ECOLOGICAL RESTORATION OF DISTURBED LOW-ARCTIC UPLAND HEATH USING LOCALLY SOURCED, TRANSPLANTED VEGETATIVE TURFS

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ABSTRACT

Arctic environments have undergone ecological disturbances from industrial resource extraction for decades, yet knowledge of arctic plant-soil systems and effective means of ecological restoration is still largely unknown and understudied. To gain a better understanding of restoring arctic plant communities following mining disturbance, we examined whole turf transplants and shredded tundra material in maintaining vegetative community characteristics and soil nutrient concentrations two years post-transplantation onto disused gravel quarries. Community characteristics, and recovery of turf harvesting locations were determined through quadrat assessments, and soil nutrients were assessed through ion chromatography of soil samples. Additional turfs were harvested and transported to the University of Saskatchewan to investigate the effects of turf-adjacent fertilization on turf and substrate community characteristics, above and belowground biomass, and distance of vegetation expanding from the turf. Quadrat assessments were conducted to investigate community characteristics, and above and belowground biomass was harvested at specific distance increments from the turf. DNA metagenomics was used to identify the species responsible for expansion

Overall, we found turf transplants were capable of surviving transplantation and extreme environmental conditions and transferred native species and vegetative cover to disturbed sites. The application of shredded tundra material may be effective at re-instating non-vascular communities over a large area, although requires greater protection from wind and water erosion. We found belowground expansion far exceeded aboveground and that graminoids were primarily responsible for this expansion. Fertilization of turf's surroundings increases belowground biomass and the development of biological soil crusts on adjacent substrates, without impacting the development of vegetation within the turfs.

We recommend that restoration practitioners seek to transplant forb and graminoid-dominated communities, as these communities will likely i) survive transplanting better than shrub-dominated communities, ii) stimulate development of organic layers and soil nutrient enrichment, iii) introduce common and critical nitrogen-fixing species, and iv) present the greatest likelihood of vegetative expansion. Further research is needed to optimize this technique in arctic environments; however, the results of this research indicate that turf transplants can maintain key plant-soil interactions allowing for the continued survival of arctic vegetative communities, along with expansion, and modification of their immediate surroundings within disturbed sites.

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LIST OF ABBREVIATIONS

ANOVA	Analysis of Variance
BSC	Biological Soil Crust
C	Carbon
CT	Control Treatment
Ca^{2+}	Calcium
Cl^-	Chloride
DMS	Degrees, Minutes, Seconds
F^-	Fluoride
K^+	Potassium
Mg^{2+}	Magnesium
N	Nitrogen
Na^+	Sodium
NDVI	Normalized-Difference Vegetative Index
NH_4^+	Ammonium
NMDS	Non-Metric Dimensional Scaling
NO_2^-	Nitrite
NO_3^-	Nitrate
OUT	Operational Taxonomic Unit
PerMANOVA	Permutational Analysis of Variance
P	Phosphorus
PO_4^{3-}	Phosphate
Q1	Quarry #1
Q2	Quarry #2
S	Shredded Treatment
SIMPER	Similarity Percentage Analysis
SO_4^-	Sulfate

SOM	Soil Organic Matter
T	Turf Treatment
TC	Total Carbon
TIC	Total Inorganic Carbon
TIN	Total Inorganic Nitrogen
TN	Total Nitrogen
TOC	Total Organic Carbon
TON	Total Organic Nitrogen
TS	Turf + Shredded Treatment

1. INTRODUCTION

Low arctic environments are characterized by extreme conditions that result in slow rates of ecosystem recovery following disturbance. Therefore, restoring these environments can be very challenging. Short growing seasons and harsh climates (*i.e.*, low temperatures and low levels of precipitation) inhibit rates of decomposition, nutrient cycling, vegetative ("vegetative" is defined here and throughout this thesis as relating to vascular and non-vascular vegetation) colonization, seed production and maturation, and reduce the number of colonizing and functional species groups (Forbes & Jefferies, 1999; Naeth & Wilkinson, 2014; Mehlhoop, Evju & Hagen, 2018). Near-continuous permafrost and cryoturbation reduce soil aeration and drainage (Naeth & Wilkinson, 2014), and contributes to small-scale periodic disturbances and larger landscape-scale patterned ground. Large variations in light availability throughout the year and low aboveground photosynthetic biomass reduce rates of photosynthesis and subsequent carbon (C) inputs. Due to these environmental conditions, timeframes of decades to centuries following large-scale disturbances may be required for natural recolonization and ecosystem recovery (Forbes, Ebersole & Strandberg, 2001).

While small-scale disturbances within intact and functioning ecosystems may recover over time, large-scale disturbances can significantly reduce resource availability, connectivity with surrounding landscapes, and the presence of critical species. As such, active restoration of these systems is needed to return both native vegetative communities and natural ecosystem functionality within a reasonable timeframe. This is particularly important for industrial activities such as mining within northern environments, which are generally large-scale disturbances with various detrimental effects on plant establishment, including altered hydrology, coarser substrates, removal of organic soil layers, and removal of organic matter-enriched mineral soils (Miller, Naeth & Wilkinson, 2021). Mining activities often result in wide-scale removal of vegetation and natural microtopography, reducing overall ecosystem connectivity and the availability of favorable microsites. This necessitates the need to re-introduce native species and to assist with the regeneration of ecosystem functions, such as characteristic nutrient cycling regimes, C sequestration, vegetative production, and development of soil microbial communities.

