can develop resources to guide beetle assessments in reclamation monitoring similar to Guidelines for Reclamation to Forest Vegetation in the Athabasca Oil Sands Region (Alberta Environment, 2010). Based on our study, we propose monitoring the non-native *Pterostichus melanarius*, along with two forest species, specifically *Pterostichus pensylvanicus* and *Platynus decentis*. Monitoring efforts should prioritize detecting a decline in the abundance of dominant non-native species and the subsequent presence and increase of forest dwelling species.

4.4. The *Pterostichus Melanarius* Problem

The density and dominance of *Pterostichus melanarius* in pitfall traps collected at reclamation sites is concerning. *Pterostichus melanarius* is ubiquitous in habitats altered by anthropogenic activity, such as use of fire in biodiversity and conservation management in tallgrass prairie communities, and are a management concern (Roughley et al., 2010). *Pterostichus melanarius* is a palearctic generalist predator, native to Europe, is a valuable natural enemy in agricultural systems, and has an expanding range into the middle of North America (Busch et al., 2021). Its life cycle can be polyvariant, meaning individuals can hibernate at different ontogenetic phases (combine one year development with hibernating larva and two year development with hibernating immature and post generative adults), monovariant life cycles are seen in poor conditions (Matalin, 2006). Over most of the growing season, *Pterostichus melanarius* was dominant in our reclamation sites, especially in the warmest months July and August. European species may constitute as much as 50 % of specimens collected in Canadian agricultural sites (Goulet, 2003). Seasonal dynamics of activity of populations in Moscow Province, Russia were characterized by two peaks in mid June to early July and late July to mid August (Matalin, 2006). Similarly in western Canada, *Pterostichus melanarius* activity peaked June to August (Niemelä et al., 1997). An estimate of the potential range of *Pterostichus melanarius* using maximum entropy modeling found annual mean temperature was the most important climate variable in predicting its presence, followed by annual precipitation (Busch et al., 2021). With hotter and drier summers expected, dominance of this non-native species could extend past October, outcompeting native species and inhibiting beetle diversity in reclaimed areas, having potential long-term implications on beetle assemblages and ecological succession.

Early research in Alberta on *Pterostichus melanarius* assessed population establishment in a natural aspen-poplar forest and effects on native beetle populations, including *Pterostichus adstrictus*, *Pterostichus pensylvanicus, Calathus ingratus*, *Agonum retractum*, *Platynus decentis*, and *Harpalus fulvilabris* (Niemelä et al., 1997). Native beetle species densities and body sizes were not affected, likely due to different peak seasonal activity periods (Niemelä et al., 1997). Wing dimorphism was observed in a recently established grassy population of *Pterostichus melanarius*, that invaded a nearby aspen-poplar forest, changes in wing length implied individuals spread by walking to the forest edge while longer distances to the forest interior was primarily through flight (Niemelä and Spence, 1999). Catches of *Pterostichus melanarius* and native species were positively correlated, suggesting it did not have a strong and consistent negative effect on native beetle species (Niemelä and Spence, 1999). Interspecific interactions did not reduce the probability of *Pterostichus melanarius* establishing populations, and colonization of new areas depended on chance factors that create opportunity to establish populations (Niemelä et al., 1997). A ten year study found *Pterostichus melanarius* had spread through flight 45 to 50 km from Edmonton, Alberta, established populations in natural aspen forests, and is expanding its range faster than any other introduced species (Bourassa et al., 2011). (Niemelä and Spence, 1999) found that native beetle assemblages lacked strong enough biological resistance to prevent *Pterostichus melanarius* from expanding in North America.

In our coal mine reclamation sites in western Alberta, non-native species, including *Pterostichus melanarius*, can pose sitewide challenges requiring ongoing management. Reclamation sites in northern Alberta have not reported high collection rates of *Pterostichus melanarius*, but in areas with established assemblages, understanding ecological succession stages, potential dominance shifts, and the impact of reclamation site parameters like microtopography and soil amendments is important for informed management decisions in forest reclamation sites.

5. CONCLUSIONS

The impact of land reclamation methods, specifically direct soil placement and natural revegetation versus stockpiled soil and planting, on beetle assemblages, diversity, and abundance were similar between reclamation sites, and diversity and abundance were generally similar to Reference. However, beetle assemblages were significantly impacted by land reclamation methods, and differences in site soil water content and proximity to Reference. Beetle assemblages in reclamation sites were homogenized and significantly different relative to the Reference beetle assemblages.

Catch rates and patterns of abundant beetle species and beetle species of interest over the growing season (May to October) varied considerably, yet clear trends emerged. Beetle abundance and diversity was generally higher earlier in the growing season, with lower catch rates in September and October. Relative to Reference, catch rates in August were significantly higher in reclaimed sites, while species evenness and diversity indices were significantly lower.

These results support the use of species level taxonomy rather than higher level lumping of Coleoptera as reclamation success indicators. However, that requirement limits the feasibility of beetle species serving as reliable indicator species due to the difficulty and time required in identifying certain groups to the species level, posing a practical challenge for most reclamation practitioners and monitors.

Tracking the abundance of *Pterostichus melanarius* relative to forest beetle species, shows some promise. Technological innovations such as machine learning algorithms for identification of beetle images (e.g, iNaturalist) could make species level beetle assessment more feasible. Understanding the ecological and functional implications of reclamation sites dominated by a single non-native beetle species is important. Investigating the range and migratory potential of *Pterostichus melanarius* is needed, particularly regarding its expansion into northern Alberta. While not yet a dominant species listed in northern Alberta's reclamation and forestry research, its prevalence in our research sites prompts questions about its range limits and potential risk for young boreal forest reclamation sites.

The implications for reclamation sites under arrested succession are substantial, influencing long-term beetle diversity and assemblages. Hotter, drier months over the growing season, with non-native and dominant species in reclamation sites, suggests potential species loss. Future research should explore timelines of beetle and vegetation succession, particularly when non-native species decrease, and how those timelines are correlated with habitat succession such as increasing canopy cover.

6. TABLES AND FIGURES

Table 4.1. Summary of environmental variables. Soil samples were collected in July 2017 and vegetation assessments were completed in July 2017 to 2019. Values are mean and standard error.

	Reference	Aspen	Spruce
Soil Reclamation, Soil Type	Orthic Gray Luvisol	~ 20 cm directly placed LFH and Ae horizons, coarse woody material, straw 2009	~ 15 cm of salvaged stockpiled topsoil, straw 2010
Soil Properties (0 to 30 cm)			
Texture	Silt loam, Sandy loam, Loam	Loam	Clay loam, Loam
Saturation (%)	78.5 (3.0)	72.6 (2.0)	75.8 (1.4)
Hydrogen Ion Concentration (pH)	5.1 (0.07)	6.8 (0.03)	7.2 (0.02)
Electrical Conductivity (dS m^{-1})	0.54 (0.03)	0.93 (0.03)	0.69 (0.02)
Cation Exchange Capacity (meq/100g)	33.0 (1.6)	29.0 (0.9)	30.1 (0.7)
Total Nitrogen (%)	0.40 (0.02)	0.27 (0.02)	0.25 (0.01)
Total Carbon (%)	5.81 (0.4)	4.45 (0.3)	3.97 (0.2)
Revegetation Method, Forest Type	Aspen and poplar canopy with low bush cranberry and bunchberry understory	Natural succession through seed bank in LFH and Ae horizons	White spruce seedlings planted 2013
Vegetation Properties			
Shrub Cover (%)	81.4 (2.8)	51.3 (2.7)	18.5 (1.0)
Forb Cover (%)	24.7 (1.4)	44.3 (2.6)	26.5 (1.7)
Graminoid Cover (%)	3.4 (0.3)	39.8 (2.0)	49.9 (1.6)
Moss Cover (%)	4.1 (0.7)	6.7 (1.3)	12.8 (1.7)
Weed Cover (%)	1.0 (0.1)	9.2 (0.6)	11.1 (0.7)

Table 4.2. Incidence and frequency based total species richness estimates and standard error.

	Observed Richness	Chao2	Jackknife
Aspen	78	130.9 (25.8)	129.9 (10.8)
Spruce	71	98.8 (14.4)	109.2 (8.5)
Reference	78	115.5 (18.8)	122.2 (10.8)

Table 4.3. Effects of year, month, and study site on beetle assemblages using three-way PERMANOVA. Year is sampling year, 2018 and 2019; month is sampling month, May, June, July, August, September, October; and site is Aspen, Spruce, and Reference. df: degrees of freedom; SS: sum of squares; R^2: coefficient of determination; Pseudo-F: value by permutation. ** p < 0.001, * p < 0.05.

Source	df	SS	R^2	Pseudo-F	P Value	Multiple Comparisons
Year	1	0.30	0.04	1.93	0.061	
Month	6	2.28	0.35	2.50	0.001**	Pairs did not differ significantly; October and August (p = 0.063), October and May (p = 0.063), and October and July (p = 0.063)
Site	2	1.89	0.29	6.73	0.001**	Reference differed from both reclamation sites (p = 0.003*); Aspen and Spruce did not differ (p = 0.267)
Year x Month	5	0.49	0.07	0.64	0.969	
Year x Site	2	0.14	0.02	0.50	0.972	
Month x Site	10	1.15	0.18	1.48	0.065	
Residual	9	0.31	0.20			
Total	35	6.56	1.00			

Table 4.4. Indicator species analysis examining beetle assemblages. Values are beetle species significant at α = 0.05, listed by descending aspecificity (A). Aspecificity is the probability that the surveyed site is part of the target site group, given the species presence; positive predictive value. Sensitivity (B) is the probability of finding the species in sites belonging to the site group. R = Pearson correlation between beetle species and site, bolded values indicate a very strong positive association, correlation > 0.8. A total of 118 beetle species were considered, 20 were selected; ** $p < 0.001$, * $p < 0.05$.

	Species Name	A	B	R	P Value
Aspen	*Bembidion acutifrons*	0.650	0.750	0.698	0.022*
Spruce	*Notaris puncticollis*	1.00	0.333	0.577	0.025*
	Cymindis cribricollis	**0.963**	**0.833**	**0.896**	**0.001****
Reference	***Tachinus fumipennis***	**1.00**	**0.833**	**0.913**	**0.001****
	Loricera pilicornis	1.00	0.500	0.707	0.002*
	Omosita colon	1.00	0.417	0.645	0.005*
	Quedius simulator	1.00	0.333	0.577	0.023*
	Dorytomus parvicollis	1.00	0.333	0.577	0.037*
	Hippuriphila mancula	0.955	0.500	0.691	0.004*
	Platynus decentis	**0.923**	**0.917**	**0.920**	**0.001****
	Philonthus cyanipennis	0.898	0.333	0.547	0.044*
	Pterostichus pensylvanicus	**0.891**	**0.917**	**0.904**	**0.001****
	Agonum retractum	0.807	0.583	0.686	0.003*
Aspen + Spruce	***Amara cupreolata***	**0.983**	**0.792**	**0.882**	**0.001****
	Poecilus lucublandus	**1.00**	**0.750**	**0.866**	**0.001****
	Agonum cupreum	0.955	0.583	0.746	0.003*
	Sitona lineellus	0.976	0.500	0.699	0.013*
Reference + Aspen	***Catops americanus***	**0.938**	**0.875**	**0.906**	**0.002***
	Atomaria ephippiata	**0.887**	**0.750**	**0.816**	**0.028***
	Bembidion quadrimaculatum	0.970	0.500	0.696	0.009*

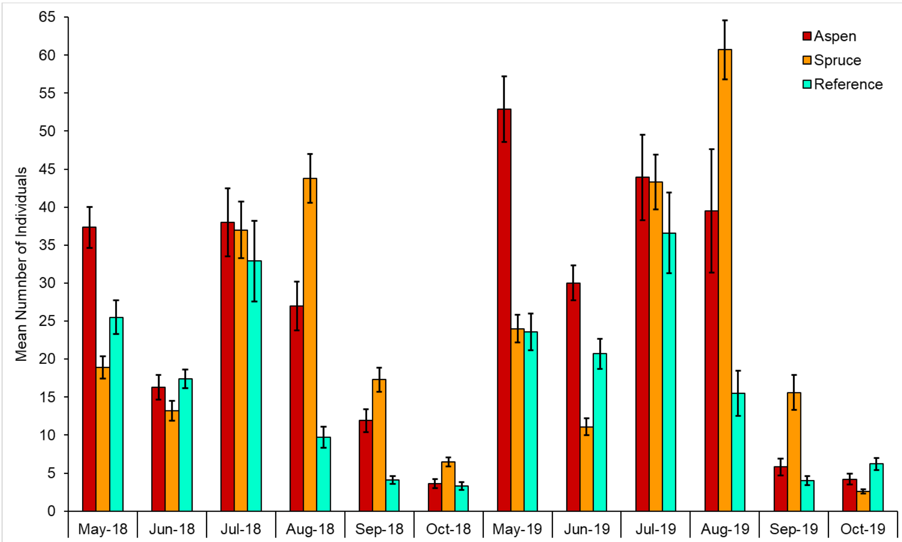


Figure 4.1. Mean number of beetle individuals per pitfall trap collected monthly. Error bars = standard error.

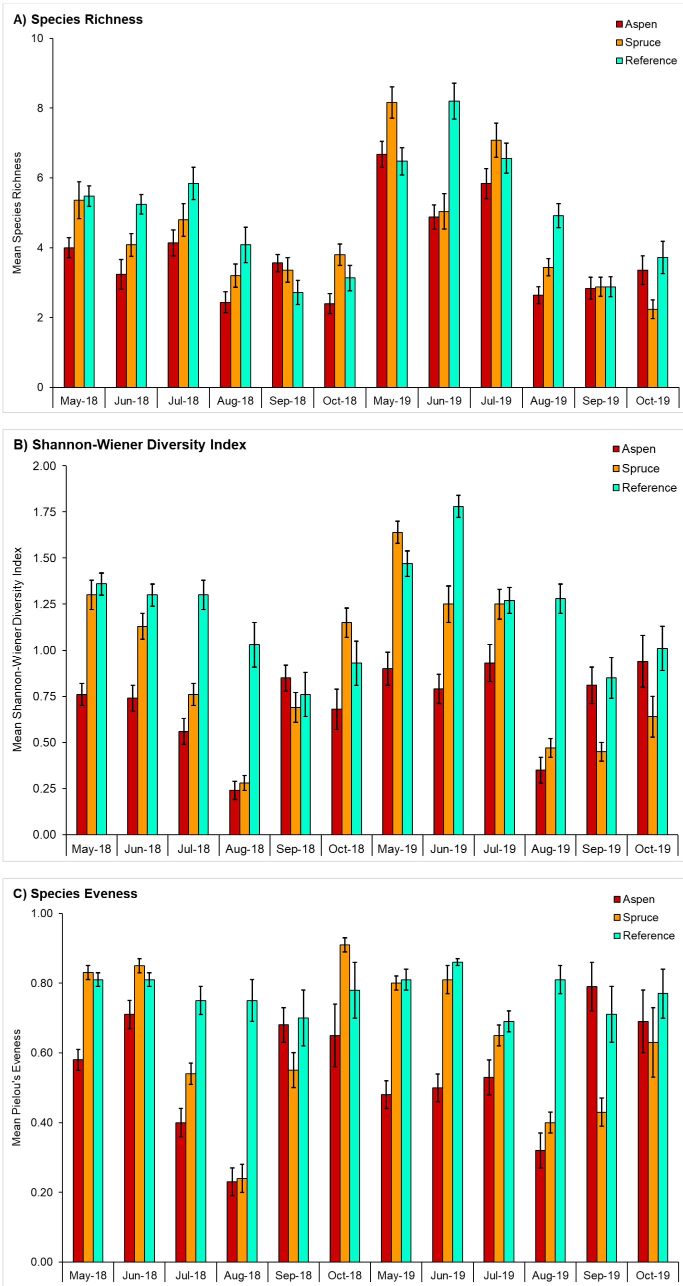


Figure 4.2. Variation in beetle diversity estimates. A) Species Richness, B) Shannon-Wiener Diversity Index, C) Pielou's Evenness. Error bars = standard error.

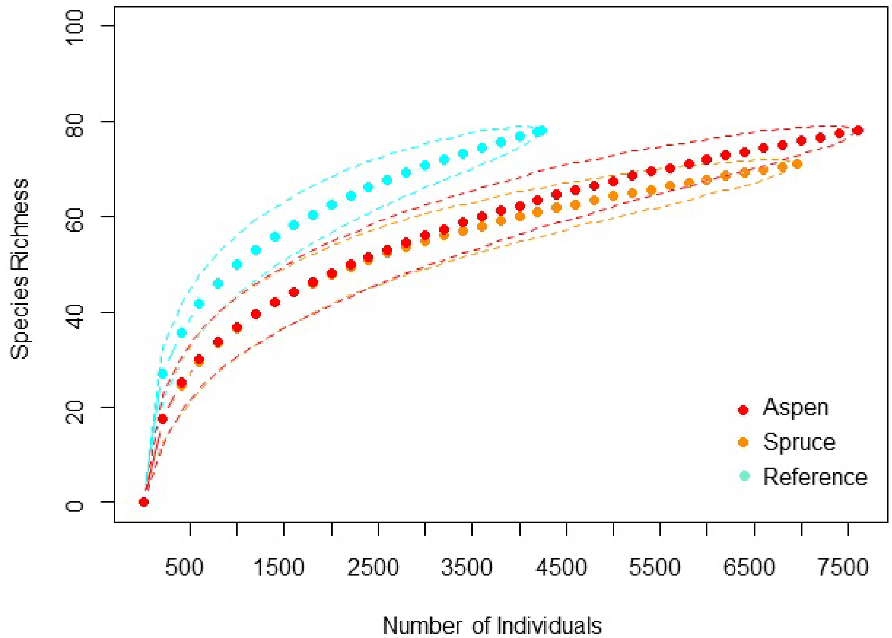
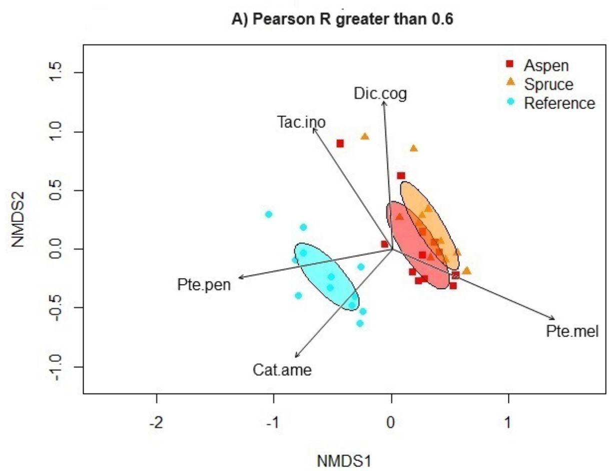


Figure 4.3. Total beetle species richness estimation using individual based rarefaction. Dashed lines represent 95 % confidence intervals.