Several common methods of ecological restoration have been attempted within northern climates such as seeding, fertilization, and individual species transplants, however, these have had conflicting results in recovering pre-disturbance conditions. For example, with seeding, invasive species can persist and may impede native species establishment (Hagen, Hansen, Graae & Rydgren, 2014; Kearns, Jean, Tissier & Johnstone, 2015; Rydgren *et al.*, 2016). Fertilization can promote the development of non-vascular and vascular communities, yet may significantly alter community structure and underlying soils (Kelley & Epstein, 2009; Gu & Grogan, 2020). In contrast, whole turf transplants (*i.e.*, harvesting and transplanting native vegetative communities, along with underlying organic and mineral soil layers) have not been as extensively investigated as seeding or fertilization, yet have shown promising results in the development of vegetative cover on disturbed arctic sites (Kidd, Streever & Jorgenson, 2006; Cater, Hopson & Streever, 2015). Due to the lack of published data on turf transplant success in different arctic vegetative communities, the efficacy of this restoration technique across differing communities is largely unknown, necessitating the need for further investigation across the circumpolar Arctic.

To further investigate the efficacy of turf transplants and to further develop our understanding of how this technique can aid in the establishment and development of low-arctic upland health vegetative communities, two studies were conducted, one *in situ* and one *ex situ*. The main research objectives were:

1. In an *in situ* restoration trial i) determine the efficacy of transplanted turfs and shredded tundra material in maintaining vegetative cover, community composition, and species richness on a disturbed upland heath tundra site, ii) examine soil nutrient concentrations within transplanted turfs, and iii) investigate the recovery of turf harvesting locations. We hypothesized that transplanted turfs would best maintain their vegetative cover and species richness, while community composition of the turfs and substrates would significantly change, with turfs promoting the increased cover of graminoids, and shredded tundra material would promote non-vascular cover. We also hypothesized that soil nutrient concentrations would not change significantly over two years and that moss species would represent the majority of vegetative cover within harvesting locations.

2. In an *ex situ* growth chamber i) determine the effect of turf-adjacent fertilizer application on turf and substrate community compositions, ii) examine the effect of fertilization on biomass expanding from transplanted turfs, and iii) identify the species responsible for this expansion. Fertilization was hypothesized to alter both turf and substrate community composition and

increase above and belowground biomasses exiting the turf. We also hypothesized that graminoid species would be primarily responsible for expansion.

This thesis is written in the manuscript format with five chapters. First, a general introduction is followed by a literature review focusing on ecological restoration, plant growth and community establishment in arctic environments, and previous methods of arctic restoration. This is followed by two research chapters, one investigating the efficacy of turf transplants in restoring species and vegetative cover in gravel quarries within a low-arctic environment, the next investigating belowground expansion from turfs and the effects of turf-adjacent fertilization. The last chapter will summarize key findings, discuss how these findings may advance our knowledge of turf transplantations within low-arctic environments, and suggest future research directions regarding ecological restoration within the low-arctic. A list of citations and appendices are included.

2. LITERATURE REVIEW

2.1 Defining Restoration Objectives and Approaches

Ecological restoration, as defined by the Society for Ecological Restoration's International Principles and Standards, is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER International, 2019, p. 15). Restoration, in general, seeks to develop the necessary components (nutrients, topography, biota, etc.) that would allow the return of assemblages of native vegetation to historical "pre-disturbance" conditions, and therefore re-create natural ecosystem services and function. A critical component of restoration is developing conditions that allow the re-instated communities to persist, evolve, and adapt to changing environmental conditions. Restoration is frequently used interchangeably with terms such as reclamation, remediation, or revegetation. However, these practices, while they may represent a component of ecological restoration, have specific definitions relating to the desired outcome. For example, reclamation is regarded as a restorative practice, however, it usually does not seek to return degraded or disturbed sites to pre-disturbance conditions. Rather, reclamation seeks to re-purpose degraded areas, such as creating limnological or wetland features from disused open-pit mining and tailings ponds. Remediation represents the practice of removal or effective neutralization of hazardous wastes that may present significant hindrances to future ecological development. Lastly, revegetation describes the practice of re-instating vegetative cover, regardless of pre-disturbance conditions (*e.g.,* preventing erosion of roadway-adjacent slopes through seeding of fast-growing, and erosion-resistant species).

"True" restoration, where a disturbed environment is returned to identical, or nearly identical, conditions before disturbances began, is often a challenging task. More so in remote locations such as the circumpolar Arctic, where knowledge of plant-soil, plant-plant, and plant-animal interactions is largely understudied (Virkkala, Abdi & Luoto, 2019). Knowledge is limited on which species, communities, or soil conditions are required for the regeneration of natural communities. In cases where knowledge is limited, or indeed where recovery to pre-disturbance conditions is not feasible (*i.e.,* where abiotic or biotic conditions have changed to prevent regeneration of pre-disturbance communities), similar functional ecosystems may arise. These alternative and stable communities are generally referred to as "novel" ecosystems, characterized by the emergence of different species, functions, and interactions previously unseen within the original pre-disturbance communities. While not representative of pre-disturbance conditions, novel ecosystems may present ecosystem functions and services similar to those of historical communities (Hobbs, Higgs & Harris, 2009; Perring, Audet & Lamb, 2014). A growing consensus among restoration scientists has begun to recognize that, while not representative of historical conditions, novel ecosystems may represent the colloquial "best-case scenario" for some degraded environments (Perring *et al*., 2014). Regardless of the desired or acceptable endpoints for restoration initiatives, the attributes of restored ecosystems should include the removal of threats hindering the establishment of vegetation, return of structural and biological diversity, connections and exchanges between neighboring environments, and development of ecosystem functionality, critical for the development of self-organizing systems resilient to further disturbances (Gann *et al*., 2019).

Identification of the most effective restoration approach, ranging from natural regeneration of disturbed sites to assisted regeneration to full reconstruction of the local environment is often a challenge for restoration practitioners (Gann *et al*., 2019). In cases of minimal disturbance or where specific activities are related to the disturbance, cessation of these activities, removal of the source of degradation, and facilitating natural regeneration may be appropriate. Many of the key components of the original ecosystem may still be present, such as topsoil and associated seedbanks, habitat connectivity, and continual transfer between nearby undisturbed environments. However, in cases of more extreme disturbance, practices such as active chemical remediation, the building of artificial habitats, and reintroduction or reinforcement of species may be warranted to regenerate the necessary components for ecological recovery (Gann *et al*., 2019). While these approaches certainly differ in the degree of active assistance, combinations of practices can be intertwined as restoration practitioners see fit. One common example of combined practice is applied nucleation, which consists of implementing small patches of vegetation throughout disturbed areas (Gann *et al.*, 2019). This practice seeks to then rely on these established pockets or islands of vegetation to further expand into disturbed areas. This may be represented directly through the development of above and belowground structures or indirectly through the production and dispersal of propagules and attraction of dispersing agents such as local wildlife. Patches of habitat may also alter the immediate surroundings, facilitating the arrival of species through amelioration of soil conditions, nutrient enrichment, and development of associated microbial communities. Regardless of specific methods of restoration, knowledge of the ecosystem being restored can help practitioners and researchers determine the best course of action.

2.2 Nutrient Availability and Plant-Soil Interactions within Arctic Environments

Arctic tundra is often characterized by low rates of primary production, largely due to the strong limitations of available nutrients. This is not to say that these systems are inherently nutrient-poor, indeed, arctic soils are widely considered to contain up to one-third of the global soil carbon (C) budget (Schuur *et al*., 2015; Jahn, Sachs, Mansfeldt & Overesch, 2010; McCulloch *et al*., 2021). Critical nutrients including C, nitrogen (N), and phosphorus (P) are instead "locked-up" within slowly decomposing organic layers and soil organic matter (SOM) and largely unavailable for plant uptake (Jonasson, Michelsen & Schmidt, 1999; Chu & Grogan, 2010; Zhao, Sundqvist, Newman & Classen, 2018). The overall concentration of nutrients within both organic layers and SOM can be very significant, with previous studies finding that as little as five percent of total N is contained within plant tissue, and the remainder residing within microbial populations, litter, and SOM (Jonasson *et al*., 1999). Abiotic factors including harsh climates (low temperatures, low precipitation, short growing seasons, high winds) (Forbes & Jefferies, 1999) are largely responsible for slow organic degradation and therefore slow nutrient cycling regimes. However, vegetation is also known to influence decomposition, and different communities can result in markedly different nutrient availabilities and net mineralization rates (Gough, Shaver, Carroll, Royer & Laundre, 2000; Eskelinen, Stark & Männistö, 2009; Chu & Grogan, 2010; Sundqvist, Wardle, Vincent & Giesler, 2014; Zhao *et al*., 2018; Hicks, Rousk, Rinnan & Rousk, 2020).

Forb, graminoid, and deciduous shrub-dominated communities are known to produce litter that decomposes faster than litter derived from woody, evergreen plants (Eskelinen *et al*., 2009; Chu & Grogan, 2010; Sundqvist *et al*., 2014). This is mainly due to forbs and other herbaceous plants containing tissue characterized by low C:N ratios, resulting in labile, N-rich organic matter that can be rapidly assimilated by microbial populations. In contrast, woody tissues are known to produce far more recalcitrant material characterized by high concentrations of stable compounds including lignin and tannins (Eskelinen *et al*., 2009; Chu & Grogan, 2010; Sundqvist *et al*., 2014). The biochemical characteristics of soils often reflect this difference in the vegetative community, with forb-dominated systems often containing higher concentrations of available N, low C:N ratios, higher pH values, bacterial-dominated microbial communities, and greater net N mineralization. Shrub-dominated communities often demonstrate the opposite, with underlying soils containing low concentrations of available N, high C:N ratios, low pH, fungal-dominated microbial communities, and lower net N mineralization (Eskelinen *et al*., 2009; Chu & Grogan, 2010; Sundqvist *et al*., 2014). This connection between soil properties and vegetation types appears to have a strong feedback loop. Herbaceous communities producing comparatively higher quality, N-rich organic matter that is rapidly recycled by bacteria, allows for the continued domination of forbs with higher nutrient requirements. Similarly, shrub-dominated communities produce lower-quality N-poor compounds that are slowly decomposed by fungi that favor low soil pH, such as mycorrhizal fungi, which are known to form strong relationships with many ericaceous arctic shrubs (Eskelinen *et al*., 2009; Zhao *et al*., 2018).

Microbial populations can act as nutrient sinks within arctic environments, further decreasing plant-available nutrients. A substantial lack of N mineralization during the growing season and a large increase during non-growing seasons (Jonasson *et al*., 1999) can be explained by the high efficiency of microbial communities to immobilize available N, often as fast as N is released (Sundqvist *et al*., 2014; Zhao *et al*., 2018; Hicks *et al*., 2020). This quick microbial uptake is further demonstrated by fertilization experiments in which plants responded only after greater fertilization rates than would normally be required for annual plant uptake (Jonasson *et al*., 1999). The increase of mineralization outside of the growing season is generally assumed to be caused by the decline of microbial populations after the growing season and the subsequent release of N. Previous research has shown that N mineralization is largely dependent on both temperature and quality of organic matter present within litter (Sundqvist *et al*., 2014) and that both N and P are rapidly re-absorbed within the decomposer food web for extensive timeframes (Jonasson *et al*., 1999).