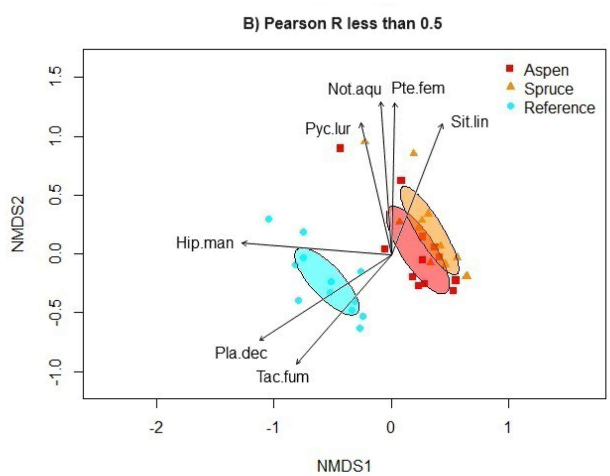


Figure 4.4. Non-metric multidimensional scaling (NMDS) ordination of beetle assemblages. Data were Hellinger transformed to improve final stress (14.9). Ellipses indicate 95 % confidence intervals around the group centroids for sites; vectors represent the strength of each significant (α = 0.001) beetle species collected with a correlation value (Pearson R) A) greater than 0.6; and B) less than 0.5. Species abbreviations are: *Pterostichus melanarius* (Pte.mel), *Catops americanus* (Cat.ame), *Pterostichus pensylvanicus* (Pte.pen), *Tachyporus inornatus* (Tac.ino), *Dicheirotrichus cognatus* (Dic.cog), *Tachinus fumipennis* (Tac.fum), *Platynus decentis* (Pla.dec), *Hippuriphila mancula* (Hip.man), *Pycnoglypta lurida* (Pyc.lur), *Notiophilus aquaticus* (Not.aqu), *Pterostichus femoralis* (Pte.fem), and *Sitona lineellus* (Sit.lin).

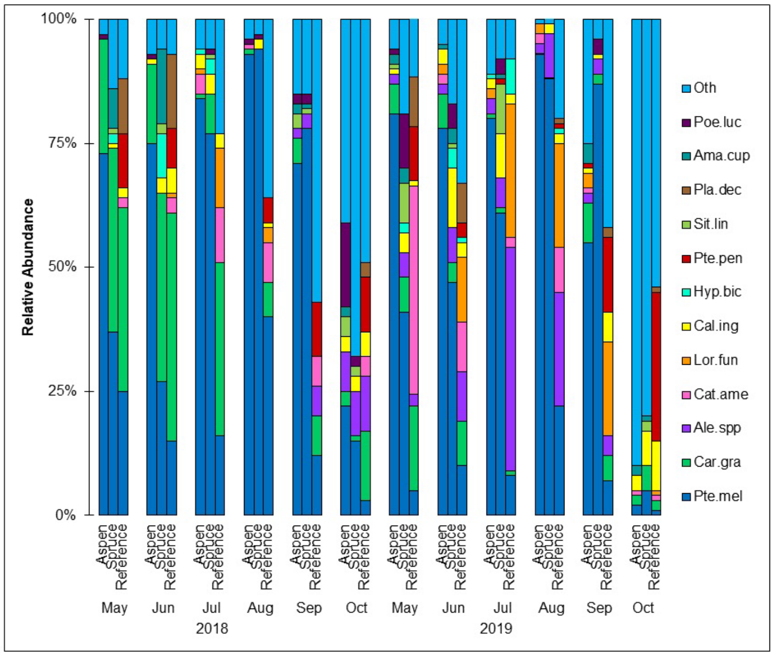


Figure 4.5. Relative abundance of 12 most abundant beetle species collected monthly. Most abundant species were selected based on total abundance not monthly abundance. Other included the remaining 145 beetle species. Species abbreviations are: *Pterostichus melanarius* (Pte.mel), *Carabus granulatus* (Car.gra), species in subfamily Aleocharinae (Ale.spp), *Catops americanus* (Cat.ame), *Lordithon fungicola* (Lor.fun), *Calathus ingratus* (Cal.ing), *Hypnoidus bicolor* (Hyp.bic), *Pterostichus pensylvanicus* (Pte.pen), *Sitona lineellus* (Sit.lin), *Platynus decentis* (Pla.dec), *Amara cupreolata* (Ama.cup), and *Poecilus lucublandus* (Poe.luc).

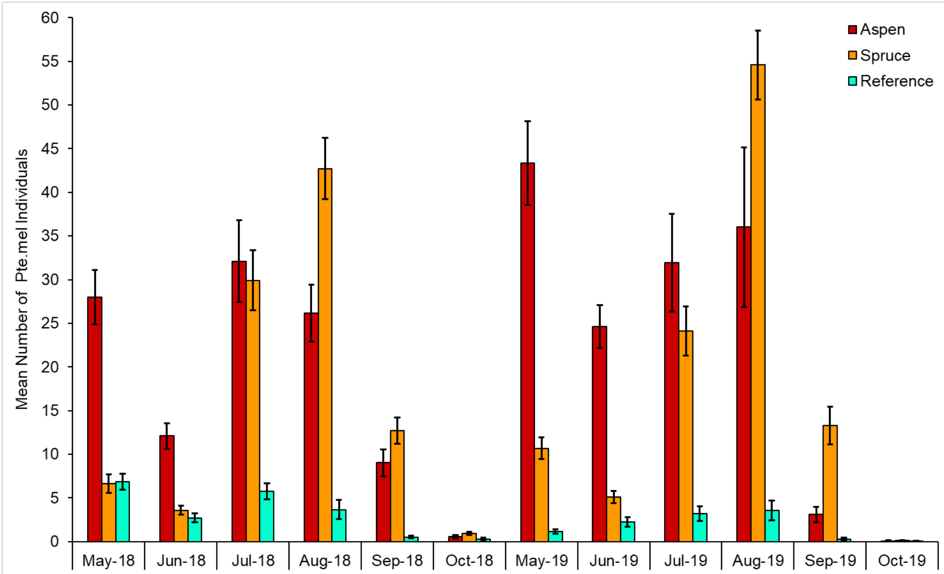


Figure 4.6. Mean number of *Pterostichus melanarius* individuals per pitfall trap collected monthly. Error bars = standard error.

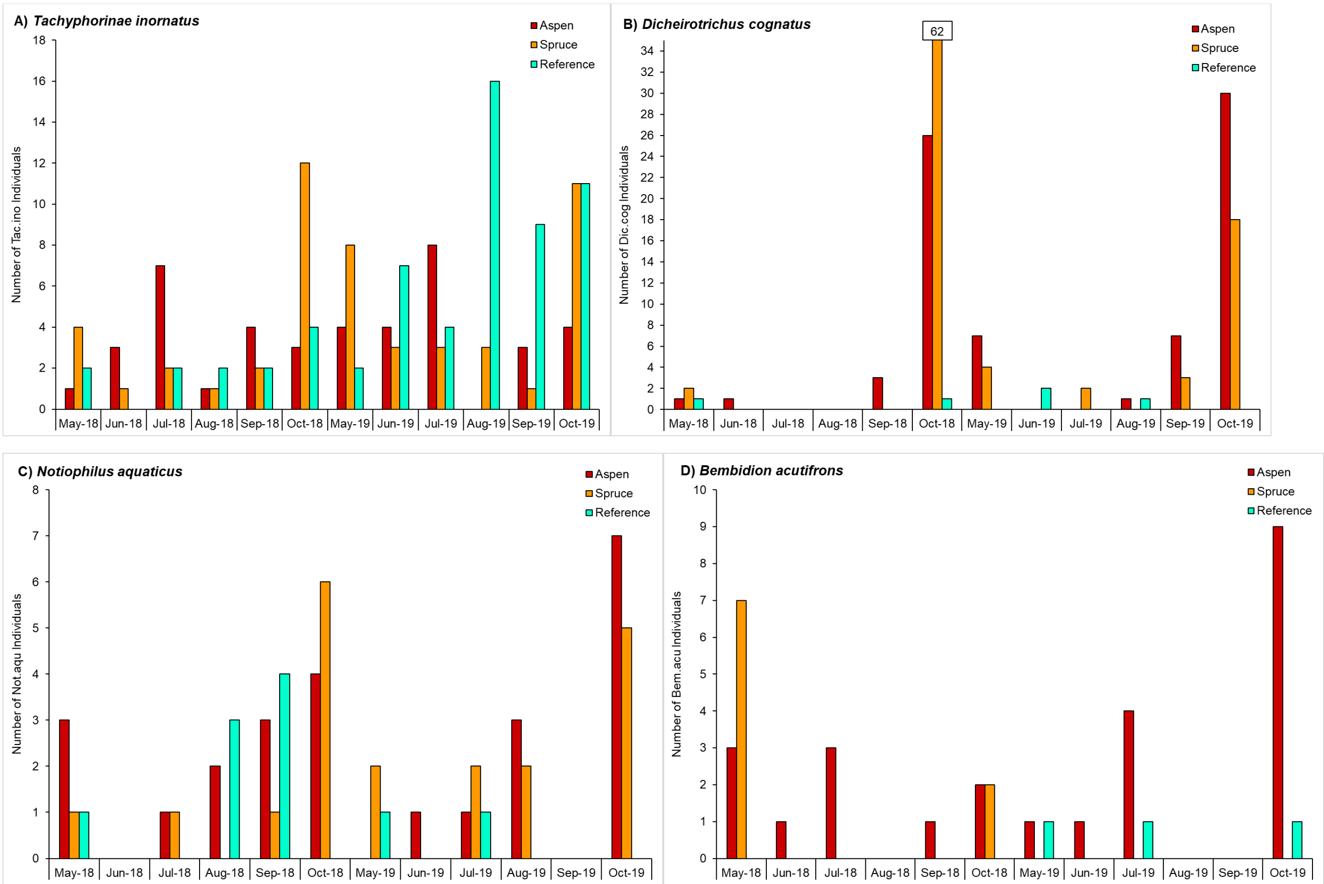


Figure 4.7. Beetle species of interest not included in 12 most abundant beetle species. Total number of individuals per pitfall trap collected monthly. A) *Tachyphorinae inornatus*, B) *Dicheirotrichus cognatus*, C) *Notiophilus aquaticus*, D) *Bembidion acutifrons*.

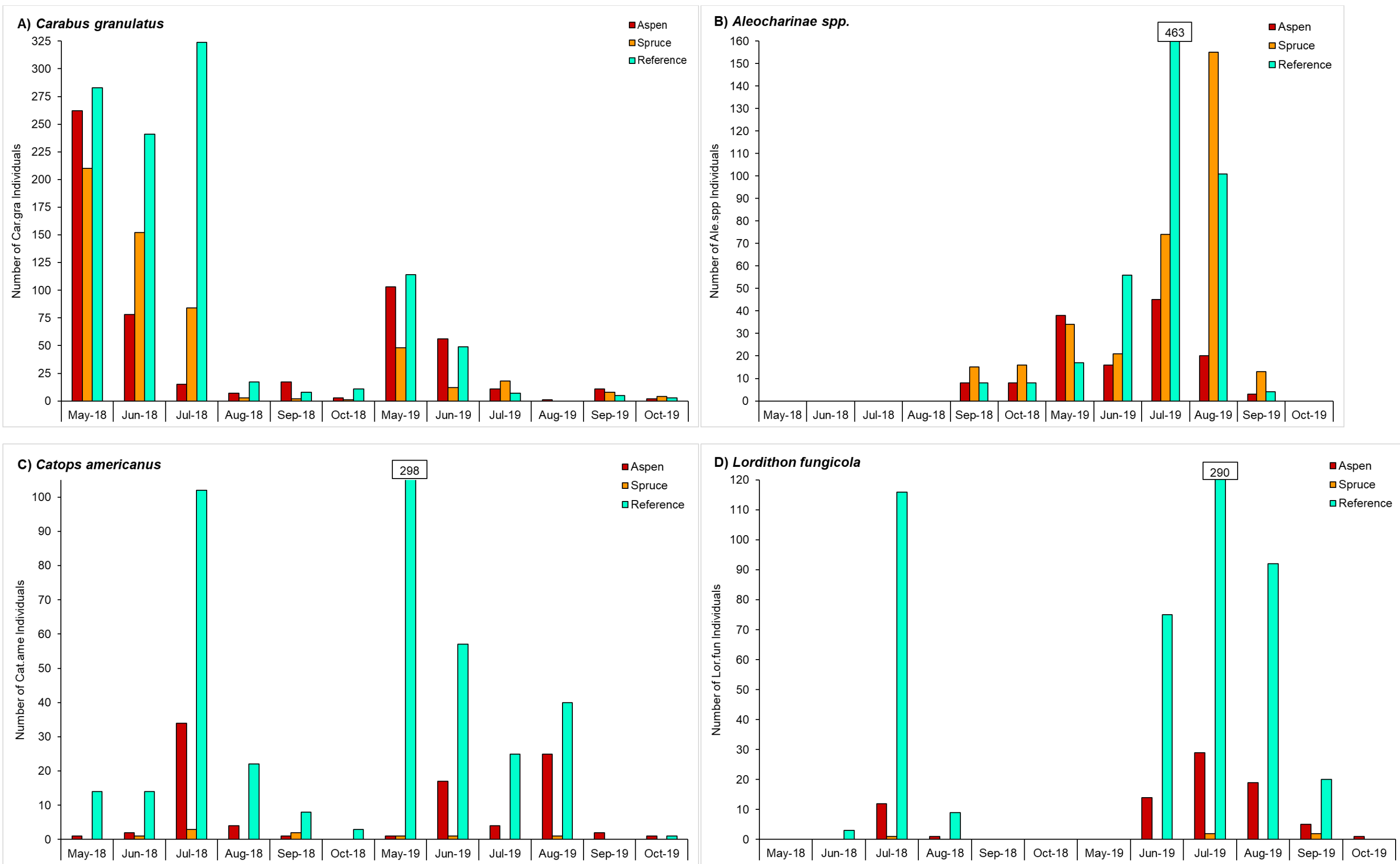


Figure 4.8. Total number of individuals collected for the 11 most abundant beetle species. A) *Carabus granulatus*, B) *Aleocharine spp.*, C) *Catops americanus*, D) *Lordithon fungicola*.

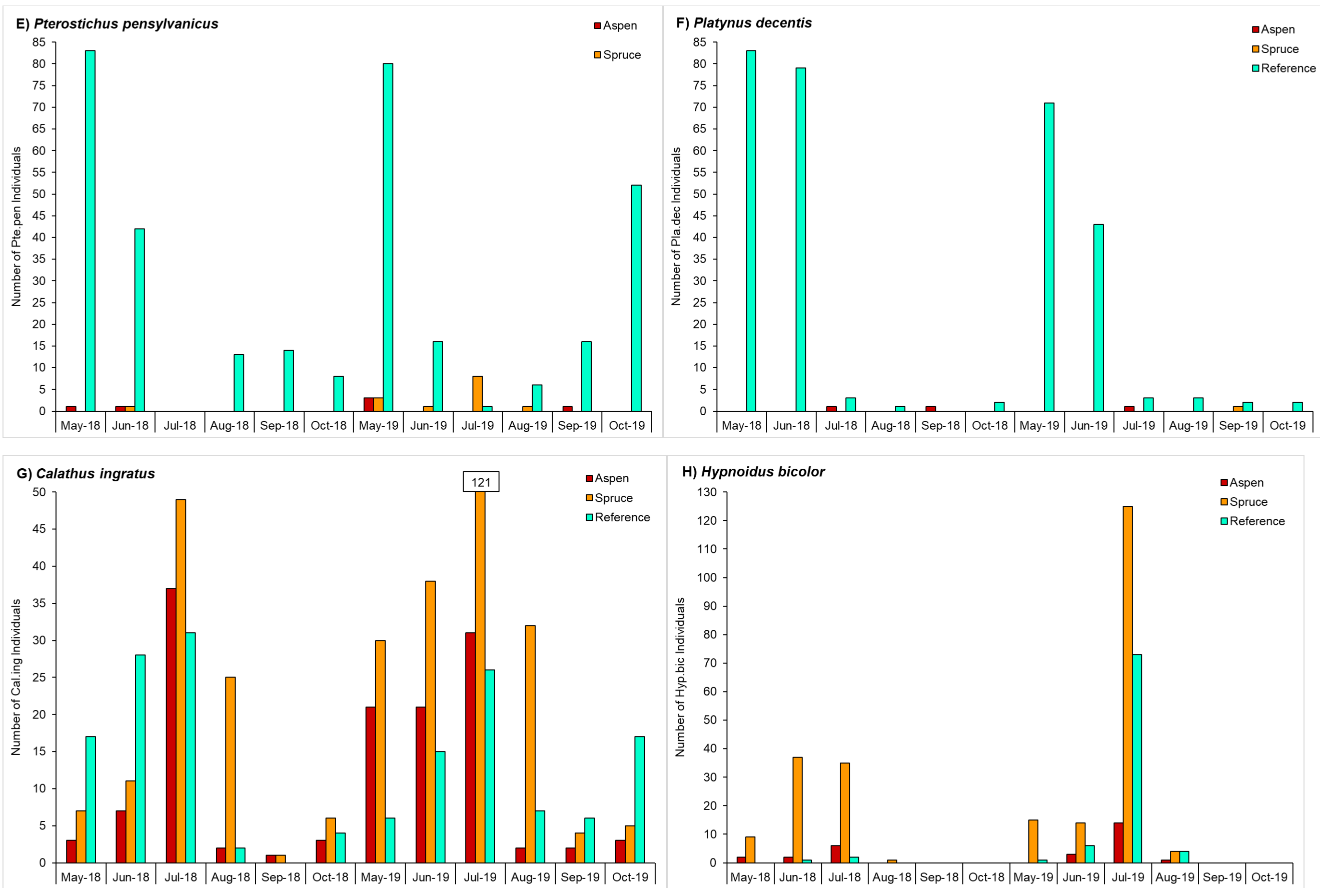


Figure 4.8 (continued). Total number of individuals collected for the 11 most abundant beetle species. E) *Calanthus ingratus*, F) *Hypnoidus bicolor*, G) *Pterostichus pensylvanicus*, H) *Platynus decentis*.

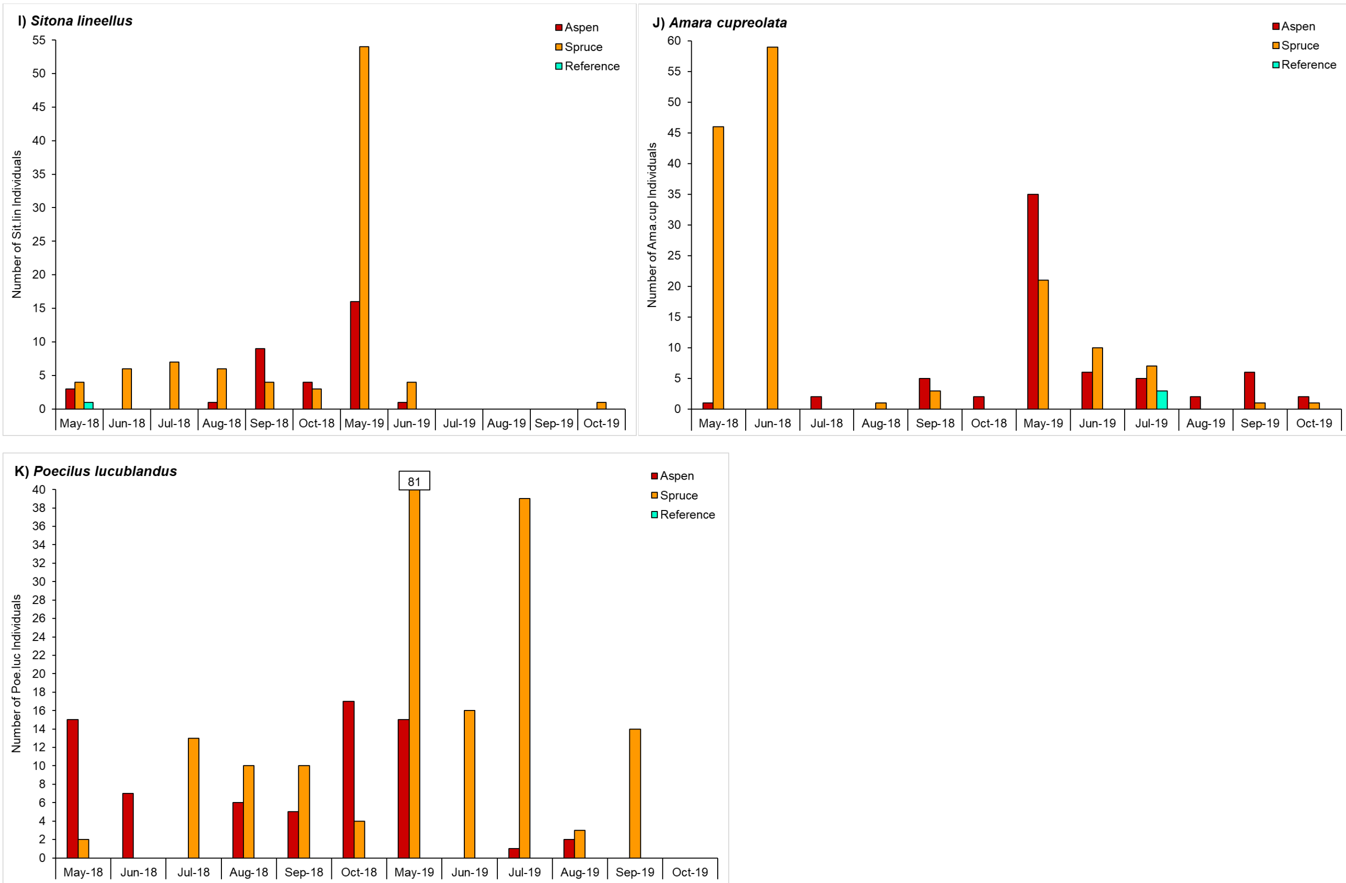


Figure 4.8 (continued). Total number of individuals collected for 11 most abundant beetle species. I) *Sitona lineellus*, J) *Amara cupreolata*, K) *Poecilus lucublandus.*

Supplemental Table 4.1. List of beetle species collected through pitfall trapping. A total of 153 species are organized by family from the most to least genus rich, and listed in alphabetical order, identification information is authority, date, and non-native status; * indicates singleton, ** indicates doubleton.

Family	Species Name	Taxonomic Authority	Non native
Carabidae	*Agonum affine*	Kirby, 1837	
	Agonum cupreum	Dejean, 1831	
	*Agonum cupripenne**	Say, 1823	
	Agonum gratiosum	Mannerheim, 1853	
	Agonum piceolum	LeConte, 1879	
	Agonum placidum	Say, 1823	
	*Agonum propinquum***	Gemminger & Harold, 1868	
	Agonum retractum	LeConte, 1846	
	Agonum sordens	Kirby, 1837	
	Agonum thoreyi	Dejean, 1828	
	*Amara apricaria**	Paykull, 1790	
	Amara cupreolata	Putzeys, 1866	
	Amara torrida	Panzer, 1796	
	Badister neopulchellus	Lindroth, 1954	
	Bembidion acutifrons	LeConte, 1879	
	*Bembidion graphicum**	Casey, 1918	
	Bembidion quadrimaculatum dubitans	LeConte, 1852	
	Bembidion rupicola	Kirby, 1837	
	Bembidion versicolor	LeConte, 1847	
	Bradycellus congener	LeConte, 1847	
	*Bradycellus lecontei**	Csiki, 1932	
	Calathus ingratus	Dejean, 1828	
	Calosoma frigidum	Kirby, 1837	
	Carabus granulatus	Linnaeus, 1758	Non native
	Chlaenius purpuricollis	Randall, 1838	
	Cymindis cribricollis	Dejean, 1831	
	Dicheirotrichus cognatus	Gyllenhall, 1827	
	*Harpalus amputates***	Say, 1830	
	*Harpalus ochropus**	Kirby, 1837	
	Harpalus somnulentus	Dejean, 1829	
	Loricera pilicornis	Fabricius, 1775	Non native
	Notiophilus aquaticus	Linnaeus, 1758	
	Platynus decentis	Say, 1823	
	Poecilus lucublandus	Say, 1823	
	Pterostichus femoralis	Kirby, 1837	
	Pterostichus melanarius	Illiger, 1798	Non native
	Pterostichus pensylvanicus	LeConte, 1873	
	Stenolophus fuliginosus	Dejean, 1829	
	Syntomus americanus	Dejean, 1831	
	Synuchus impunctatus	Say, 1823	
	Trechus apicalis	Motschulsky, 1845	
Staphylinidae	*Bryocharis analis*	Paykull, 1789	Non native
	*Bryoporus rufescens***	LeConte, 1863	
	*Eusphalerum fenyesi**	Bernhauer, 1912	
	Gabrius spp. 1	Stephens, 1829	
	Gabrius spp. 2*	Stephens, 1829	

	*Gyrohypnus fracticornis***	Muller, 1776	
	*Heterothops fusculus***	LeConte, 1863	
	*Ischnosoma fimbriatum***	Campbell, 1991	
	Lathrobium divisum	LeConte, 1880	
	Lordithon fungicola	Campbell, 1982	
	Lordithon thoracicus thoracicus	Fabricius, 1776	
	*Megaquedius explanatus**	LeConte, 1858	
	Megarthrus sinuaticollis	Lacordaire, 1835	
	Ontholestes cingulatus	Gravenhorst, 1802	
	Oxytelus fuscipennis	Mannerheim, 1843	
	Philonthus cyanipennis	Fabricius, 1792	
	*Philonthus varians**	Paykull, 1789	
	Proteinus basalis	Maklin, 1852	
	Pycnoglypta lurida	Gyllenhall, 1813	Non native
	Quedius fellmani	Zetterstedt, 1838	
	Quedius simulator	Smetana, 1971	
	Stenus spp.*	Latreille, 1797	
	Tachinus fumipennis	Say, 1832	
	Tachyphorus (Palporus) nitidulus	Fabricius, 1781	
	Tachyporus inornatus	Campbell, 1979	
Subfamily	*Aleochara bilineata*	Gyllenhal, 1810	
Aleocharinae	*Aleochara tahoensis*	Casey, 1906	
	Aleocharinae spp.		
	Atheta (Datomicra) dadopora	Thomson, 1867	
	Atheta (Dimetrota) districta	Casey, 1911	
	Atheta (Dimetrota) modesta	Melsheimer, 1844	
	Atheta (Dinaraea) angustula	Gyllenhal, 1810	
	Atheta (Dinaraea) spp.	Thomson, 1858	
	Atheta (Pseudota) klagesi	Bernhauer, 1909	
	Atheta graminicola	Casey, 1910	
	Devia prospera	Erichson, 1839	
	Dimetrota spp.		
	Drusilla canaliculata	Fabricius, 1787	
	Neothetalia spp.*	Klimaszewski, 2004	
	*Oxypoda canadensis**	Klimaszewski, 2006	
	*Oxypoda orbicollis***	Casey, 1911	
	Oxypoda spp.	Casey, 1911	
	Tetralaucopora spp.*	Bernhauer, 1928	
Curculionidae	*Anthonomus lecontei*		
	Ceutorhynchus punctiger	Gyllenhal, 1837	
	Dorytomus parvicollis	Casey, 1892	
	*Lepyrus nordenskioeldi canadensis***	Casey, 1895	
	*Nedyus flavicaudis**	Boheman, 1844	
	Notaris puncticollis	LeConte, 1876	
	Otiorhynchus ovatus	Linnaeus, 1758	
	*Pelenomus ventralis**	Sleeper, 1957	
	Sitona lineellus	Bonsdorff, 1785	
	*Trypodendron retusum**	LeConte, 1868	
	Tychius melliloti	Stephens, 1831	
Cryptophagidae	*Atomaria ephippiata*	Zimmermann, 1869	
	Atomaria longipennis	Casey, 1900	
	Atomaria pusilla	Paykull, 1798	
	Caenoscelis parallela	Casey, 1900	

	Cryptophagus cellaris	Scopoli, 1763	
	*Cryptophagus croceus**	Zimmermann, 1869	
Coccinellidae	*Hyperaspis consimilis*	LeConte, 1852	
	*Hyperaspis oregona***	Dobzhanksy, 1941	
	Hyperaspis quadrivittata	LeConte, 1852	
	*Hyperaspis undulata***	Say, 1824	
	Scymnus apicanus	Chapin, 1973	
Chrysomelidae	*Graphops marcassita***	Croch, 1873	
	Hippuriphila mancula	LeConte, 1861	
	*Phyllotreta cruciferae***	Goeze, 1777	
	Phyllotreta striolata	Fabricius, 1803	
Latridiidae	*Corticaria & Melanophthalma* spp.		
	Corticaria valida	Fall, 1899	
	Latridius minutus	Linnaeus, 1767	
	Melanophthalma americana	Mannerheim, 1844	
Nitidulidae	*Eupeura* spp.		
	Glischrochilus siepmanni	Brown, 1932	
	Omosita colon	Linnaeus, 1758	
Scarabaeidae	*Aphodius (Cryptoscatomaster) browni**	Hinton, 1934	
	*Aphodius (Oscarinus) rusicola**	Melsheimer, 1845	
	*Phyllophaga anxia**	LeConte, 1850	
Hydrophilidae	*Cercyon assecla*	Smetana, 1978	
	Helophorus sempervarians	Angus, 1970	
	*Hydrobius fuscipies***	Linnaeus, 1758	
	*Sphaeridium scarabaeoides**	Linnaeus, 1758	
Dytiscidae	*Hydaticus aruspex**	Clark, 1864	
	*Rhantus frontalis**	Marsham, 1802	
Silphidae	*Nicrophorus defodiens*	Mannerheim, 1846	
	Nicrophorus investigator	Zetterstedt, 1824	
Leiodidae	*Catops americanus*	Hatch, 1928	
	Leiodes punctostriata	Kirby, 1837	
Anthicidae	*Anthicus cervinus*	LaFerte, 1847	
	Anthicus coracinus	LeConte, 1852	
Byrrhidae	*Byrrhus americanus*	LeConte, 1850	
	*Cytilus alternatus**	Say, 1825	
Pyrochroidae	*Hister furtivus**	LeConte, 1860	
	Pedilus abnormis	Horn, 1874	
Brentidae	*Apion centrale*		
Buprestidae	*Dicerca tenebrica**	Kirby, 1837	
Cantharidae	*Podabrus laevicollis***	Kirby, 1837	
Elateridae	*Hypnoidus bicolor*	Eschscholtz, 1829	
Eucnemidae	*Epiphanis cornutus*	Eschscholtz, 1829	
Lucanidae	*Platycerus depressus***	LeConte, 1850	
Mordellidae	*Mordellina nigricans*	Melsheimer, 1846	
Orsodacnidae	*Orsodacne atra*	Ahrens, 1810	
Phalacridae	*Olibrus semistriatus*	LeConte, 1856	
Scirtidae	*Cyphon variabilis***	LeConte, 1853	

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CHAPTER V: SPIDER (ARANEAE) RESPONSE TO FOREST RECLAMATION METHODS AFTER COAL MINING

1. INTRODUCTION

Spiders have emerged as valuable indicators of forest harvesting and management practices across Canada, offering insights into disturbance levels, edge effects, forest cover types, disturbance timelines, microhabitats, and ecological succession (Larrivée et al., 2008; Pearce et al., 2005, 2004; Pinzon et al., 2016, 2012; Work et al., 2004). Forest spider assemblages from Alberta and Quebec have clear regional differences, yet disturbance response is similar, as indicated with partial cutting leading to functional homogenization and prevalence of generalist hunting spiders (Buddle and Shorthouse, 2008). This similar response, despite significant regional influences on spider populations, demonstrates that spiders have potential as indicators for national scale forest management and monitoring in Canada (Buddle and Shorthouse, 2008). Extensive research on functional diversity and development of comprehensive DNA reference libraries bolster spiders as strong indicators. University of Guelph's Centre for Biodiversity Genomics regularly adds novel spider species records, expanding a DNA reference library covering 92 % of Canada's 1477 known spider species, significantly enhancing specimen identification. This includes taxa such as Linyphiidae, which lacked adequate morphology based diagnostic images or reliably identified voucher specimens for some species(Bennett et al., 2019). The proposed global classification of spider guilds by Cardoso et al (2011) based on foraging strategy suggests that spider families may serve as reliable ecological surrogates, enabling consistent comparisons across regions and predictable guild structures independent of taxonomic composition (Cardoso et al., 2011).

In land reclamation, spiders offer valuable insights into ecosystem recovery and function, suggesting their potential as indicators even at family level or for assessing functional guilds (Mannu et al., 2020; Sylvain et al., 2019). Mannu et al. (2020) studied the potential role of spiders as bioindicators of natural succession after mining across a chronosequence in the Mediterranean, and attributed differences in spider assemblages to variation in habitat complexity and structure from past mining activities, and suggested spider use in post-mining monitoring. Forestry management research in Hungary revealed that while spiders serve as reliable indicators, their sensitivity and discriminatory power vary with disturbance scale, emphasizing the need for context specific assessments (Samu et al., 2021). In western North Dakota, despite variation in plant community and habitat structure between reclamation sites and native prairies, total spider abundance did not show significant differences (Sylvain et al., 2019), suggesting spiders may not be reliable indicators of reclamation success at higher taxonomic levels.

Hammond et al. (2022) compared spider assemblages in reclaimed borrow pits in arrested succession and undisturbed forest fragments, and examined effects on spider assemblages of reclamation methods, including mounding and herbicides, at an in-situ oil facility near Cold Lake, Alberta. They found soil structure, plant cover, and soil organic matter significantly affected arthropod assemblage establishment in reclamation sites. Seismic line reclamation in treed boreal peatlands using inverted mounding lowered ground dwelling spider abundance, richness, and diversity in three years (Echiverri et al., 2023). Ground dwelling spiders, rove beetles, and ant assemblages differed in mounded and untreated seismic lines, but not in untreated seismic lines and reference assemblages (Echiverri et al., 2023). Knowledge is limited on spider response to soil building materials and reclamation methods after disturbances such as surface mining, and we do not clearly understand spider dispersal and succession in reclamation development.

Building on previous spider research in land reclamation in Alberta, we developed two research objectives: i) to evaluate effects of land reclamation methods (direct soil placement and natural revegetation versus stockpiled soil and planting) on spider assemblages and diversity; and ii) to determine spider indicator species and feasibility of being incorporated into current reclamation monitoring criteria in Alberta. We expected direct soil placement with natural revegetation and proximity to an undisturbed forestwould lead to spider assemblages similar to a reference site.

2. MATERIALS AND METHODS

2.1. Research Sites

The study area is located at the Genesee Coal Mine, 80 km west of Edmonton, Alberta in the Dry Mixedwood Natural Subregion of the Boreal Forest Natural Region (Natural Regions Committee, 2006). Reclaimed areas include agricultural land, wetlands, forests, and future recreational areas (Capital Power, 2021). Soils were mapped as Orthic and Dark Gray Luvisols with calcareous parent material, and heavy clay in the Bt, BC, and Ck horizons (Lindsay et al., 1968). Mean annual temperature is 2.0 °C and mean annual precipitation 536 mm (Alberta Parks, 2015).

Coal mining reclamation includes smoothing and leveling of landforms to slopes that are considered appropriate for end land uses, and then drainage pathways are re-established

(TransAlta, 2022). Subsoil is placed over the mined area, allowed to settle for approximately a year, then any large rocks are removed and the subsoil is de-compacted with large scale rippers or subsoilers. Topsoil placement depth and potential amendments are determined based on end land use and available materials at the sites or nearby areas; fertilizing with agricultural fertilizer mixes, mulching, and deep ripping are common practices. Revegetation is prioritized to control soil erosion and to reduce infestation and spreading of non-native weedy species. Forest reclamation areas are typically planted with shrub and tree species, including willows (*Salix* spp.), trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), white spruce (*Picea glauca*), and some native shrubs.