2.3 Plant Establishment and Successional Models

Non-vascular components, generally cyanobacterial mats, are often the first vegetative organisms to colonize newly uncovered or disturbed sites within arctic environments (Hodkinson, Coulson & Webb, 2003; Breen & Lévesque, 2006). Recognized as pioneering communities, these non-vascular assemblages are responsible for modifying the surrounding environment and facilitating the development of vascular species (Williams *et al*., 2017; Agnelli, Corti, Massaccesi, Ventura & Acqui, 2021). Biological soil crusts (BSC) are tightly-bound surface-layer amalgamations of cyanobacteria, bacteria, fungi, various bryophytes, and lichens, found in almost all terrestrial ecosystems (Bowker, 2007). While the majority of BSC research has been conducted on communities within dry-land ecosystems (Antoninka *et al*., 2020; Bowker, Antoninka & Chuckran, 2020), increasing numbers of studies are being conducted within arctic and alpine environments (Gold, Dickson & Glew, 2001; Breen & Lévesque, 2006; Stewart, Coxson & Grogan, 2011a). Biological soil crust communities have been well documented to aid in enhancing resistance to erosion through increased soil stability (Bowker *et al*., 2020) and soil particle aggregation from the secretion of extra-cellular polysaccharides (Belnap, 2006), altering rates and pathways of water infiltration and resulting in greater soil moisture retainment (Antoninka *et al*., 2020; Gold *et al*., 2001). They also protect and promote seed germination through soil temperature regulation (Gold, 1998), the capture of wind-blown seeds (Breen & Lévesque, 2006), and contribution to landscape-scale N fixation within low-arctic tundra (Stewart *et al*., 2011a; Stewart, Lamb, Coxson & Siciliano, 2011b). Increased soil moisture and soil N concentrations underlying BSC communities have shown a positive relationship with the increasing cover of several common colonizing and late-stage successional arctic species. This suggests the benefits of BSC cover within arctic environments are not only limited to alteration of underlying substrates, and influences on vascular community composition and structure may extend well-past initial colonization (Breen & Lévesque, 2006)

Forbs, graminoids, and deciduous shrubs frequently arrive alongside or after the establishment of BSC communities (Hodkinson *et al*., 2003). Several of these species, particularly *Salix arctica* and *Salix oppositifolia*, are regarded as both pioneer species and continued members of late-stage climax communities (Breen & Lévesque, 2006; Mori, Osono, Uchida & Kanda, 2008, Mori, Uchida & Kanda, 2013). This suggests that several species that arrive early in succession are not always replaced by future members of the community, and like BSCs, may present significant influences on community structure that extend past initial vascular colonization. Further modification from labile forbs and graminoids include the development of SOM, organic soil layers, and microbial communities, all of which can assist in the establishment of many late-stage species, including *Dryas integrifolia, Cassiope tetragona*, and *Bistorta vivipara* (Hodkinson *et al*., 2003; Breen & Lévesque, 2006). The development of characteristic late-stage communities, often represented by the significant cover of evergreen shrubs, is known to be a long process with Hodkinson *et al*. (2003) reporting 100 years before the establishment of 100% vegetative cover on proglacial arctic chronosequences. An additional 50 years was required before the establishment of characteristic late-stage vegetation, such as *C. tetragona*, along with significant depths of organic soil layers, SOM, and soil acidification. However, the establishment of *C. tetragona* and *D. integrifolia* was noted after only ~40 years within a polar oasis (Jones & Henry, 2003), suggesting that the timeframes for natural establishment can vary widely depending on the climate and region-specific environmental conditions.

Several different models of plant succession have been proposed in arctic ecosystems. For example, three separate studies, all conducted on Ellesmere Island, Nunavut, observed three separate successional patterns. Breen and Levesque (2006) found evidence to support directional-replacement, Mori *et al*. (2008) found evidence to support directional-without replacement, and Jones and Henry (2003) found evidence of directional-replacement, directional non-replacement, and non-directional non-replacement models. The observations of Jones and Henry (2003) note that directional-replacement models were observed within polar oases where environmental stressors were not so severe as to eliminate species, while directional non-replacement and non-directional non-replacement models were observed within harsher polar-desert environments. Environmental stressors within these environments exert considerable challenges for vegetative establishment and survival, allowing the establishment of vegetation during optimal conditions, and subsequent decreases or removal if conditions deteriorate. It can be assumed, with the more favorable conditions within low-arctic environments, that plant succession likely follows a directional-replacement model, where species that arrive first modify their immediate surroundings to inhibit or facilitate the arrival of new species. However, what species impart which forces and species-specific requirements for establishment are still under investigation. Microsite modification may largely dictate which species can colonize newly modified sites, and subsequently direct future community development.