Research sites were selected based on reclamation prescriptions, end land use, and distance to nearby undisturbed forest. Reclamation sites were named after their dominant tree species of aspen and spruce. Reclamation in the aspen reclamation site (hereafter Aspen) included direct placement of approximately 20 cm of salvaged forest floor litter (LFH) and topsoil (A horizons), coarse woody material and straw were used as additional soil amendments. The spruce reclamation site (hereafter Spruce) had approximately 15 cm of stockpiled salvaged topsoil and straw added as an amendment for soil.

2.2. Spider Collection

Each research site had ten plots (5 x 5 m) with similar drainage and slope, and at least a 5 m buffer between plots and 10 m buffer from site boundaries or forest edges. Plots were divided into a grid system of 25 subplots. Five randomly selected subplots per plot were used for soil and vegetation assessments including percent cover by species and soil core collection for soil chemical and physical analyses (July 2017). Vegetation assessments were completed in both 2018 and 2019 (Table 5.1).

Holes from which the soil cores were taken were further used to install pitfall traps. Traps were made of clear 16 oz plastic containers (height = 7.62 cm, top diameter = 11.75 cm), that were filled with 200 mL of propylene glycol, and a styrofoam plate roof (diameter = 17.1 cm) was anchored with bamboo skewers, and placed < 3 cm above the trap. Pitfall traps were opened the second Thursday of each study month and left in the field for 1 week. Monthly sampling started in May and ended in October 2018 and 2019. Spiders were identified from May to August in 2018. Monthly samples were collected from three randomly selected pitfall traps, for a total of 30 samples per site, per month. Sampling in 2018 was May 10, June 7, July 12, and August 9.

Individuals in pitfall traps were sorted into coarse taxonomic categories and stored in containers with 75 % isopropyl alcohol for preservation and taxonomic identification. All identifications were completed by Kirra Kent and voucher specimens were deposited at the Northern Forestry Centre in Edmonton, Alberta.

2.3. Statistical Analyses

Spider communities were compared among reclamation and forest reference sites using alpha diversity indices, rarefied estimates of total species richness, and unconstrained nonparametric ordination. We consolidated male and female spiders collected from May to August 2018 at each pitfall trap location, a total of 30 sampling points per site. This dataset was used for all analyses. We were not interested in the effects of sampling month for this analysis as we did not have a dataset for the complete growing season and only one year of data.

2.3.1. Spider alpha diversity

Spider alpha diversity indices were calculated for each site including species richness, Pielou's evenness (J'), Shannon diversity index (H'), and Simpson diversity index (D) with these formulas.

$\mathrm{J}' = \frac{\mathrm{H}'}{\mathrm{H}'\max}$ and $\mathrm{H}'\max = \ln(\mathrm{S})$, where S is species richness.

$\mathrm{H}' = -\sum \mathrm{p} \ln(\mathrm{p})$, where p is the proportion of individuals of each spider species in a community.

$\mathrm{D} = 1 - \frac{\sum \mathrm{n}(\mathrm{n}-1)}{\mathrm{N}(\mathrm{N}-1)}$, where n is the number of individuals of each species and N is the total number of individuals of all species.

Pielou's evenness (J') and Simpson's diversity index (D) are constrained between 0 and 1, where 0 means that there is no evenness and low diversity and 1 means that all species occur in equal abundance or there is high diversity.

2.3.2. Rarefaction and total species richness estimates

Analyses were performed with the R statistical package (version 4.1.3) (R Core Team, 2023). Rescaling sample based rarefaction to the number of individuals allows for understanding the sensitivity of estimates of species richness to the number of individuals collected while being derived using sample based rarefaction curves; it measures overall community richness (Buddle et al., 2005). We used the rarefy function in vegan, multiples of 200, and total number of individuals for each site. We used the specpool function in vegan to calculate incidence based estimators of the total species richness, Chao2 (S_2) and Jacknife (S_{jack}), for each site with the following formulas.

$S_2 = S_{obs} + \frac{(Q1)^2}{2(Q2)}$, where S_{obs} is the number of species in the sample, Q1 is the species occurring in only one sample (singletons), and Q2 is the number of species occurring in two samples (doubletons) (Chao, 1987; Colwell and Coddington, 1994).

$S_{jack} = S_{obs} + [\frac{Q1(2m-3)}{m} - \frac{Q2(m-2)^2}{m(m-1)}]$, where m represents the total number of samples (Smith and van Belle, 1984).

2.3.3. Spider assemblages

Spider assemblages were visualized using non metric multidimensional scaling (NMDS) unconstrained ordination with the metaMDS function from the vegan package, and Bray-Curtis as the distance measure (Oksanen et al., 2015). Spider species data were Hellinger transformed, with decostand function in vegan, to account for low counts and many zeros (Legendre and Gallagher, 2001). Number of dimensions were determined using dimcheckMDS in goeveg and by scree plots and reduction in stress with decreasing dimensionality (McCune and Grace, 2002). Significant correlated species and environmental parameters were visualized on the ordination plot, using vec and envfit functions in the vegan package (alpha = 0.001, permutations = 999).

Permutational Multivariate ANOVAs (perMANOVA) were performed with the adonis function in the vegan package, to determine whether there were significant differences between spider assemblages recorded in the undisturbed forest reference and in the reclamation sites (alpha = 0.001, permutations = 999). P values were corrected by Bonferroni for repeated tests using the RVAideMemoire package. We evaluated differences among sites in spider assemblages using homogeneity of dispersions tests (betadisper), followed by ANOVAs to compare mean distance-to-centroid of spider assemblages among study sites. Significant ANOVAs (alpha < 0.001) were followed by Tukey's HSD post-hoc test to identify pairwise differences in spider assemblages between the three study sites.

2.3.4. Indicator species analysis

Indicator species analysis (ISA) determines significant indicators based on aspecificity (A), which is the probability that the surveyed site is part of the target site group; and given a species presence and sensitivity (B), probability of finding the species in sites belonging to the target site group. The indicator value index assesses the predictive value of a species as an indicator of a combination of site groups (De Cáceres et al., 2010). Indicator species analysis was performed using the multipatt function from the indicspecies package (Dufrêne and Legendre, 1997; De Cáceres and Legendre, 2009) to determine which spider species were indicators of the Reference or reclamation sites. We used a quantitative or binary response with a randomization test for our analysis (alpha = 0.001, permutations = 999).

3. RESULTS

We identified 8,159 spiders, from 14 families, and 102 species; immature spiders identified to family compromised 32.0 % of the total number of spiders, and were not included in diversity and subsequent analysis (Supplemental Table 5.1). We collected 2,608 immature spiders, 17.3 % from Reference, 34.0 % from Aspen, and 48.6 % from Spruce. We collected 5,553 adult individuals. The nine most abundant species represented five families and accounted for 74.9 % of the total. One species, *Pardosa moesta* (Banks 1892), accounted for almost a third of the total catch at 30.5 %. The next most abundant species, *Trochosa terricola* (Thorell 1856), accounted for 13.3 % of the total, followed by *Pardosa modica* (Blackwall 1846) accounting for 6.5 %. The remaining species included *Agroeca ornata* (Banks 1892) at 5.4 %, *Xysticus emertoni* (Keyserling 1880) at 4.5 %, *Diplocephalus subrostratus* (Pickard-Cambridge 1873) at 4.0 %, *Pardosa xerampelina* (Keyserling 1877) at 3.6 %, *Alopecosa aculeata* (Clerck 1758) at 3.6 %, and *Neoantistea magna* (Keyserling 1887) at 1.2 %.

Singletons are species that were represented by only a single individual in the collections; doubletons were represented by only two individuals. We collected 29 singletons and 9 doubletons (Supplemental Table 5.1). Reference had 8 singletons and 2 doubletons, Aspen had 3 singletons and 1 doubleton, and Spruce had 17 singletons and 2 doubletons. Reference and Spruce shared 1 doubleton, and reclamation sites shared 3 doubletons.

3.1. Spider Alpha Diversity

Mean alpha diversity indices were similar among sites (Figure 5.1). Species richness was highest in Reference; reclamation sites were similar to Reference (Figure 5.1A). Aspen had more variation in species evenness and Simpson and Shannon indices than the other sites. Spruce species evenness and Shannon index was more similar to Reference than Aspen (Figure 5.1B, D).

3.2. Rarefaction And Total Species Richness Estimates

Based on rarefaction curves and total species richness estimators, we underestimated total species richness in all sites (Figure 5.2, Table 5.2). Approximately 20 additional species are expected in both Reference and Aspen, while 53 additional spider species are expected for Spruce (Table 5.2). If the estimated total species richness for each site were accurate there would likely be significant differences in alpha diversity indices among sites. Therefore, the similarity we observed between alpha diversity indices in reclamation sites and Reference does not indicate recovery of spider species diversity and richness. If spiders from September and October 2018 and all individuals from 2019 were identified, rarefaction curves may have plateaued.

3.3. Relative Abundance Of Major Spider Species

A comparison of relative abundance of the 9 most abundant species and other spider species shows that 3 species were primarily captured in Reference and 2 in reclamation sites; the remaining 4 species were captured across all sites (Figure 5.3). *Pardosa moesta* was collected at all sites, but showed preference for reclamation sites with relative abundance 51.5 % and 32.9 % in Aspen and Spruce, respectively. *Trochosa terricola* was collected at all sites with relative abundance in reclamation sites half that of Reference. *Pardosa modica* and *Xysticus emertoni* were primarily collected in reclamation sites, relative abundance was higher in Spruce . *Agroeca ornata* was not captured in Spruce and 9 individuals (of 302) were in Aspen. *Diplocephalus subrostratus* was primarily captured in Reference; four individuals were collected in Aspen, and two in Spruce (of 221). There was one *Pardosa xerampelina* in Spruce and four in Aspen (of 200).

3.4. Other Spider Species Of Interest

While not included in the most abundant spider species, there were other species of interest. Two sheetweb spider species (Linyphiidae) had individuals primarily collected in Reference; a few individuals were in Spruce, with Aspen having significantly more than Spruce. *Allomengea dentisetis* (Grübe 1861) had one individual in Spruce, 24 in Aspen, and 48 in Reference. Similarly, *Diplostyla concolor* (Wider 1834) had 9 individuals in Spruce, 18 in Aspen, and 74 in Reference.

Three spider species collected across sites showed a clear preference for reclamation sites. *Agyneta simplex* (Emerton 1926), a sheetweb spider species, had the highest number of individuals in Spruce (48 of 81), followed by 20 individuals in Aspen and 13 in Reference. Reclamation site preference was seen with two ground hunter spider species. *Pardosa distincta* (Blackwall 1846) had one individual collected in Reference (of 122), 20 in Aspen, and 101 in Spruce. *Gnaphosa parvula* (Banks 1896) collected in reclamation sites was more balanced, 33 in Aspen and 43 in Spruce, while only 7 individuals were collected in Reference.

3.5. Spider Assemblages

The NMDS two dimensional solution had low final stress of 0.136; however, the best solution was not repeated after 100 tries (Bray-Curtis; non-metric fit, $R^2 = 0.982$; linear fit, $R^2 = 0.941$). Our ordination plot illustrated separation of spider assemblages and displayed ellipses with 95 % confidence intervals for grouping sites, the Reference ellipse did not overlap with either reclamation site (Figure 5.4). Reclamation site ellipses were close together with some small overlaps, indicating spider assemblages in reclamation sites were more similar to each other than to Reference. The Aspen ellipse was closer to Reference than Spruce, but likely not significant.

Significant spider species vectors ($p = 0.001$) were organized by correlation strength (Pearson R) and divided into four categories: 0.51 to 0.9 (high), 0.31 to 0.5 (moderate), 0.2 to 0.3 (low), and 0.1 to 0.19 (very low). Spiders with high correlation had three species associated with Reference and three with reclamation sites; *Agroeca ornata* and *Pardosa modica* vectors were in opposite directions (Figure 5.4A). Species with moderate correlation included six generally associated with Reference and two, *Pardosa distincta* and *Alopecosa aculeata*, associated with reclamation sites (Figure 5.4B). Spider species with low correlation value had three species associated with Reference, three with reclamation sites, and three that did not show strong site preference; *Maso sundevalli* (Westring 1851). *Xysticus ferox* (Hentz 1847) vectors are in opposite directions (Figure 5.4C). Spider species with very low correlation value had four species associated with Reference, one showed no strong associations, and one was associated with reclamation sites (Figure 5.4D).

Significant environmental vectors ($p = 0.001$) were categorized into vegetation and soil properties (Figure 5.5). Shrub cover was associated with Reference; non-native forb and grass and weed cover with reclamation sites (Figure 5.5A). Vectors for shrub cover and non-native grasses were in opposite directions. Native grasses did not have a strong site association. Soil pH and total inorganic carbon were associated with reclamation sites (Figure 5.5B). Soluble sodium and chloride, and total nitrogen and organic carbon were associated with Reference (Figure 5.5B).

There were significant site effects (PERMANOVA, df = 2, $F = 39.9$, $R^2 = 0.48$, $p = 0.001$). The spider assemblages in the Reference were significantly different from that of the reclamation sites ($p = 0.003$), which were significantly different from each other ($p = 0.003$). Dispersion of spider assemblages differed by site (betadisper, ANOVA, df = 2, $F = 5.74$, $p = 0.007$); spider assemblages were more variable in Reference than Spruce (Tukey's HSD, 95% CI = 0.004 to 0.08, $p = 0.024$), with no differences between Reference and Aspen ($p = 0.898$) and Spruce and Aspen ($p = 0.072$).

3.6. Indicator Species Analysis

Indicator species analysis (ISA) considered 102 spider species and 40 species were selected, with 8 species being indicators for both reclamation sites (Table 5.3). The strongest three species indicators for the reclamation sites included *Pardosa modica*, *Xysticus emertoni*, and *Gnaphosa parvula*. Reference had 21 significant indicator species ($p < 0.05$), while both Aspen and Spruce had three each. Reference indicator species had relatively high aspecificity, eight species had scores of 1, sensitivity varied more considerably, two species had scores of 1. Five spider species with high indicator scores and correlation, and strong significance ($p = 0.001$), were selected as Reference indicator species: *Bathyphantes pallidus* (Banks 1892), *Diplocephalus subrostratus*, *Pardosa xerampelina*, *Agroeca ornata*, and *Cybaeopsis euopla* (Bishop and Crosby 1935). Some of these species were significant vectors in our ordination plots. The strongest species indicator for Aspen was *Agyneta fabra*, and for Spruce it was *Pardosa distincta*. Interestingly there were also four significant indicator species for Reference+Aspen, all with high indicator scores and correlation. These indicators include *Allomengea dentisetis*, *Kaestneria pullata* (Pickard-Cambridge 1863), *Bathyphantes concolor*, and *Neriene clathrate*.

4. DISCUSSION

Our study provides valuable insights into spider and individual species responses to disturbances and reclamation methods. Our reclamation sites were predominantly inhabited by ground hunting spiders of the genus *Pardosa*, which are generalists that thrive in grass dominated areas. This abundance of ground hunters, relative to the lower presence of sheet and space web weaving spiders, indicates a homogenization trend in spider assemblages within young reclamation sites. Despite our expectation that spider diversity would be higher in Aspen than in Spruce due to the closer proximity of Aspen to Reference and its higher shrub and native species cover, spider assemblages did not align with this prediction. Spruce sites exhibited greater heterogeneity, suggesting complex factors influence spider assemblages in the reclaimed ecosystems. Shifts in spider guild dominance in reclamation sites underscores the potential of spiders as economically feasible and ecologically sensitive indicators for inclusion in reclamation monitoring efforts.

4.1. Spider Response To Reclamation

Effects of reclamation methods on epigaeic spiders varied depending on the measured parameter, although the total number of adult spiders collected in each site were relatively similar.

Reclamation sites having slightly higher spider abundance is likely related to vegetation composition and food availability. Sylvain et al. (2019) found total spider abundance was positively correlated with Kentucky bluegrass (*Poa pratensis*) prevalence, which was the dominant non-native grass species in both reclamation sites. Similar species richness and diversity measures among sites, and different spider assemblages in our study were seen in other studies across multiple types of disturbances (Alcalde et al., 2021; Mannu et al., 2020; Pearce et al., 2004; Samu et al., 2021; Work et al., 2004). Variation in spider assemblages across diverse forest types reflect habitat preferences and microhabitat changes from harvesting is associated with differences in ground vegetation structure, shade levels, litter depth, and soil water content (Pearce et al., 2004).

Reclamation site dominance by *Pardosa moesta*, a ground hunting wolf spider (Lycosidae), is unsurprising. Lycosidae is frequently the dominant ground spider family. Research on the life history of *Pardosa moesta* in central Alberta, found it was one of the most dominant wolf spiders in forests, had reproductive peaks in May and June, and underwent a two year development cycle with two overwintering periods (Buddle, 2000). The response of spider assemblages to alternative harvesting methods in two large scale forestry experiments, EMEND (Ecosystem Management Emulating Natural Disturbance) in Alberta and SAFE (Sanctuaire pour animaux de Ferme de l'Estrie) in Quebec, found *Pardosa moesta* was the most frequently collected spider species (Buddle and Shorthouse, 2008). It thrives in various habitats including open areas, deciduous forests, and clear cut sites, with notable prevalence even 7 years post-harvest, particularly in deciduous stands (Buddle, 2000; Buddle and Shorthouse, 2008; Larrivée et al., 2008; Pinzon et al., 2012). Pearce et al. (2004) found that *Pardosa moesta* had higher abundance in grass and humus microhabitats than moss or leaf litter microhabitats, and was more likely to be captured with high soil disturbance from pitfall trap installation, indicating its versatility in exploiting various ground cover types, enhancing its adaptability to diverse environments (Pearce et al., 2004). The clear dominance of one species in both Aspen and Spruce sites suggests similarities between these reclamation sites, despite many differences in soil handling and reclamation methods, distance from forested areas, dominant tree species, and cover of major vegetation groups. This suggests that disturbance history is one of the important contributors to epigaeic spider assemblages in young (under 10 years) forest reclamation sites.