2.4 Belowground Establishment

Within arctic environments, root growth is an important aspect of recolonization of disturbed areas, as plant roots are likely to provide the impetus for the generation of belowground organic matter and nutrient cycling regimes. With a greater understanding of how arctic plants establish and grow both above and belowground, restoration practitioners can take advantage of species-specific traits, such as the type of root growth best suited for excavation and transplanting, best suited for generation of root exudates, and/or rapid expansion. Understanding species-specific soil requirements for root growth and the expected interactions between plants and the surrounding environment may assist in determining optimal communities to be promoted. In the review of root behavior and mechanisms of growth by Novoplanksy (2019), it is stated that the structure and distribution of plant roots are directly determined by the genetic makeup of any species. Yet root development is further determined by the interactions of a plant with its surrounding environmental conditions, a phenomenon termed “phenotypic plasticity”. This plasticity results in the ability to exploit resources within the near vicinity, to grow towards areas rich in resources (*i.e.*, water, nutrients, oxygen), and to sense the physical makeup of the substrate with preferential growth directed towards areas of least resistance (Novoplansky, 2019). Root growth and spatial distributions are influenced by light stimulation, chemical cues, and even electrical fields. Evidence suggests that plants can recognize root systems belonging to not only themselves, but also to others of the same species, and different species altogether. This recognition can result in many differing responses, including inhibition of the others' growth or increasingly competitive behaviors, with the highest degrees of competition correlated to the geographic distances between species' native ranges (Novoplansky, 2019). Plants have been shown to respond to chemical cues from each other, different species, or interconnected mycorrhizal associations. Further studies are needed to examine factors determining arctic root growth, structure, and distribution as they reside within a complex mosaic.

Aboveground growth and reproduction of arctic plants have been more extensively examined than belowground establishment and growth. Regardless, there are several important generalizations regarding arctic root growth that may assist restoration practitioners. Ratios of belowground:aboveground biomass generally range from 2-4:1(Kummerow & Russell, 1980) yet have been estimated at proportions of up to 90% (Iversen *et al.*, 2015; McCulloch *et al.*, 2021). Forb species generally demonstrate the lowest belowground biomass ratios and sedges the highest. Despite the high ratios of belowground to aboveground biomass, total belowground biomass is still comparatively low compared to more temperate and productive systems. Root growth in arctic systems usually lags behind aboveground growth and has been demonstrated to stop with the onset of leaf senescence, although roots may remain active after leaf senescence, even if not actively growing (Iversen *et al*., 2015). This cessation of root growth with leaf senescence occurs even when soil temperatures are considered adequate for the formation of root structures (Kummerow & Russell, 1980; Iversen *et al*., 2015). Conversely, more recent evidence suggests that root growth and aboveground characteristics may not be as tightly coupled, with no evidence of root growth coinciding with earlier leaf-out in sub-arctic heath (Blume-Werry, Jansson & Milbau, 2017). As rooting dynamics are still under investigation, and roots comprise the vast majority of arctic biomass, it is important for restoration practitioners to focus not only on the success of aboveground vegetation but also actively encourage and monitor the development of belowground structures.

In comparison to other ecosystems, root distribution within arctic environments is constrained within the top 30 cm of the soil profile, with previous studies indicating that up to 93% of roots were found within this depth range (Jackson *et al*., 1996). This shallow root system is in response to soil moisture, air temperature, soil insulation, and seasonal shifts in the depth of arctic soils active layer (*i.e.*, the depth of thaw during the growing season). Frost heaves, rocky terrain, ephemeral to permanent pooling, nutrient availability, and soil temperature gradients can all act to influence root distribution in arctic environments (Iversen *et al*., 2015). Along with leaf senescence, root growth has also been correlated with light availability, with day lengths of less than 15 hours resulting in growth cessation during controlled conditions. Root turnover rates are species-specific, with most phenotypic groups having an estimated root lifespan of more than five years, yet a few exceptions with lifespans of one to two years. Regardless of lifespan, decaying roots are an important aspect of arctic soils, as arctic roots often contain higher nutrient concentrations than temperate species, together with relatively large root biomass, and the slow release of these stored nutrients through decomposition (Iversen *et al*., 2015; McCulloch *et al*., 2021).

Arctic soils are often regarded as nutrient-limited, and many plants have developed specific traits to counteract this limitation, including species-specific preferential uptake, higher rates of exudate release, extreme internal nutrient cycling, and strong mycorrhizal relationships. There is evidence that some tundra species receive up to 86% of aboveground N from these fungal associations and can improve their access to P (Timling *et al*., 2012; Iversen *et al*., 2015).

Shrubs are reported as having the highest amount of mycorrhizal biomass, while forbs the lowest. The distribution of roots may be influenced by competition from surrounding species, with deeper-rooted distributions theorized to benefit from access to N that had been frozen in the slowly thawing active layer. Many tundra species rely heavily on internal stores of C and other nutrients, with some arctic sedges capable of surviving for several years on internal nutrient supplies (Iversen *et al*., 2015).

2.4.1 Characterizing Belowground Communities

Investigating belowground community composition is a challenging task, albeit an important one, given the dominance of belowground biomass in arctic systems, as well as investigations demonstrating belowground community composition may differ significantly from aboveground community compositions (Hiiesalu *et al*., 2012). Often analyzed through the physical excavation of root systems, visual identification, or the use of imaging systems such as minirhizotrons, these methods can be destructive, time-consuming, and labor-intensive, and may not present enough information to make holistic determinations of belowground compositions (Kesanakurti *et al*., 2011; Lamb, Winsley, Piper, Freidrich & Siciliano, 2016). Investigations of belowground communities are becoming more easily accessible using DNA barcoding and metagenomics. Unfortunately, there remains significant sources of error derived from the multiple stages of DNA barcoding (Aird *et al*., 2011), such as human error and laboratory contamination when extracting DNA, and biases from sequencing platforms and amplification procedures. Data analysis is poorly standardized, with incomplete sequence databases available to researchers (Elbrecht & Leese, 2015). Despite these shortcomings, recent approaches using metagenomics have found success with the chloroplast trnL intron for the determination of species identity from mixed environmental samples (*i.e.*, samples containing roots from several species), a significant advantage over other chloroplast genes which often require solitary root fragments (Lamb *et al*., 2016). While the use of the trnL region is known to result in relatively low taxonomic resolution, this is countered by the highly conserved primer regions (*i.e.*, these regions are unlikely to change over generational evolution), the relatively high number of quality sequences obtainable from reference databases, and ability to amplify highly degraded DNA sequences (Taberlet *et al*., 2007).

2.5 Previous Restoration Methods in Arctic Ecosystems

2.5.1 Soil Amendments

Previous arctic restoration efforts have used soil amendments such as fines, fertilizers, organic matter, and sewage to alleviate physical and chemical stressors limiting plant establishment or development (Naeth & Wilkinson, 2014). Commercial fertilizer amendments, most often N in combination with P and potassium (K^+)(Dormann & Woodin, 2002), have resulted in increases in overall plant growth rate, increases and decreases in the cover and abundance of mosses (Kelley & Epstein, 2009; Aradottir, 2012) and increases in graminoid species (Dormann & Woodin, 2002; Cater *et al.*, 2015). These fertilization treatments vary in terms of actual nutrient load, often ranging from 10-25 g m^{-2} of commercial product [*e.g.* 4 g N (Aradottir, 2012), 10 g N (Gough & Hobbie, 2003; Kelley & Epstein, 2009)], yet in some cases, extreme fertilization of 100 g m^{-2} (*e.g.* 20 g N) (Cater *et al.*, 2015). Graminoids are the functional group that responds most frequently and rapidly to nutrient additions within arctic and alpine environments (Dormann & Woodin, 2002; Gough & Hobbie, 2003; Kelley & Epstein, 2009; Aradottir & Oskarsdottir, 2013). For example, in moist acidic Alaskan tundra, Kelley and Epstein (2018) found that graminoid cover significantly increased over four years when exposed to N, in comparison to control and P amendments. Forbs significantly increased in cover when exposed to both N and P, whereas the opposite was found for evergreen shrubs, mosses, and lichens.

Some authors have found while benefits are present, the survivability rates of species did not increase with fertilizer addition (Cole & Spildie, 2006). Increases in plant productivity often appear to have a significant lag following amendment application. In a study conducted ~300 km north of Yellowknife, NWT, Naeth and Wilkinson (2014) found no noticeable effects of amendments for the initial two-three years following application and significant increases in plant abundance arising after five years. Similarly, Cole and Spildie (2006) reported the greatest plant growth during the second through the fourth growing season following the application of organic material in a subalpine forest.

2.5.2 Seeding

Seeding as a restoration practice in the arctic has been employed since at least the 1970s (Chapin & Chapin, 1980; Younkin & Martens, 1987; Jorgenson, Kidd, Carter, Bishop & Racine, 2003; Johnstone & Kokelj, 2009). Previous restoration work in the Arctic has focused on revegetation and testing the efficacy of seed mixes and fertilizer amendments (Younkin & Martens, 1987; Deshaies, Bondreau & Harper, 2009; Naeth & Wilkinson, 2014). Seeding initiatives in arctic climates have often shown contradictory results. Due to the lack of commercially available native seeds (Jorgenson & Joyce, 1994; Forbes & Jefferies, 1999), and the often slow growth of seeded arctic plants (Ficko & Naeth, 2021), fast-growing species capable of quick establishment and erosion control are traditionally used for revegetation efforts (Hagen *et al*., 2014; Rydgren *et al*., 2016). While these non-native species have shown promise in terms of quick revegetation, they have also shown the potential to reduce overall biodiversity, become invasive, and persist for several years after introduction (Forbes & Jefferies, 1999; Hagen *et al*., 2014; Kearns *et al*., 2015; Rydgren *et al*., 2016). In recent studies, Hagen *et al*. (2014) found that roadsides seeded and fertilized with commercial products resulted in higher vegetative cover when compared to unaltered roadsides; however, a lower presence of native vegetation and continued dominance of the seeded grass species (particularly *Festuca rubra*) occurred. Similarly, Barni, Freppaz & Siniscaleo (2007) found that introduced species, including *F. rubra*, still dominated vegetative communities on revegetated ski slopes. In contrast, seeding may facilitate colonization by native vegetation (Magnusson, 1992 as cited in Aradottir & Oskardottir, 2013). Johnstone and Kokelj (2008) found no evidence of invasive nor non-native species appearing on a sump cap intentionally seeded 30 years prior, theorizing that some non-native species may not be able to persist through several stages of arctic succession.

2.5.3 Individual Transplants

In contrast to initiatives involving seeding, there is a paucity of published works examining individual transplants within arctic environments. In alpine environments, the degree of success from planting live plugs of individual species appears to be strongly influenced by the species' root growth form (May, Weber & May, 1982). Species characterized by deep taproots, well-developed secondary roots, fleshy roots, and dense fibrous roots without rhizomes performed best during individual transplanting, while those having shallow and fibrous roots were less successful (May *et al*., 1982; Conlin & Ebersole, 2001; Cole & Spildie, 2006). Dwarf shrubs such as *B. glandulosa* have shown significant sensitivity to even minor degrees of disturbance (Naeth & Wilkinson, 2014) and did not regenerate following transplantation. Further challenges associated with individual transplants include the lack of infrastructure, such as greenhouses, required for the mass production of arctic species. Knowledge surrounding the propagation of arctic plants, by seeds or cuttings, is still limited, and while several species have shown rapid regeneration from cuttings (Ficko & Naeth, 2021), several others have shown limited propagation success within greenhouse settings (Hagen, 2002). Due to limited success with transplanting individual arctic species, specifically of later successional shrubs, attempts to transplant whole turfs (*i.e.*, intact assemblages of native vegetation and underlying soils) have been examined in arctic and alpine restoration initiatives.