Our finding of more homogeneous spider assemblages in reclamation sites than Reference is similar to other studies. Clearcutting at boreal stands in Alberta and Quebec resulted in homogenization of ground dwelling spider species (Buddle and Shorthouse, 2008). In Alberta, boreal ground dwelling spider communities shifted dominance patterns with diverse cover types and retention harvesting levels, and replacement of forest specialist spider species by open habitat generalist species (Pinzon et al., 2016, 2012). Plant communities on reclaimed limestone quarry sites were dominated by sown plant species while spider communities were dominated by locally abundant species with good dispersal capabilities (Wheater et al., 2000).

Our young reclamation sites, with high graminoid cover, might not have habitat requirements for species in other web building and hunting spider guilds. Habitat quality was a primary determinant of litter spider species distributions; canopy closure, litter development, and prey availability were associated with variation in litter spider species composition (McIver et al., 1992). Across the landscape in Yukon Territory in Canada, habitat had a greater influence on spider assemblages than elevation (Bowden and Buddle, 2010). Unharvested forests have more structurally complex habitats and microhabitats for web weaving spiders (Pinzon et al., 2016). Pinzon et al (2011, 2013) found spider assemblages across forest layers (overstory, understory, ground layer) in aspen and spruce, were influenced by forest cover type and harvesting treatments, and exhibited vertical stratification reflecting microhabitat variations and prey availability, contributing to unique spider communities in major forest layers (Pinzon et al., 2013, 2011). Clear cuts provide a range of niches but less vertical structure, and favour ground hunting spiders (Pearce et al., 2004). Our reclamation sites currently lack an overstory and understory, although shrub cover in Aspen is almost three times greater than Spruce. We expect continued growing canopy cover will introduce habitat complexity for specialist spider species and diverse spider guilds.

We expected spider assemblages in Aspen to be more similar to Reference than Spruce, because of its proximity to Reference, higher shrub cover, and higher native vegetation cover. Spider assemblages in Aspen were different from Spruce, but overall were more similar to Spruce than to Reference. Besides sharing similar disturbance history, both reclamation sites are young, at the time of sampling it had been 9 years since soil placement for the aspen site, 8 years for spruce. There is no canopy and both are still early successional stands with high herbaceous cover and an emerging shrub layer. Full recovery of forest spider assemblages will require significant time, strong negative effects were still observed five years post harvest at EMEND (Pinzon et al., 2016). Orthopteran and spider abundances showed reclamation efforts successfully recovered productivity of herbivores and predators, although composition remained distinct in reclaimed areas relative to an undisturbed native prairie even 30 years after pipeline reclamation in Northern United States Great Plains (Sylvain et al., 2019).

Reclamation sites differed in soil properties, specifically pH, soluble calcium, and soluble sulfate, and vegetation composition including non-native species, shrub cover, and graminoid cover. Soil structure, total plant cover, and amount of soil organic matter were significant in establishing arthropod communities in reclamation sites after disturbances from borrow pits at a Cold Lake in situ heavy oil facility (Hammond et al., 2022). We predict differences in spider assemblages between Aspen and Spruce will become much more pronounced over time, as the woody species outcompete the grasses, litter composition changes, and canopy forms and closes. Dominant tree species contribute to spider assemblage structure, with conifer dominated and deciduous dominated stands having different noted assemblages (Bergeron et al., 2013; Pearce et al., 2004; Pinzon et al., 2016). Spider assemblages are expected to transition from dominance by ground hunting spiders to a higher abundances of other web building and hunting guilds of spiders. McIver et al. (1992) observed a dynamic shift in litter spider guild composition along the successional gradient, with a transition from predominantly hunting spiders in 3 to 7 year old clear cut forests to sheet web, funnel web, and trapdoor spiders in 30 year old clear cut forests and old growth sites (McIver et al., 1992).

Aspen is in a later successional stage than spruce, indicated by higher shrub cover, lower non-native grass cover; colonization by forest spider species is facilitated by proximity to Reference. *Agroeca ornata*, a significant indicator for Reference, and *Allomengea dentisetis*, a significant indicator for Reference + Aspen, are colonizing nearby Aspen. These species are associated with deciduous stands, with *Allomengea dentisetis* showing dominance in deciduous stands (Buddle, 2001; Work et al., 2004; Bergeron et al., 2013; Pinzon et al., 2016). This demonstrates the importance of proximity to an undisturbed site to ecological succession and ecosystem recovery of some spider species after disturbance.

Species richness and abundance in Spruce were higher than expected given soil reclamation methods and distance from undisturbed areas and Reference. Proximity to the settling pond, higher cover of willow (*Salix* spp.), and highest moss cover, suggest higher overall water content, a factor known to influence spider assemblage structure (Matveinen-Huju and Koivula, 2008; Samu et al., 2021). The prevalence of edges in Spruce, such as those created by the nearby highway, aspen plantation, and gravel road, may provide diverse microhabitats. Spiders are sensitive to forest edge disturbance type, such as wildfire or clear cuts, and disturbance shape (Kowal and Cartar, 2012; Larrivée et al., 2008). However, impact to spider assemblages varied; forest spider species avoided disturbance edges from logging clearcuts, gravel roads, and buried gas pipelines (Kowal and Cartar, 2012). In variable retention harvesting edges, forest spider species were not excluded from the edge environment, and there was no invasion from adjacent open habitats (Pearce et al., 2005). Employing more rigorous sampling techniques, such as transects covering the entire reclamation area at predetermined intervals, could provide further insight on how edge effects in reclamation sites impact spider assemblages.

4.2. Spiders As Indicators

Many species were selected as indicators for both reclamation sites, these are likely open habitat species, generalist species, or both. Two indicator species were ambush hunters of the *Xysticus* genus (Thomisidae) and one indicator species was a ground hunter of the *Pardosa* genus, *Pardosa modica*. These ground hunting spiders do not build webs and colonize and establish in young forest reclamation sites. Interestingly, *Pardosa moesta* was not a significant indicator for reclamation sites, even though it was the most abundant spider collected in reclamation sites. *Pardosa distincta* was a significant indicator for Spruce, but no species were significant in Aspen. *Pardosa xerampelina* and *Pardosa mackenziana* were both significant Reference indicators. Other studies contradict our finding that *Pardosa xerampelina* is primarily a forest species. Pearce et al. (2003, 2005) found it dominated in some clear cut habitats and bare soil environments along with two other species, *Pardosa mackenziana* and *Pardosa moesta*. The low number of *Pardosa xerampelina* collected in reclamation sites could be related to high abundance of *Pardosa moesta*, indicating potential interspecies competition. Another significant reclamation site indicator, X*ysticus emertoni* has been associated with clear cut grass habitats and unharvested forest control sites (Pearce et al., 2004; Pinzon et al., 2016).

We identified one species we believe is more strongly associated with forest habitats than young reclamation sites, and thus sensitive to soil disturbances. *Diplocephalus subrostratus* (Linyphiidae) was primarily collected in Reference and was a significant indicator for Reference. Spider communities after forest harvesting were mainly dominated by ground runners, primarily Lycosidae, dominance decreased with reduced forest harvest; sheet and tangle weaver Linyphiidae had an opposite trend (Pinzon et al., 2012). *Agyneta fabra* (Linyphiidae), a sheet weaver, the strongest indicator for Aspen, supports a later successional stage relative to Spruce.

Our results highlight the importance of multiple indicator measures for assessing spiders for inclusion in land reclamation success monitoring. If spider indicators were measured based on classic diversity measures such as Shannon-Wiener index or species richness, Aspen and Spruce would be classified as recovered since values were similar to a nearby Reference. We suggest shifts in spider guild dominance across reclamation successional stages could be a feasible metric for reclamation monitoring. This would of course require research to monitor changes in ground hunting spider guild dominance in reclamation sites and determine thresholds and benchmarks for spider guild assemblage at successional stages. Closed canopy systems have greater functional diversity; sheet web weavers are prevalent and hunting strategies are more diverse than in open canopy environments (Košulič et al., 2016). Feeding guilds of ground dwelling spiders showed shifts, predominantly in the hunters and web builders, across the various forest harvesting methods, in spruce dominated forests in south-central Finland (Matveinen-Huju and Koivula, 2008).

Research on soil building materials, such as peat mineral mix and biochar, and their impacts on spider guild composition should be conducted, given spiders use downed wood material as habitat (Buddle, 2001). Coarse woody materials for soil amendment in land reclamation contribute soil organic matter and create microsites. Given the ease of passive pitfall trap spider collection it is feasible to collect spiders in reclamation sites and to develop a toolbox to identify major guilds or families. Spiders identified to family summarized spider responses to logging disturbances in an Argentinian piedmont forest (Alcalde et al., 2021). Extensive training and resources required for family level identification decrease the likelihood of incorporating this type of reclamation criterion into monitoring protocols. Spider species identification through DNA extraction from composite samples is considered a technically viable alternative; however, cost is significant and requires expertise to interpret results.

4.3. Spider Species Of Interest

Some spider species exhibited interesting trends of note. *Trochosa terricola* was a Reference indicator; however, it was also collected consistently in the reclamation sites and had relatively high abundance there. *Trochosa terricola* has been found in young forest stands, is negatively associated with canopy openness, and is positively associated with litter volume and dryer sites (Bouchard and Hebert, 2021; Samu et al., 2021). Our research results suggest *Trochosa terricola* is an overall habitat generalist that is not overly sensitive to soil disturbances. Another interesting species, *Cybaeopsis euopla*, was commonly referred to in the literature on reclamation or disturbance studies. We collected 33 individuals, mainly in Reference. It had high abundance in lowbush cranberry aspen dominated ecosites and is a dominant species and indicator for deciduous stands at EMEND (Work et al., 2004; (Bergeron et al., 2013; Pinzon et al., 2012; Work et al., 2004). Bouchard and Herbet (2021) found that *Cybaeopsis eupola* had the strongest association with warm microclimate. The low number of individuals collected in Reference is surprising, but is expected in reclamation sites. This could be related to the relatively small size of Reference, less than 1 ha, and edge effects that take away from forest interior and required habitat for forest specialist species. Large forest patches have diverse microhabitats and uncommon spider species, while smaller patches primarily support common species unless clustered with other small patches (Pearce et al., 2005).

5. CONCLUSIONS

Our study sheds light on spider responses to disturbances, colonization, and subsequent succession over time, particularly in surface coal mine reclamation sites in central Alberta. We found reclamation sites were predominantly inhabited by ground hunting spiders in the genus *Pardosa*, known to be open habitat generalist species favoring grass dominated areas. This shift in ecological function, with an abundance of ground hunters relative to sheet and space web weaving spiders, suggests a homogenization trend of spider assemblages in reclamation sites.

While there were no significant differences in alpha diversity indices between sites, rarefaction analysis suggests observed species richness may not accurately reflect actual species richness due to insufficient sampling. Although spider assemblages differed between sites, the discrepancy in spider identification (species collected from May to August in 2018) matching sampling effort (pitfall traps from May to October in 2018 and 2019) prevents making definitive conclusions about the impacts of reclamation treatments on entire spider assemblages.

Despite expectations of higher diversity in Aspen than Spruce due to the closer proximity of Aspen to Reference and higher shrub and native species cover, spider assemblages did not reflect this. Spruce exhibited more site heterogeneity due to its proximity to roads and the settling pond, suggesting complex factors influence spider assemblages in reclaimed systems.

Disturbance history emerged as an important factor in young reclamation sites, highlighting the importance of water retention, particularly for predicted drier growing seasons. The shift in spider guild dominance in forest reclamation sites underscores the potential of spiders as economically feasible and ecologically sensitive indicators for inclusion in reclamation monitoring.

6. TABLES AND FIGURES

Table 5.1. Summary of measured environmental variables. Topsoil (0 to 15 cm) samples were collected in July 2017. Vegetation assessments including percent cover by functional group were completed in July from 2017 to 2019. Values represent mean and standard error of the mean.

Parameter	Aspen	Spruce	Reference
Hydrogen Ion Concentration (pH)	6.60 (0.04)	7.01 (0.03)	5.28 (0.09)
Electrical Conductivity	0.95 (0.04)	0.86 (0.02)	0.63 (0.05)
Cation Exchange Capacity	33.52 (1.32)	35.19 (0.83)	40.53 (2.53)
Total Nitrogen	0.36 (0.02)	0.35 (0.01)	0.52 (0.03)
Total Inorganic Carbon	0.25 (0.01)	0.19 (0.01)	0.11 (0.01)
Total Organic Carbon	5.52 (0.40)	5.20 (0.24)	7.75 (0.52)
Soluble Calcium	132.86 (11.02)	102.81 (3.73)	73.67 (7.92)
Soluble Magnesium	30.81 (2.54)	29.02 (1.09)	26.14 (2.62)
Soluble Potassium	18.63 (1.84)	20.23 (1.41)	22.89 (2.77)
Soluble Sodium	22.41 (2.25)	17.58 (1.12)	49.55 (7.96)
Soluble Chloride	12.40 (0.82)	12.09 (0.41)	19.11 (1.13)
Soluble Sulfate	136.82 (20.62)	29.60 (1.78)	59.67 (4.26)
Moss Cover	5.8 (1.6)	12.0 (2.0)	5.2 (1.1)
Non-Native Forb Cover	8.1 (1.5)	6.5 (1.4)	-
Native Forb Cover	40.1 (2.5)	20.3 (1.7)	24.5 (1.6)
Non-Native Grass Cover	25.5 (2.0)	46.7 (2.1)	1.2 (0.2)
Native Grass Cover	12.7 (1.4)	5.8 (1.1)	2.3 (0.3)
Shrub Cover	49.1 (3.4)	16.9 (1.2)	81.1 (3.6)
Weed Cover	13.4 (1.3)	17.2 (1.8)	0.9 (0.2)

Table 5.2. Incidence and frequency based total species richness estimates and standard error.

	Observed Richness	Chao2	Jackknife
Aspen	55	73.9 (13.9)	77.1 (4.3)
Spruce	74	126.6 (25.8)	127.7 (8.5)
Reference	63	83.4 (15.9)	84.1 (4.2)

Table 5.3. Indicator species analysis examining spider assemblages. Values are spider species significant at α = 0.05, listed by descending aspecificity (A). Aspecificity is the probability that the surveyed site is part of the target site group, given the species presence; positive predictive value. Sensitivity (B) is the probability of finding the species in sites belonging to the site group. R = Pearson correlation between spider species and site, bolded values indicate a very strong positive association, correlation > 0.8. A total of 102 spider species were considered, 40 species were selected as indicators; ** p < 0.001, * p < 0.05.

	Species Name	A	B	R	p value
Aspen	*Clubiona riparia*	1.00	0.133	0.365	0.023*
	Praestigia kulczynskii	0.860	0.233	0.448	0.004*
	Agyneta fabra	0.702	0.700	0.701	0.001**
Spruce	*Robertus arcticus*	1.00	0.167	0.373	0.018*
	Micaria rossica	0.874	0.200	0.418	0.008*
	Haplodrassus signifier	0.833	0.167	0.373	0.034*
	Pardosa distincta	**0.790**	**0.833**	**0.812**	**0.001****
Reference	*Maso sundevalli*	1.00	0.367	0.606	0.001**
	Clubiona kulczynskii	1.00	0.300	0.548	0.001**
	Emblyna hentzi	1.00	0.233	0.483	0.003*
	Tunagyna debillis	1.00	0.167	0.408	0.009*
	Grammonota gigas	1.00	0.167	0.408	0.006*
	Pityohyphantes subarcticus	1.00	0.133	0.365	0.041*
	Ceraticelus fissiceps	1.00	0.133	0.365	0.034*
	Lepthyphantes alpinus	1.00	0.133	0.365	0.027*
	Bathyphantes pallidus	**0.968**	**0.667**	**0.803**	**0.001****
	Diplocephalus subrostratus	**0.944**	**0.967**	**0.955**	**0.001****
	Dictyna minuta	0.941	0.433	0.639	0.001**
	Robertus riparius	0.937	0.367	0.586	0.001**
	Ozyptila sincera canadensis	0.931	0.467	0.659	0.001**
	Pardosa xerampelina	**0.929**	**0.933**	**0.931**	**0.001****
	Pardosa mackenziana	0.919	0.367	0.581	0.001**
	Agroeca ornata	**0.918**	**1.00**	**0.958**	**0.001****
	Ceratinella brunnea	0.831	0.433	0.600	0.001**
	Cybaeopsis euopla	0.786	0.600	0.687	0.001**
	Microneta viaria	0.759	0.200	0.390	0.025*
	Hypselistes florens	0.756	0.200	0.389	0.022*
	Walckenaeria digitata	0.673	0.300	0.449	0.034*
Aspen + Spruce	*Xysticus ferox*	1.00	0.400	0.632	0.001**
	Islandiana flaveola	1.00	0.283	0.532	0.002*
	Thanatus formicinus	1.00	0.250	0.500	0.008*
	Pardosa modica	**0.978**	**0.933**	**0.955**	**0.001****
	Xysticus emertoni	**0.959**	**0.917**	**0.937**	**0.001****
	Thanatus rubicellus	0.935	0.483	0.672	0.001**
	Gnaphosa parvula	0.887	0.683	0.779	0.001**
	Zelotes fratris	0.881	0.300	0.514	0.029*
Reference + Aspen	*Allomengea dentisetis*	0.979	0.600	0.766	0.001**
	Kaestneria pullata	0.970	0.450	0.661	0.001**
	Bathyphantes concolor	0.878	0.667	0.765	0.001**
	Neriene clathrate	0.875	0.483	0.650	0.003*

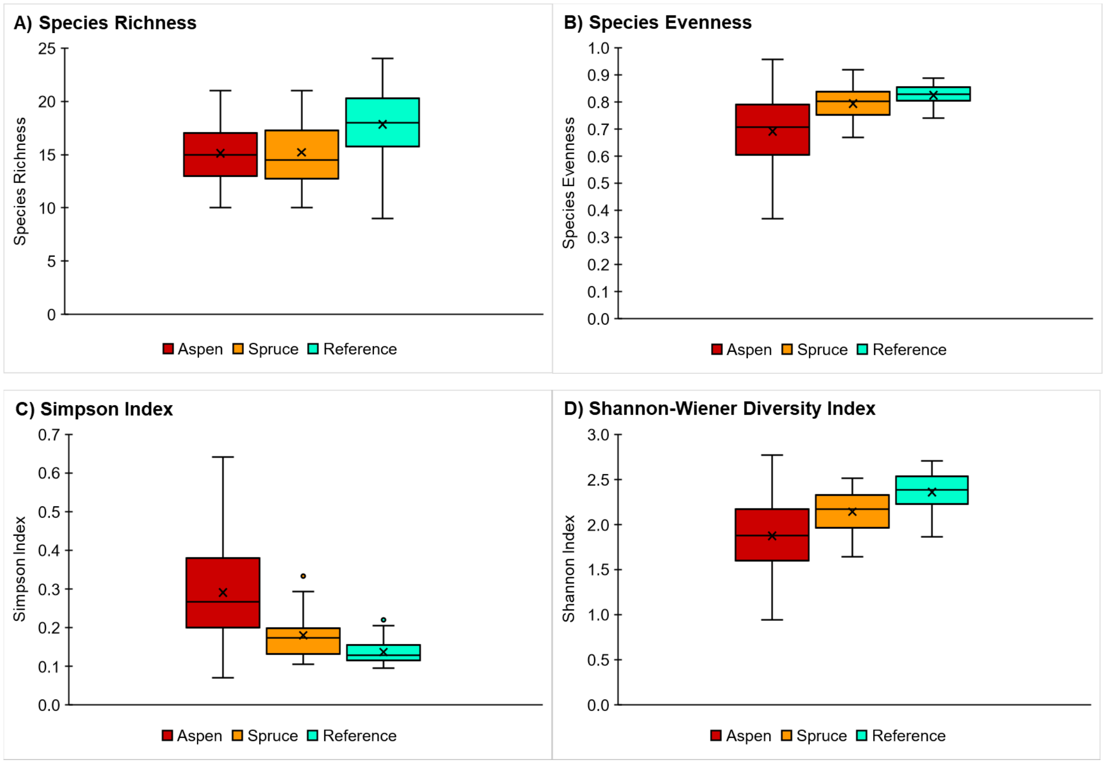

Figure 5.1. Boxplots illustrating alpha diversity indices including: A) Richness, B) Evenness, C) Simpson index, and D) Shannon index. Means are marked with an X, median values are the line, and the box represents interquartile ranges, standard error bars and outliers are included.

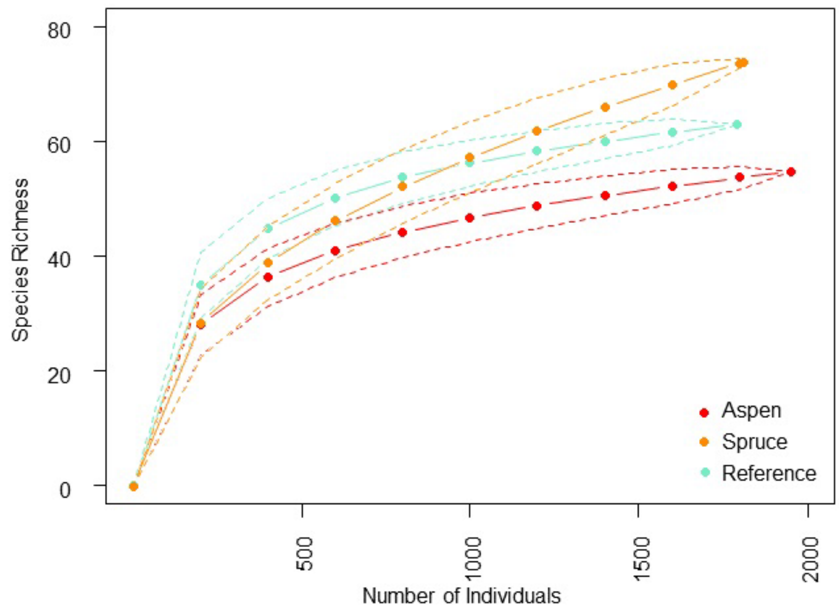

Figure 5.2. Total spider species richness estimation using individual based rarefaction. Dashed lines represent 95 % confidence intervals.

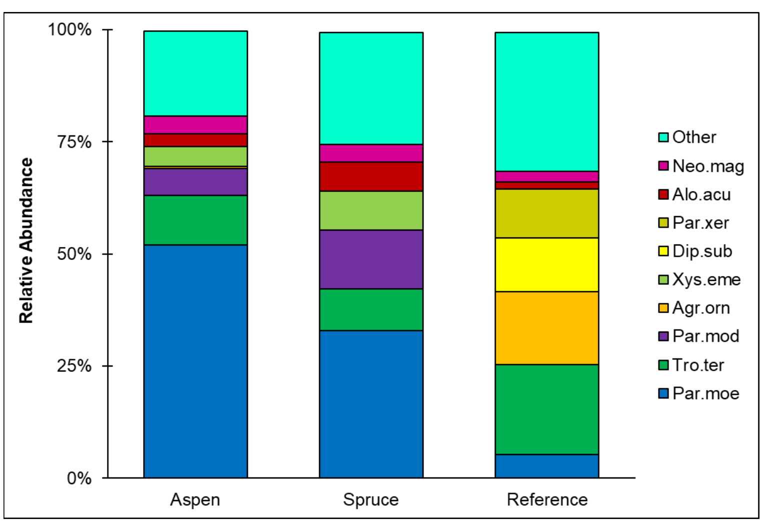

Figure 5.3. Relative abundance of the 9 most abundant spider species collected from May to August 2018. The Other category includes the remaining 93 spider species. Species abbreviations are as follows: *Pardosa moesta* (Par.moe), *Trochosa terricola* (Tro.ter), *Pardosa modica* (Par.mod), *Agroeca ornata* (Agr.orn), *Xysticus emertoni* (Xys.eme), *Diplocephalus subrostratus* (Dip.sub), *Pardosa xerampelina* (Par.xer), *Alopecosa aculeata* (Alo.acu), and *Neoantistea magna* (Neo,mag).

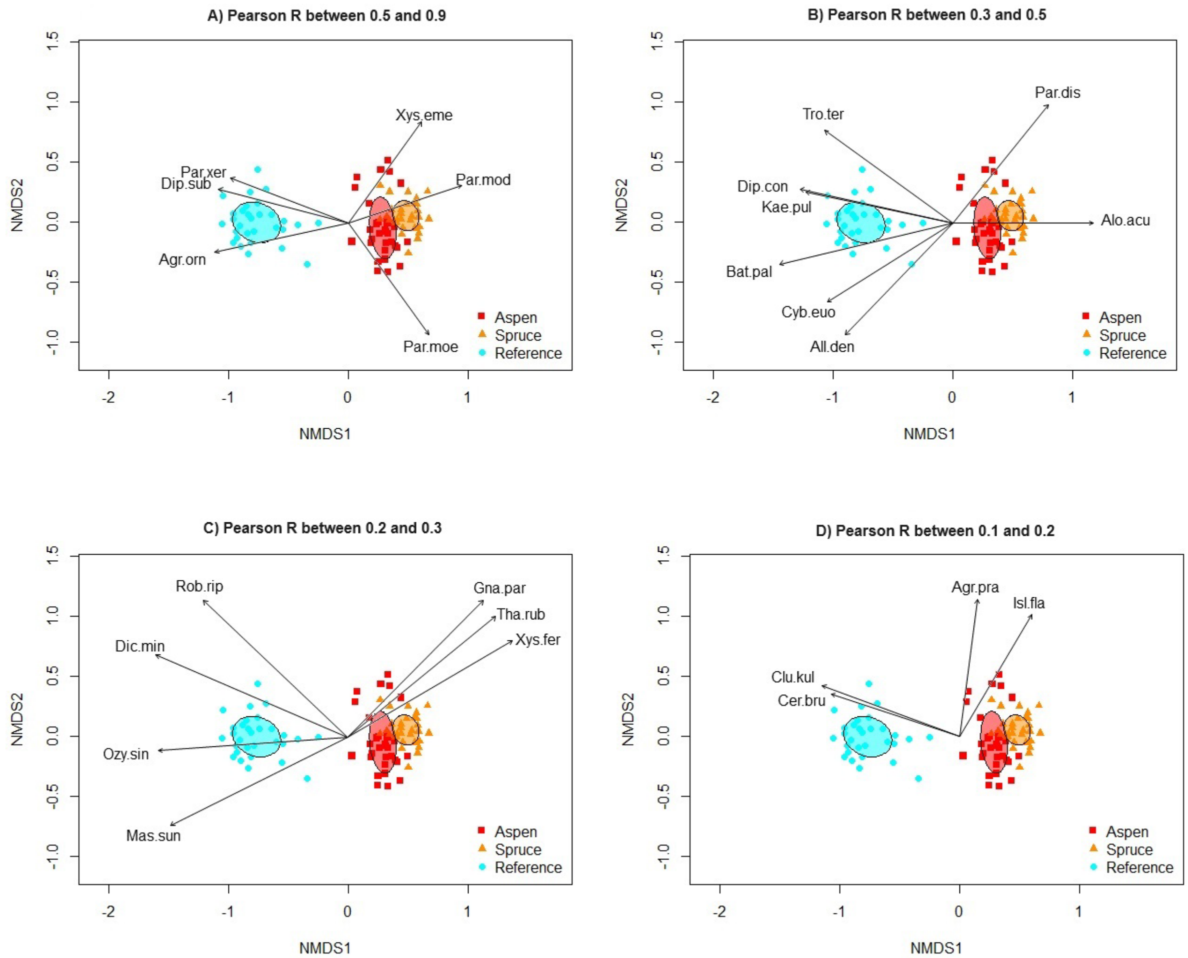

Figure 5.4. Non metric multidimensional scaling (NMDS) ordination of spider assemblages. Data were Hellinger transformed to improve final stress (13.6). Ellipses indicate 95 % confidence intervals around the group centroids for sites, vectors represent the strength of each significant (α = 0.001) spider species collected with a correlation value (Pearson R) of A) 0.5 to 0.9; B) 0.3 to 0.5; C) 0.2-0.3; and D) 0.1-0.2. Species abbreviations are as follows: *Pardosa moesta* (Par.moe), *Agroeca ornata* (Agr.orn), *Diplocephalus subrostratus* (Dip.sub), *Pardosa xerampelina* (Par.xer), *Xysticus emertoni* (Xys.eme), *Pardosa modica* (Par.mod), *Alopecosa aculeata* (Alo.acu), *Allomengea dentisetis* (All.den), *Cybaeopsis euopla* (Cyb.euo), *Bathyphantes pallidus* (Bat.pal), *Kaestneria pullata* (Kae.pul), *Diplostyla concolor* (Dip.con), *Trochosa terricola* (Tro.ter), *Pardosa distincta* (Par.dis), *Maso sundevalli* (Mas.sun), *Ozyptila sincera canadensis* (Ozy.sin), *Dictyna minuta* (Dic.min), *Robertus riparius* (Rob.rip), *Gnaphosa parvula* (Gna.par), *Thanatus rubicellus* (Tha.rub), *Xysticus ferox* (Xys.fer), *Ceratinella brunnea* (Cer.bru), *Clubiona kulczynskii* (Clu.kul), *Agroeca pratensis* (Agr.pra), and *Islandiana flaveola* (Isl.fla).

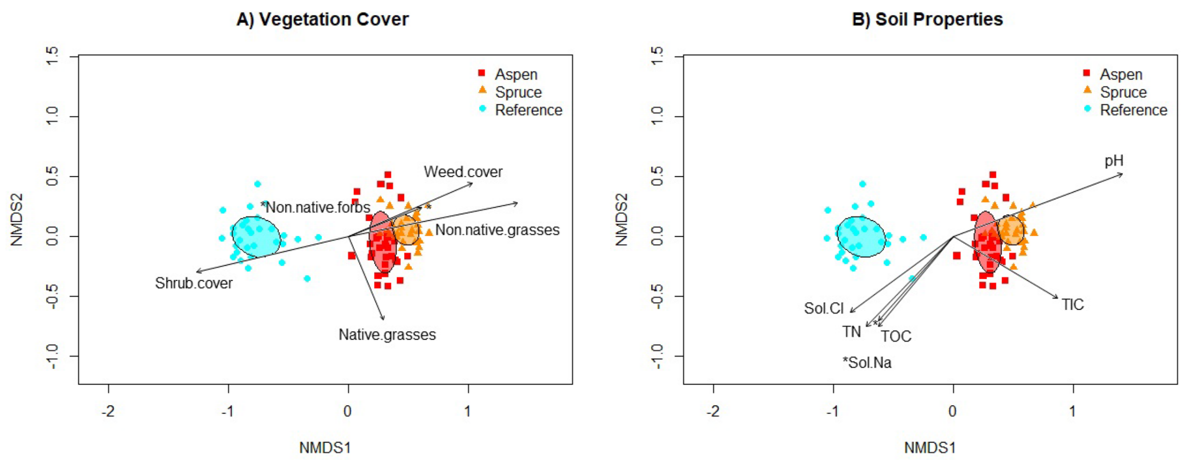

Figure 5.5. Non metric multidimensional scaling (NMDS) ordination of spider assemblages. Data were Hellinger transformed to improve final stress (13.6). Ellipses indicate 95 % confidence intervals around the group centroids for sites, vectors represent the strength of each significant (α = 0.001) environmental variables including: A) Percent cover of major vegetation ecological groups; and B) Soil chemical properties. Summary of vegetation and soil properties in Table 5.1.

Supplemental Table 5.1. List of spider species collected through pitfall trapping. A total of 102 species are organized by family and listed in alphabetical order. Identification information includes authority, and date. Ecological information provides functional role. * indicates singleton, ** indicates doubleton.

Family	Species Name	Taxonomic Authority	Ecological Information
Agelenidae	*Agelenopsis utahana*	Chamberlin and Ivie, 1933	Sheet web weaving
Amaurobiidae	*Amaurobius borealis*	Emerton, 1909	Sheet web weaving
	Cybaeopsis euopla	Bishop and Crosby, 1935	Sheet web weaving
Clubionidae	*Clubiona abboti**	Koch, 1866	Other hunters
	*Clubiona bishopi**	Edwards, 1958	Other hunters
	*Clubiona canadensis**	Emerton, 1890	Other hunters
	*Clubiona johnsoni**	Gertsch, 1941	Other hunters
	*Clubiona kastoni**	Gertsch, 1941	Other hunters
	Clubiona kulczynskii	Lessert, 1905	Other hunters
	Clubiona riparia	Koch 1866	Other hunters
Dictynidae	*Argenna obesa***	Emerton, 1911	Space web weaving
	Dictyna minuta	Emerton, 1888	Space web weaving
	Emblyna hentzi	Kaston, 1945	Space web weaving
	*Emblyna maxima**	Banks, 1892	Space web weaving
Gnaphosidae	*Drassodes neglectus*	Keyserling, 1887	Ground hunter
	*Gnaphosa borea**	Kulczynski, 1908	Ground hunter
	Gnaphosa parvula	Banks, 1896	Ground hunter
	Haplodrassus hiemalis	Emerton, 1909	Ground hunter
	Haplodrassus signifer	Koch, 1839;	Ground hunter
	Micaria pulicaria	Sundevall, 1831	Ground hunter
	Micaria rossica	Thorell, 1875	Ground hunter
	Zelotes fratris	Chamberlain, 1920	Ground hunter
Hahniidae	*Neoantistea agilis**	Keyserling, 1887	Sheet web weaving
	Neoantistea magna	Keyserling, 1887	Sheet web weaving
Linyphiidae	*Agyneta fabra*	Keyserling, 1886	Sheet web weaving
	Agyneta simplex	Emerton, 1926	Sheet web weaving
	Allomengea dentisetis	Grübe, 1861	Sheet web weaving
	*Baryphyma gowerense***	Locket, 1965	Other hunters
	Bathyphantes pallidus	Banks, 1892	Sheet web weaving
	*Ceraticelus atriceps**	Pickard-Cambridge, 1874	Other hunters
	*Ceraticelus crassiceps**	Chamberlin and Ivie, 1939	Other hunters
	Ceraticelus fissiceps	Pickard-Cambridge, 1874	Other hunters
	*Ceraticelus laetabilis***	Pickard-Cambridge, 1874	Other hunters
	*Ceraticelus similis***	Banks, 1892	Other hunters
	Ceratinella brunnea	Emerton, 1882	Other hunters
	Diplocentria bidentata	Emerton, 1882	Other hunters
	Diplocephalus subrostratus	Pickard-Cambridge, 1873	Other hunters
	Diplostyla concolor	Wider, 1834	Sheet web weaving
	Dismodicus decemoculatus	Emerton, 1882	Other hunters
	Erigone aletris	Crosby and Bishop, 1928	Other hunters
	Grammonota gigas	Banks, 1896	Other hunters
	*Helophora insignis**	Blackwall, 1841	Sheet web weaving