2.6 Restoration with Locally Available Materials

2.6.1 Whole turf Transplants

Whole turf transplants also referred to as turfs, sods, or plugs are intact assemblages of vegetation, organic and mineral soils, and have been used in varying arctic and alpine environments, including wet-sedge meadows (Kidd *et al*., 2006, Cater *et al*., 2015), grasslands (Bay & Ebersole, 2006) and dry heaths (Aradottir & Oskarsdottir, 2013; Mehlhoop *et al*., 2018). Whole turf transplants can address problems noted with other restoration techniques, such as the use of soil amendments, non-native species, and the requirement to propagate native species. Importantly, whole turf transplants represent a form of applied nucleation and allow for the installation of functioning patches of reference vegetative communities. These patches of vegetation can range dramatically in size within published literature (*e.g.*, surface areas of 132 cm^2 to 2.6 m^2) (Shirazi, Haggerty, Hendricks & Reporter, 1998; Cater *et al.*, 2015). Specific benefits of utilizing whole turf transplants include using native species (Cater *et al*., 2015), a reduced impact of disturbance to individual plant species within the turf (Bay & Ebersole, 2006), and inclusion of soil resources including roots, organic matter, and soil organisms (Conlin & Ebersole, 2001; Klimeš, Jongepierová, Doležal & Klimešová, 2010). The use of whole turfs allows for the capture of seedbanks and provides safe spaces for seed germination (Mehlhoop *et al*., 2018), maintaining high diversity and presence of rare species post-transplantation (Aradottir, 2012; Aradottir & Oskarsdottir, 2013) and turfs have been shown to reduce overall subsidence, a factor directly related to turf survival rates (Cater *et al*., 2015). There are many benefits to transplanting whole turf, yet successful implementation of these restoration techniques requires several considerations.

Regardless of the plant community investigated, the use of turf transplants has consistently shown changes in community composition, most often a significant increase in graminoid cover (Shirazi *et al.*, 1998; Conlin & Ebersole, 2001; Kidd *et al*., 2006; Aradottir, 2012; Cole, 2013). This increase in graminoids is often regarded as a result of increasing N availability associated with harvesting disturbance (Aradottir & Oskarsdottir, 2013). Conversely, several studies have noted decreases in non-vascular cover post-transplantation (Shirazi *et al*., 1998; Kidd *et al*., 2006), although this is assumed to be a result of increased vascular production, effectively shading and out-competing lower-stature species for light. Shrub species, particularly evergreens, generally do not fare transplantation well, with Aradottir (2012) demonstrating turfs of ≤ 30 cm^2 ubiquitously declining in evergreen shrub cover. Several factors may explain this decline in evergreen cover, ranging from potential damage to root systems (Aradottir & Oskarsdottir, 2013), changes in nutrient availability (Zamin, Bret-Harte & Grogan, 2014; Gu & Grogan, 2020), or alterations to soil pH (Eskelinen *et al*., 2009). It must be recognized that harvesting turf transplants from undisturbed reference ecosystems represents a form of disturbance itself, requiring further investigation into the vegetative recovery of donor sites.

Despite the community modifications observed with whole turf transplants, turfs have shown promising results in the development of vegetative cover on disturbed sites, with overall survival rates ranging from 50-100% (Aradottir & Oskarsdottir, 2013; Cole, 2013), and have shown considerable rates of expansion over time. Mean turf area increases have been observed ranging from 55% in alpine heath (Aradottir, 2012), and 74% in Alaskan tundra (Shirazi *et al*., 1998) over two years, 77% in subalpine campsites over four years (Cole & Spildie, 2006) and roughly 10 to 50-fold increases over seven years in Alaskan wetlands (Kidd *et al*., 2006). If turf transplants can have similar rates of expansion within low-arctic upland heath tundra, they may represent an effective method of restoring large-scale disturbances, such as those attributed to resource extraction. The reasons for differing expansion and survival rates are still unknown, whether due to the vegetative communities used, local climates, or soil moisture, necessitating greater experimentation with this promising technique.

2.6.2 Shredded Tundra Material

The shredding of whole turfs has also been studied and adopted as a general practice for the recovery of bryophyte populations within peatland restoration (Rochefort, Quinty, Campeau, Johnson & Malterer, 2003), as many bryophytes are capable of regeneration from almost any physical structure including stems, leaves, and spores. The deposition of large quantities of soil and organic matter (known as sod dumping) has been used to transfer species from donor sites (Kiehl, Kirmer, Donath, Rasran & Hölzel, 2010). This technique has been mainly investigated within southern environments, such as European meadows, where 48 out of 64 species were successfully transferred from donor sites (Vécrin & Muller, 2003). Sod dumping was found to be more successful than both natural revegetation and seed sowing in the recreation of the original donor site (Vécrin & Muller, 2003). Similarly, the spreading of shredded turfs in Wales was found to be comparable in species recruitment and community composition to the use of whole turfs for the restoration of herb-rich grasslands (Good, Wallace, Stevens & Radford, 1999).