	Hypselistes florens	Pickard-Cambridge, 1875	Other hunters
	Islandiana flaveola	Banks, 1892	Other hunters
	Kaestneria pullata	Pickard-Cambridge, 1863	Sheet web weaving
	Lepthyphantes alpinus	Emerton, 1882	Sheet web weaving
	Maso sundevalli	Westring, 1851	Other hunters
	Mermessus triloblatus	Emerton, 1882	Other hunters
	*Mermessus undulatus***	Emerton, 1914	Other hunters
	Microlinyphia mandibulata	Emerton, 1882	Sheet web weaving
	Microneta viaria	Blackwall, 1841	Sheet web weaving
	Neriene clathrate	Sundevall, 1830	Sheet web weaving
	Pityohyphantes subarcticus	Chamberlin and Ivie, 1943	Sheet web weaving
	Pocadicnemis americana	Millidge, 1976	Other hunters
	*Porrhomma terrestre**	Emerton, 1882	Sheet web weaving
	Praestigia kulczynskii	Eskov, 1979	Other hunters
	*Scotinotylus sanctus**	Crosby, 1929	Other hunters
	*Soucron arenarium**	Emerton, 1925	Other hunters
	*Tapinocyba simplex**	Emerton, 1882	Other hunters
	Tunagyna debillis	Banks, 1892	Other hunters
	Walckenaeria atrotibialis	Pickard-Cambridge, 1878	Other hunters
	*Walckenaeria cornuella**	Chamberlin and Ivie, 1939	Other hunters
	Walckenaeria digitata	Emerton, 1913	Other hunters
Liocranidae	*Agroeca ornata*	Banks, 1892	Ground hunter
	Agroeca pratensis	Emerton, 1890	Ground hunter
Lycosidae	*Alopecosa aculeata*	Clerck, 1758	Ground hunter
	*Arctosa alpigena***	Doleschall, 1852	Ground hunter
	Pardosa distincta	Blackwall, 1846	Ground hunter
	*Pardosa furcifera**	Thorell, 1875	Ground hunter
	Pardosa fuscula	Thorell, 1875	Ground hunter
	*Pardosa groenlandica**	Thorell, 1872	Ground hunter
	*Pardosa hyperborea**	Thorell, 1872	Ground hunter
	Pardosa mackenziana	Keyserling, 1877	Ground hunter
	Pardosa modica	Blackwall, 1846	Ground hunter
	Pardosa moesta	Banks, 1892	Ground hunter
	*Pardosa ontariensis***	Gertsch, 1933	Ground hunter
	*Pardosa tesquorum**	Odenwall, 1901	Ground hunter
	*Pardosa uintana**	Gertsch, 1933	Ground hunter
	Pardosa xerampelina	Keyserling, 1877	Ground hunter
	Pirata piraticus	Clerck, 1758	Ground hunter
	Trochosa terricola	Thorell, 1856	Ground hunter
Philodromidae	*Thanatus coloradensis**	Keyserling, 1880	Other hunters
	Thanatus formicinus	Clerck, 1758	Other hunters
	Thanatus rubicellus	Mello-Leitão, 1929	Other hunters
	Thanatus striatus	Koch, 1845	Other hunters
	Tibellus maritimus	Menge, 1875	Other hunters
Phrurolithidae	*Scotinella pugnata*	Emerton, 1890	Ground hunter
Salticidae	*Evarcha proszynskii**	Marusik and Logunov, 1998	Other hunters
Theridiidae	*Crustulina sticta**	Pickard-Cambridge, 1861	Space web weaving
	Robertus arcticus	Chamberlin and Ivie, 1947	Space web weaving
	Robertus riparius	Keyserling, 1886	Space web weaving

	*Rugathodes aurantius**	Emerton, 1915	Space web weaving
Thomisidae	*Ozyptila gertschi**	Kurata, 1944	Ambush hunter
	Ozyptila sincera canadensis	Dondale and Redner, 1975	Ambush hunter
	Xysticus britcheri	Gertsch, 1934	Ambush hunter
	*Xysticus canadensis**	Gertsch, 1934	Ambush hunter
	*Xysticus discursans**	Keyserling, 1880	Ambush hunter
	*Xysticus elegans**	Keyserling, 1880	Ambush hunter
	Xysticus emertoni	Keyserling, 1880	Ambush hunter
	Xysticus ferox	Hentz, 1847	Ambush hunter
	*Xysticus luctuosus***	Blackwall, 1836	Ambush hunter
	*Xysticus obscurus***	Collett, 1877	Ambush hunter

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CHAPTER VI: RESEARCH SUMMARY AND RECOMMENDATIONS

1. INTRODUCTION

Land reclamation is a critical field that carries the weight of the world on its shoulders because it is responsible for mitigating and offsetting anthropogenic disturbances and impacts. Large scale disturbances are allowed to move forward because there is a promise that this area can be returned to equivalent land capability. We are gambling current ecosystems with hopes that future technology and human ingenuity will solve the problems we are struggling to answer today. Given that we are gambling on our future it is all the more important to know that reclamation methods and resources are being used efficiently and effectively to ensure maximum long-term ecosystem benefits. We want and need to create sustainable and resilient ecosystems that may not be the same as what was there before, but will have similar functions. We need to understand what reclamation success looks like, how we measure it to ensure success has been achieved, and the steps involved in achieving success.

Soil invertebrates are important to soil health and function; they are incredibly diverse, and in some cases well studied in various ecosystems and disturbances. Our research centered on soil and litter dwelling invertebrates and their response to reclamation practices and responses over the growing season in central Alberta, Canada. We wanted to explore whether soil invertebrates could be included in reclamation success monitoring, specifically if soil invertebrate metrics told the same story or added additional insight into reclamation trajectories as compared with commonly used metrics such as vegetation community composition and soil properties. We asked if there were soil invertebrate taxa with consistency and sensitivity to be included as indicators in reclamation monitoring. By assessing various soil invertebrate groups, we sought to identify the most robust candidates for inclusion in reclamation criteria. We aimed to compare the effectiveness of assessing abundance at the taxon level versus species level detail and evaluating community composition, particularly focusing on beetles and spiders, to provide insights into their utility for reclamation monitoring.

Our research was conducted at the Genesee coal mine, located approximately 70 km southwest of Edmonton, Alberta, in the Dry Mixedwood Natural Subregion of the Boreal Forest Natural Region. We had three research sites of comparable slope, aspect, topography, and drainage. The Aspen reclamation site had directly placed, salvaged, forest surface soil placed to a depth of 20 cm in January 2009; soil amendments were salvaged coarse woody debris and straw spread to a depth of 2 to 5 cm. Revegetation resulted from seeds and vegetative propagules from the forest soil seedbank and from non-native species from the surrounding agroecosystems. The Spruce reclamation site had stockpiled soils placed to15 cm depth in 2010, with white spruce (*Picea glauca*) trees planted in 2013. The forest Reference was close to the Aspen reclamation site and to a lesser extent Spruce, and was a relatively undisturbed late seral trembling aspen (*Populus tremuloides*) dominated mixedwood stand.

2. KEY RESEARCH FINDINGS

2.1. Oribatid Mites

Oribatid mites, simply identified to the coarse level of 'Oribatida exclusive of Astigmata', emerged as the most suitable indicators for assessing environmental conditions in our study. Their consistency across research sites and monthly and yearly variations underscore their reliability as indicators. They discriminated between Aspen and Spruce ecosystems, demonstrating sensitivity to habitat differences. The absence of clear seasonality trends facilitates sampling over the growing season, rendering them acceptable for long-term monitoring programs. Our research findings corroborate existing studies indicating that oribatid mites exhibit minimal sensitivity to seasonal variations, maintaining relatively consistent and stable abundance levels throughout the growing season from May to October. This consistency suggests a degree of resilience in oribatid mite populations. Such stability in abundance underscores the importance of oribatid mites as key components of soil ecosystems, contributing to nutrient cycling, decomposition, and overall soil health across various environmental conditions.

Challenges persist in utilizing oribatid mites in monitoring, primarily due to their high abundance and diversity in topsoil. While identifying oribatid mites relative to other mesofauna in soil and litter samples is relatively easy to do, their high abundance can make counting the number of individuals in a given sample very time consuming. The necessity of microscopic identification and differentiation from other small mesofauna, such as springtails and various other mite species, poses some serious practical hurdles. There are some limitations in utilizing environmental DNA for oribatid mites that stem from the incomplete knowledge of species and the absence of DNA profiles in existing databases. This highlights the ongoing need for DNA barcode database development and continuing species identification efforts. Oribatids mites from Alberta are well represented in the BOLD DNA library due to extensive work from the Alberta Biodiversity Monitoring Institute (https://www.boldsystems.org/, https://beta.abmi.ca/biobrowser).

2.2. Abundance Versus Assemblage Structure

Our research underscores the value of considering more than abundance measures when assessing reclamation success. While spider and beetle abundances initially suggested recovery in reclaimed sites relative to reference areas, closer examination of species and assemblage structure revealed that one species often dominated, significantly influencing relative abundance and discrimination of reference and reclaimed sites in multivariate space. This highlights the inadequacy of solely relying on the abundance of higher taxa for assessing soil invertebrate communities. Disturbance loving species such as *Pardosa moesta* and *Pterostichus melanarius* are likely to dominate other invertebrate taxa in reclaimed sites. This prompts the need for further investigation into the implications of such dominance on ecosystem function and biodiversity. Understanding the acceptable level of species loss in forest reclamation sites without compromising ecosystem function emerges as a crucial area for future research.

2.3. Importance Of Site Factors

Our research highlights the significance of site specific factors, particularly in young reclaimed sites, over different soil reclamation methods. Dominated by non-native agronomic and weed species, early stage reclaimed sites initially appear similar but diverge over time. Aspen sites exhibit higher shrub cover, native species cover, and faster juvenile growth of balsam poplar (*Polulus balsamifera*) and aspen (*Polulus tremuloides*) trees relative to white spruce (*Picea glauca*). Variability in soil water content, notably in Spruce sites, contributes to higher soil water content, possibly increasing species richness and diversity indices for beetles and spiders. Surrounding factors, including settling ponds, highways, and gravel roads impact Spruce sites, while Aspen sites are influenced by agriculture and stockpiled soils with weed species. These research findings underscore the necessity of considering site factors in reclamation planning and management associated with reclamation success.

Our findings highlight the importance of nearby undisturbed forest patches as dispersal sources for forest soil invertebrate species. This is evidenced by the presence of forest beetle and spider species in our Aspen sites but not in our Spruce sites. Further research is needed to determine the effectiveness and optimal distances to facilitate invertebrate dispersal into reclamation sites. Investigating the use of forest litter and forest topsoil islands to inoculate reclaimed systems presents a promising avenue for future research, aiming to assess its comparability to proximity to a nearby forested area for recruitment of invertebrates.

2.4. Direct Placement Versus Stockpiled Soils

While our study does not allow us to separate the effects of reclamation soil type, dominant tree planted, and proximity to Reference, direct placement of salvaged soil remains the preferred soil reclamation method, provided that the timing and planning align with operational requirements. This approach promotes greater native species diversity and accelerates forest succession during reclamation. Sites with directly placed salvaged soil exhibited higher abundance of oribatid mites than those with stockpiled soil. While the proximity of an undisturbed forest site is less critical for oribatid mites, it is significant for more mobile soil invertebrate taxa such as beetles and spiders.

Measuring soil invertebrate assemblages and abundance in the subsoil (below 15 cm) is important for understanding the complex reclaimed ecosystem dynamics. We found significant differences in subsoil properties, underscoring the necessity to assess their implications on soil invertebrates. Further investigation is imperative to understand soil invertebrate assemblages and their abundance in deeper soil layers below 30 cm. Challenges encountered during soil pit excavation, such as impenetrable clay and limited evidence of soil forming processes below 65 to 75 cm, highlight the need to determine maximum depth for soil invertebrate collection in reclaimed soils versus undisturbed soils.

3. RESEARCH LIMITATIONS AND FUTURE RESEARCH CONSIDERATIONS

3.1. Soil Invertebrate Identification

Some research limitations were related to soil invertebrate identification. Not separating spiders from harvestmen (Opiliones) resulted in these distinct taxa being grouped together, neglecting potentially significant trends, especially given the number of Opiliones in our reclamation sites. Our focus on assessing dominant invertebrate taxa with pitfall traps skewed the representation of all collectible taxa, limiting the comprehensiveness of our analysis. Some taxa with low abundance were consolidated under the generic category of Other, possibly overlooking their importance as potential indicators, akin to rare species. These limitations highlight the necessity for meticulous taxonomic categorization and a more inclusive approach in future ecological studies to ensure comprehensive data analysis and interpretation.

While our results suggest oribatid mite abundance shows promise as an indicator of reclamation success in young forest reclamation sites, its practical application is hindered by challenges in mite identification. Unlike other soil invertebrate taxa, oribatid mites exhibit strong sensitivity to site conditions, soil amendments, soil building materials, and native plant species. Previous research has advocated for the use of various soil invertebrate taxa as indicators, but the feasibility of species based identification in reclamation monitoring is limited. To address this lack of feasibility, there is a need for easily measurable indices that can be readily taught to reclamation practitioners and assessors, and easily incorporated into regular monitoring criteria. There are various options to achieve this. One potential solution lies in leveraging photo sorting algorithms, akin to those used in vegetation and insect identification apps (e.g. Critterpedia, iNaturalist, and PlantNet), to facilitate rapid and accurate identification of oribatid mites relative to other soil mesofauna, enhancing the efficiency and effectiveness of reclamation monitoring efforts.

3.2. Capturing Soil Biodiversity

There were research limitations to assessing complete soil biodiversity. The exclusion of soil bacterial and fungal assemblages from our assessment compromises the comprehensiveness of our analysis, potentially overlooking important ecological interactions and sources of site variability. Our collection methods failed to capture certain invertebrate groups integral to the soil community, such as nematodes, and thus our sampling did not fully represent the soil food web. Logistical constraints prevented sampling over winter, precluding insights into complete seasonal trends and potential impacts of climate change on soil invertebrate dynamics. While the separation of litter and soil samples provided valuable insights, the lack of deeper subsoil sampling limits our understanding of soil invertebrate assemblages beyond surface layers. Given significant differences in soil properties among sites, particularly in deeper subsoil layers, future research should investigate soil invertebrate dynamics throughout the soil profile, including reclaimed and undisturbed soils, to provide a more comprehensive understanding of ecosystem functioning and reclamation success. Human modified soils, Anthroposols, are becoming more common (Naeth et al., 2023, 2012), and thus understanding the impacts our approaches to soil building are having on soil invertebrate assemblages is important to ecosystem function.

3.3. Capturing Site Variability

Our research encountered limitations in fully capturing site variability. While we successfully documented soil and vegetation variability across the sites, the absence of daily soil temperature and soil water measures hindered our ability to test hypotheses regarding site hydrologic regimes. This data gap also limited our capacity to elucidate the observed site differences over the growing season. Transects across sites could have provided a more comprehensive understanding of site variability and shed light on edge effects, which may have contributed significantly to the variability within each site. Notably, all three sites featured areas of ecosystem transition, suggesting the importance of considering edge effects in future research endeavors. The lack of detailed reclamation methods for Spruce, including information on prior seeding and planting densities, posed challenges in interpreting the observed vegetation dynamics. For example, planting densities in Spruce deviated from forested reclamation standards due to leftover stock from Genesee, leading to uncertainty regarding the ecological implications of planting spruce seedlings in areas designated as rangeland rather than forest terrain.

3.4. Reclamation Treatments And Scale

In our research we encountered limitations concerning scale and reclamation treatments. The absence of site replicates, common in reclamation areas with limited soil building materials, posed challenges in generalizing our findings to other reclaimed areas. Given variability in soil building materials and revegetation methods used across sites, creating exact replicates was not possible, potentially limiting the extrapolation of our results. Our reclamation treatments also confounded reclamation soil type, dominant tree planted, and proximity to Reference, making it difficult to know which factor was driving site-specific differences. Our overall scale was relatively small, focusing on an in depth examination of specific areas. While this approach allowed for important detailed insights into site dynamics and processes, it did restrict the broader applicability of our research findings to larger spatial scales or to diverse reclamation contexts. Therefore, future research endeavors should consider broader spatial scales if they are available to enhance robustness and generalizability of findings in reclamation.

3.5. Oribatid Mites

Oribatid mites emerged as the most promising candidate among major epigaeic and soil dwelling soil invertebrate taxa for further research and monitoring efforts. Key areas for future investigation include understanding major oribatid mite successional stages in reclaimed systems, assessing the impacts of soil amendments and building materials such as biochar and peat mineral mix on oribatid mite assemblages and abundance, and developing a reliable toolkit comprising appropriate sampling methods and identification protocols that are cost effective and time efficient for potential users. There is a need for more research to establish links between oribatid mites and ecosystem functions, particularly those associated with soil carbon storage potential, similar to research by (Barreto et al., 2024).

The integration of oribatid mite abundance as an indicator in reclamation criteria faces significant challenges. These challenges are associated with the complexity of updating current reclamation criteria and regulatory requirements and the additional investment required, especially amidst industry struggles to meet existing criteria and achieve reclamation success. Addressing knowledge gaps surrounding oribatid mites in reclaimed systems in Alberta, encompassing various disturbances, soil reclamation methods, revegetation methods, and ecozones, necessitates further research. Incentivizing and encouraging the measurement and monitoring of oribatid mite abundance in reclaimed systems across Alberta is essential. This may involve incorporating them into biodiversity offsetting schemes if feasible, leveraging their presence to enhance public perception, and potentially granting the industry some social license.

4. APPLICATIONS FOR RECLAMATION

Based on our findings, I recommend monitoring ant abundance in pitfall traps at reclaimed sites, and if time and budget permit, oribatid mite abundance in reclaimed and reference soil samples. Continued monitoring of oribatid mites would provide a straightforward approach to establishing a trajectory of site age and mite abundance, offering insights into when reclaimed sites can be expected to reach similar abundance levels to reference sites. Key non-native beetle species should be monitored in reclamation sites. Species that are ecologically important, easily caught in pitfall traps, and readily identifiable should be prioritized. Species such as *Pterostichus melanarius*, which was abundant in our reclamation sites, are prime candidates. By focusing on key invertebrate taxa and species, reclamation practitioners can achieve reliable monitoring without time intensive identification of all soil invertebrate taxa and beetle species present.

The proposed soil invertebrate indicators should be assessed at reclamation sites to further validate our findings. This targeted approach would provide critical data on practicality and effectiveness of using ant and oribatid mite abundance as indicators while minimizing broad scale monitoring. Future research should focus on sampling at a single, standardized point in time, during periods of peak invertebrate activity (early growing season). This would allow for assessment of a diversity of sites, maximizing study breadth while providing meaningful insights into soil invertebrate assemblages and responses to reclamation. While it may not capture full seasonal dynamics, it would streamline the process, making it feasible for widespread application.

Our research findings offer valuable insights into the applicability of soil invertebrate dynamics to reclamation efforts, particularly in young forest reclamation sites. The added cost of incorporating soil invertebrates into reclamation monitoring is not appealing to industry and shareholders. Encouraging companies involved with reclamation to be innovative in research and experimental protocols could be a possible solution as it plays into the competitive nature between mining companies (first to reclaim a wetland or remediate a tailings pond.) Social licence could entice companies into using soil invertebrates in reclamation monitoring. Companies involved in resource extraction can have a negative global image due to environmental impacts. Providing a company the opportunity to announce they are going above and beyond required reclamation legislation and are creating diverse and sustainable ecosystems, could be invaluable to obtaining social licence with community, stakeholders, and international markets.

Soil invertebrate research needs to be more accessible and manageable with experimenter bias minimized or eliminated. Research can be streamlined using simplified specimen sorting manuals (Alberta Biodiversity Monitoring Institute, 2009, 2012), taxonomic keys (Cannings and Scudder, 2005; Walter et al., 2014; Walter and Lumley, 2021), and extensive picture databases of invertebrate groups (Meehan and Turnbull, 2017). Often the choice of soil invertebrate group to be studied has been related to interests of the researchers, availability of taxonomic expertise, and why diversity or composition of the particular taxonomic group might provide significant information (Andersen, 1999; Orabi et al., 2010). Support of broader and efficient taxonomic resources could eliminate experimenter bias, as researchers will not be limited by their taxonomic strengths, and could focus on multiple soil invertebrate groups. These changes will encourage more research focused in the area of soil invertebrates and reclamation methods and monitoring.

The observed loss of diversity in beetle and spider species in both of our reclamation sites, relative to the undisturbed reference, underscores the impact of reclamation on soil biodiversity. Despite variations in reclamation methods, location, and proximity to undisturbed areas, soil invertebrate assemblages in reclamation sites exhibited greater similarity to each other than to the reference. This convergence highlights the strong influence of dominant vegetation functional groups, such as extensive graminoid cover and non-native weedy species, on soil invertebrate communities in young reclamation sites. Consequently, effective management of these young reclamation sites becomes important, necessitating strategies to enhance soil invertebrate diversity through planting and soil building methods. One potential approach is to promote areas with higher soil water content through techniques that create microtopographic variability, such as small scale rough and loose soil handling methods, which could facilitate the introduction and proliferation of diverse soil invertebrate species. Incorporating such practices into reclamation strategies can contribute to fostering resilient and ecologically functional ecosystems in reclaimed areas.

Significant implications arise from our research results for reclamation efforts, particularly concerning soil invertebrate communities. The dominance of 'weedy' soil invertebrate taxa in reclaimed sites, possibly correlated with the high cover of non-native and agronomic grass and forb species, raises concerns. Research has highlighted apprehensions regarding arrested succession and the emergence of novel ecosystems in reclamation areas. Our findings suggest that this concern can be extrapolated to soil invertebrate assemblages, implying potential shifts in community composition and ecological dynamics in reclaimed environments.

The observed homogenization of beetle and spider assemblages, with one species dominating relative abundance, underscores the need to understand the implications for overall ecosystem function. Homogenization may lead to reduced functional diversity within reclamation sites, potentially impacting ecological processes such as nutrient cycling, decomposition, and predator-prey interactions. It becomes important for reclamation practitioners to consider consequences of assemblage homogenization and take proactive measures to promote biodiversity and ecosystem resilience in reclaimed areas. Strategies aimed at enhancing habitat complexity, fostering species diversity, and minimizing the dominance of single species could be effective in mitigating adverse homogenization effects on ecosystem function within reclaimed landscapes.

Reclaimed soil invertebrate assemblages were dominated by a non-native ground beetle species, *Pterostichus melanarius*. This was most obvious in peak growing season when temperatures were highest, and only subsided later in the growing season with lower temperatures. This has big implications when we consider climate predictions for the area. Future climate scenarios project significant changes in temperature patterns, including potential for early springs and notably short winter durations across various ecozones in southern and western Alberta, with winters consistently remaining above freezing, particularly in the western Prairie ecozone, where up to 21 % of the years between 2071 to 2100 are projected to have above- freezing temperatures (Newton et al., 2021). Projected climate change impacts are expected to lead to significant decreases in total forest biomass within the Boreal Plains ecozone, prompting consideration of planting more drought and fire tolerant species and emphasizing the importance of integrating climate projections into reclamation policies and practices for enhanced resilience (Nenzén et al., 2020). With predicted winters being shorter, it is more likely that dominance of *Pterostichus melanarius* will continue over the growing season, likely having negative impacts on native ground beetle species and assemblages. This species has not been reported in reclamation sites further north in the Athabasca oil sands region, but reclamation activities are likely to expand the range of nuisance species and therefore the importance of studying this species at this time.

Implications of our research for reclamation methods should be considered, particularly in understanding impact of disturbances and subsequent reclamation on oribatid mite populations. The observed decrease in oribatid mite abundance following disturbances underscores the vulnerability of these key soil organisms to environmental changes associated with land disturbances. Oribatid mite assemblages are likely dominated by species within the *Tectocepheus* spp. and smaller bodied oribatid mites, such as in the *Eniochthonius* genus. This observation emerged during initial stages of our research program when attempting species level identification of oribatid mites; further confirmation of these dominant genera is warranted to validate this. Changes in community composition, akin to those observed in beetle and spider assemblages, suggest profound shifts in soil invertebrate assemblages post-reclamation. Such changes not only have implications for Canadian mite diversity but also raise concerns regarding the potential loss of undiscovered species and the homogenization of soil invertebrate communities in anthropogenic systems. Urban soils in Baltimore, Maryland, had lower mite and springtail diversity in habitats with higher soil disturbance relative to reference (Huang et al., 2020). Our findings highlight the importance of incorporating considerations for oribatid mite populations and other soil invertebrates into reclamation planning and monitoring strategies to mitigate adverse impacts on soil biodiversity and ecosystem functioning during the reclamation process.

While soil invertebrates may have limited applicability in the immediate and practical goals of reclamation monitoring, they offer significant advantages in assessing the ecological integrity and progress of restoration projects. The diversity and ecological roles of soil invertebrates make them sensitive indicators of habitat quality, land management, and ultimately restoration success, highlighting the differences between the functional and ecological objectives of reclamation and restoration. Soil invertebrate indicators can be valuable in reclamation areas with specific end land use targets, sensitive habitats where similar ecological functions need to be demonstrated, or where mitigation requirements necessitate showing a net positive environmental benefit. Understanding these distinctions enhances the design and monitoring of both reclamation and restoration efforts, ensuring more effective and ecologically sound outcomes for both of them. There is a need for decision support tools to guide reclamation practitioners and regulators and remove subjectivity from the decision process (Tokay et al., 2019).

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APPENDIX: RESEARCH SITES INFORMATION

Table A.1. General horizon descriptions and characteristics for the Macola and Maywood Soil Series as reported in the 1968 Alberta Soil Survey Report No. 24: Soil Survey of the Buck Lake and Wabamun Lake Areas (Lindsey et al., 1968).

			Exchangeable (%)					
Horizon	Thickness (cm)	pH	Total % Nitrogen	Calcium	Magnesium	Potassium	Sodium	Description
Macola								
L-H	5							Deciduous leaf litter
Ah	12.5	5.7	0.37	64	13	3	1	Black (10YR 2/1, moist), silty clay loam, granular, friable
Ae	5	4.9	0.07	53	17	2	1	Light brownish gray (10YR 6/2, moist) silt loam, coarse platy, firm
AB	5	4.6	0.07	53	23	1	1	Light brownish gray (10YR 6/2, moist) silty clay, subangular blocky, firm
Bt	27.5	4.6	0.09	60	24	1	1	Dark brown (10YR 4/3, moist) heavy clay, blocky, firm
Ck	17.5	7.3	0.04					Dark grayish brown (10YR 4/2, moist) heavy clay, massive
Maywood								
L-H	2.5							Deciduous leaf litter
Ah	2.5	5.5	0.38	71	4	4	0	Black (10YR 2/1, moist) silty clay, granular, friable
Ae	7.5	5.5	0.11	60	20	1	0	Light brownish gray (10YR 6/2, moist) silty clay loam, coarse platy, friable
AB	7.5	5.2	0.10	58	20	1	0	Brown (10YR 5/3, moist) silty clay, subangular blocky, firm
Bt	25	5.0	0.04	54	21	1	0	Dark grayish brown (10YR 4/2, moist) heavy clay, prismatic breaking to coarse blocky, firm
BC	17.5	6.0	0.06					Dark grayish brown (10YR 4/2, moist) heavy clay, massive, firm
Ck	15	7.5	0.05					Dark gray (10YR 4/1, moist) heavy clay

Table A.2. General horizon descriptions and properties for soil profiles in research sites Aspen, Spruce, and Reference. Reclaimed soils are classified as Terro Fusco Spolic Anthroposols, while soil in Reference is classified as an Orthic Gray Luvisol.

Site	Horizon	Depth (cm)	Texture	Colour	Comments
Aspen					
	Dp	0-15	Loam	10YR 3/1 (m)	Some organic matter, medium to high root density, predominantly fine roots, granular structure
	D	15-66	Loam	10YR 5/3 (m)	5 % coarse fragments, compacted, medium to coarse angular blocky, low root density
Spruce					
	Dp	0-11	Clay loam	10YR 4/2 (m)	Little organic matter, medium to high root density, predominantly fine roots, granular structure
	D	12-62	Clay loam	10YR 4/3 (m)	7 % coarse fragments, compacted, massive structure, low root density
Reference					
	LFH	6-0			Deciduous leaf litter, predominantly L layer with smaller F and H layers
	Ae	0-19	Sandy loam	10YR 6/2 (m)	Weak, fine, platy; very friable; medium rooting density, rooting depth and size diverse
	Bt	19-51	Silt loam	10YR 3/4 (m)	Medium, fine subangular blocky; mostly firm, low to medium rooting density, coarse roots
	BC	51-58	Loam	10YR 5/4 (m)	
	Cca	58-70		10YR 5/6 (m)	Massive structure

Table A.3. Dominant soil textures and mean percentages of sand, silt, and clay for topsoil (0 to 15 cm) and subsoil (15 to 30 cm).

		% Sand	% Silt	% Clay	Texture
Topsoil	Reference	55.1	35.8	9.1	Sandy loam (65 %), Loam (35 %)
	Aspen	40.5	39.0	20.5	Loam
	Spruce	35.2	35.3	29.5	Clay loam (85 %), Loam (15 %)
Subsoil	Reference	42.1	46.4	11.5	Silt loam (30 %), Sandy loam (10 %), Loam (60 %)
	Aspen	42.1	36.1	21.8	Loam
	Spruce	36.1	31.1	32.8	Clay loam (90 %), Loam (10 %)

Table A.4. List of plant species identified during vegetation assessments. Species are alphabetically organized by native and non-native functional groups, included are common names and associated codes for analysis. Weed status represented as *Nuisance, **Noxious, ***Prohibited noxious.

		Scientific Name	Common Name	Code
Grasses and Sedges	Native	*Agropyron trachycaulum*	Awned wheatgrass	Agrtra
		Bromus carinatus	Mountain brome	Brocar
		Bromus ciliatus	Fringed brome	Brocil
		Bromus pumpellianus	Awnless brome	Bropum
		Calamagrostis canadensis	Bluejoint grass	Calcan
		Carex spp.	Forest sedge	Carspp
		Poa palustris	Fowl blugrass	Poapal
		Schizachne purpurascens	False melic	Schpur
	Non-native	*Bromus inermus*	Smooth brome	Broine
		*Bromus japonicas***	Japanese brome	Brojap
		*Elytrigia repens**	Quack grass	Elyrep
		Phalaris arundinacea	Reed Canary grass	Phaaru
		Phleum pratense	Timothy grass	Phlpra
		Poa compressa	Canada bluegrass	Poacom
		Poa pratensis	Kentucky bluegrass	Poapra
Forbs	Native	*Achillea millefolium*	Common yarrow	Achmil
		Agrimonia striata	Agrimonia	Agrstr
		Aster ciliolatus	Lindley's aster	Astcil
		Aster conspicuus	Showy aster	Astcon
		Aster laevis	Narrow leaved aster	Astlae
		Cornus canadensis	Bunchberry	Corcan
		Epilobium spp.	Fireweed	Epispp
		Equisetum arvense	Horsetail	Equarv
		Fragaria virginiata	Wild strawberry	Fravir
		Galium boreale	Northern bedstraw	Galbor
		Geum aleppicum	Yellow avens	Geuale
		Geum triflorum	Three-flowered avens	Geutri
		Heracleum maximum	Cow parnsip	Hermax
		Lathyrus ochroleucus	Cream colored pea vine	Latoch
		Maianthemum canadense	Wild lily of the valley	Maican
		Mentha arvensis	Wild mint	Menarv
		Mertensia paniculata	Tall lungwort	Merpan
		Mitella nuda	Bishop's cap	Mitnud
		Moehringia lateriflora	Blunt leaved sandwort	Moelat
		Petasites frigidus	Palmate Coltsfoot	Petfri

		Potentilla norvegica	Rough cinquefoil	Potnor
		Pyrola asarifolia	Common pink wintergreen	Pyrasa
		Ranunculus cymbalaria	Alkali buttercup	Rancym
		Sanicula marilandica	Snakeroot	Sanmar
		Smilacina stellata	Star flowered Solomon's Seal	Smiste
		Solidago canadensis	Goldenrod	Solcan
		Thalictrum spp.	Meadow rue	Thaspp
		Vicia americana	American vetch	Vicame
		Viola canadensis	Western Canada Violet	Viocan
	Non-native	*Chenopodium album*	Lamb's quarters	Chealb
		Medicago sativa	Alfalfa	Medsat
		Melilotus officinalis	Sweet clover	Meloff
		Polygonum convolvulus	Wild buckwheat	Polcon
		*Stellaria media**	Chickweed	Stemed
		*Thlaspi arvense**	Stinkweed	Thlarv
		Trifolium hybridum	Hybridum clover	Trihyb
		Trifolium pratense	Alslike clover	Tripra
Shrubs	Native	*Amelanchier alnifolia*	Saskatoon	Amealn
		Cornus stolonifera	Dogwood	Corsto
		Lonicera dioica	Twining honeysuckle	Londio
		Lonicera involucrata	Bracted honeysuckle	Loninv
		Ribes americanum	Black currant	Ribame
		Ribes oxyacanthoides	Gooseberry	Riboxy
		Rosa acicularis	Prickly rose	Rosaci
		Rubus idaeus	Wild raspberry	Rubida
		Rubus pubescens	Running raspberry	Rubpub
		Salix spp.	Willows	Salspp
		Symphoricarpos albus	Snowberry	Symalb
		Viburnum edule	Lowbush cranberry	Vibedu
Trees	Native	*Alnus crispa*	River/Green alder	Alncri
		Picea glauca	White spruce	Picgla
		Populus balsamifera	Balsam poplar	Popbal
		Populus tremuloides	Trembling aspen	Poptre
Weeds	Non-native	*Alliaria petiolate****	Garlic mustard	Allpet
		*Cirsium arvense***	Canada thistle	Cirarv
		*Erysimum cheiranthoides**	Wormseed mustard	Eryche
		*Galeopsis tetrahit**	Hemp nettle	Galtet
		*Ranunculus acris***	Tall buttercup	Ranacr
		*Silene latifolia***	White cockle	Sillat
		*Sonchus arvensis***	Perrenial sow thistle	Sonarv
		*Taraxacum officinale**	Dandelion	Taroff
		*Tripleurospermum inodorum***	Scentless Chamomile	Triino

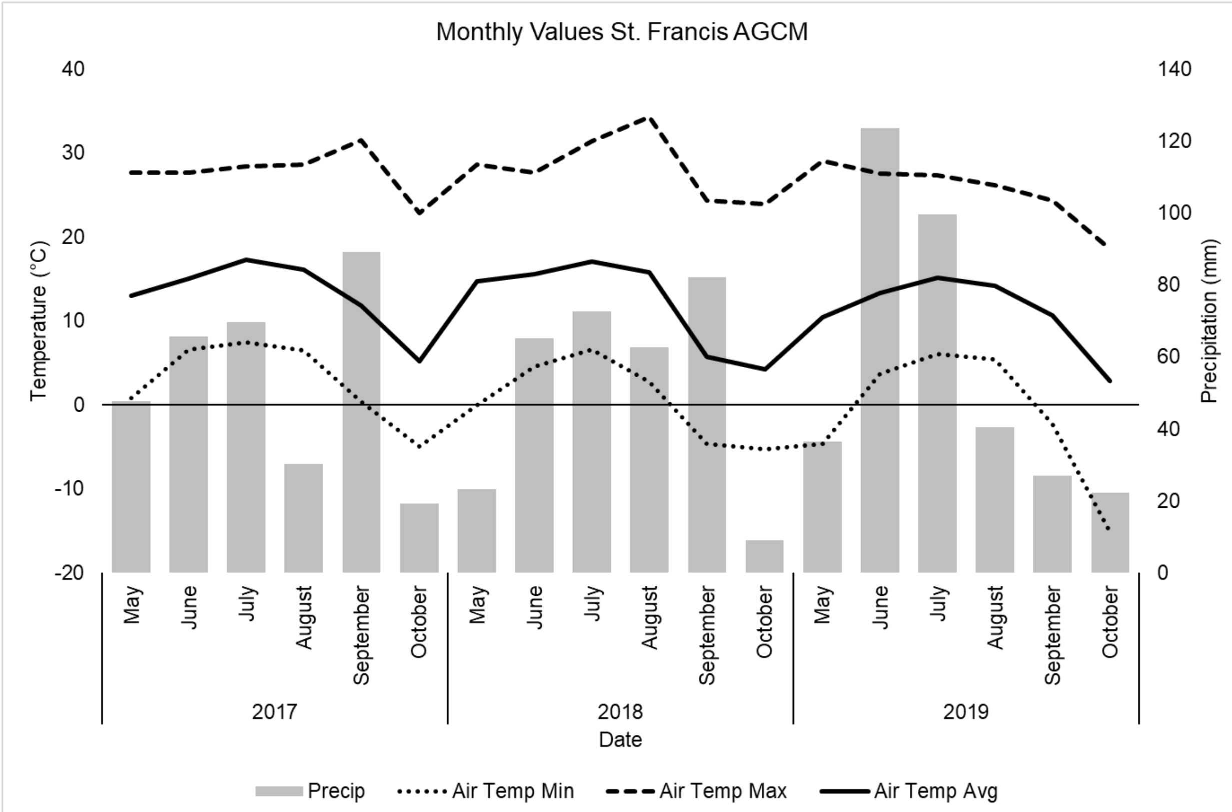

Figure A.1. Monthly air temperature and precipitation values recorded at St. Francis AGCM, 10 km east of the research area, during data collection years 2017 to 2019. Data retrieved from Alberta Climate Information Service (ACIS) on May 31, 2023 (Government of Alberta 2020; https://acis.alberta.ca/).