Limited studies involving sod dumping have been conducted in northern climates; however, Aradottir (2012) experimented with the shredding and spreading of tundra material, which demonstrated the promotion of moss-dominated communities, a cornerstone of low-arctic vegetative communities and an important early colonizer within BSC communities. Shrubs and rhizomatous species including *Carex* spp. did not demonstrate high rates of survival following shredding, however, several species were noted emerging from released seed banks (Aradottir, 2012). Like other restoration methods, sod dumping has specific challenges. Most notable is the sensitivity of the newly deposited materials to desiccation (Rochefort *et al*., 2003). Most studies involving sod dumping frequently use thick layers of sod (up to 50 cm) and doner:receptor ratios (*i.e.*, the ratio of total area harvested to total area treated) of 1:1, although greater donor:receptor ratios have demonstrated comparable results regarding species richness and similarity to donor sites (Buisson, Jaunatre, Römermann, Bulet & Dutoit, 2018). Greater donor:receptor ratios may be particularly beneficial, allowing disturbances from harvesting to restore larger areas and significantly less disturbance to cover the same area as whole turfs. Similar to turf transplants, shredding has resulted in significantly different vegetative communities when compared to reference systems. The establishment of early successional species like bryophytes from shredding is not inherently deleterious to the overall restoration effort, and long-term changes in composition have yet to be fully investigated.

3. TRANSPLANTED TUNDRA TURFS –RE-INTRODUCTION OF SPECIES AND COVER TO DISTURBED LOW-ARCTIC SITES[1]

3.1 Preface

The remote locations, environmental conditions, and lack of knowledge regarding natural recolonization and plant-soil processes within arctic environments create significant challenges for restoration of disturbed arctic sites. Although lacking in published data, transplantation of whole turfs has demonstrated successful transfer of native species cover and species diversity. This chapter investigates the maintenance of vegetative cover, community composition, and species richness within turfs and shredded tundra material two years following transplantation in low-arctic upland heath tundra. This chapter examines the maintenance of soil nutrient conditions and vegetative expansion, along with the vegetative recovery of turf harvesting locations

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[1] This chapter was submitted for review on April 25th, 2022 to *Ecological Restoration* as Hnatowich, I.G., Lamb, E.G., & Stewart, K.J. (2022). Transplanted tundra turfs - Re-introduction of species and cover to disturbed low-arctic sites. The manuscript is currently under revision, under the manuscript number: 042222-022. Dr Stewart and Dr. Lamb developed the experimental design, assisted with sample/data collection and data analysis, and provided expertise to address findings and editorial input. I assisted with experimental design, and sample/data collection. I completed the data analysis and completed the initial manuscript draft. Funding for the research was provided by NSERC's Collaborative Research and Development grant (CRDPJ 518281-17) in collaboration with Agnico Eagle Mines Ltd.

3.2 Abstract

Disturbed low-arctic environments provide many challenges for ecological restoration, from harsh climates and remote locations to limited knowledge on plant establishment and successional pathways within tundra ecosystems. Due to limited commercially available materials for the restoration of native low-arctic plant communities, transplantation may provide an effective technique for revegetation in these difficult-to-restore environments. In this study, whole turfs and shredded turfs were harvested from undisturbed upland heath tundra near Rankin Inlet, Canada, and transplanted onto nearby disturbed gravel quarries to investigate species survivability and development of upland heath vegetative communities. Two years following transplantation, turfs were found to maintain 85% of the initial vegetative cover and 91% of the initial species richness, with expansion up to 8 cm into the surrounding substrate, and production of seeds and spores. Shredded turfs could establish non-vascular plant cover over a larger area than intact turfs but required greater protection from environmental stressors. Vascular plants did not establish from shredded material. Our results demonstrate that whole turfs are resilient to harvesting and transplantation stresses, flooding, drought, and poor soil conditions, and are an effective means of species transfer promoting development of vegetative cover on disturbed substrates. High species survivability indicates that turfs may maintain species characteristic of the surrounding tundra, providing disturbed areas the species required for sustainable and self-organizing assemblages of native vegetation.

3.3 Introduction

Sensitive arctic environments are experiencing increasing disturbance from resource exploitation (Forbes *et al*., 2001) with mining activities in tundra environments known to result in severe disturbances, including removal of vegetative and organic layers, increased soil pH, and altered hydrology (Naeth & Wilkinson, 2014; Miller *et al*., 2021). Natural regeneration of vegetative communities within disturbed low-arctic environments can be a slow process. Decades or even hundreds of years are required to fully establish native vegetative communities (Forbes & Jefferies, 1999; Hodkinson *et al*., 2003), necessitating active restoration to recreate pre-disturbance or near pre-disturbance conditions. Low-arctic environments provide considerable challenges for restoration practitioners. Extreme abiotic conditions, such as low temperature and precipitation and short growing seasons, can impede key ecosystem functions such as decomposition, nutrient cycling, and vegetative production (Naeth & Wilkinson, 2014; Kearns *et al*., 2015; Mehlhoop *et al*., 2018).

The limited number of colonizing species (Forbes & Jefferies, 1999), and lack of commercially available tundra plant propagules often leads restoration practitioners to use non-native species and/or fertilizer amendments to quickly establish cover (Hagen *et al*., 2014). These practices may alter the natural establishment of indigenous species, leading to the promotion and persistence of non-native vegetation (Younkin & Martens, 1987; Barni *et al*., 2007; Hagen *et al*., 2014; Kearns *et al*., 2015; Rydgren *et al*., 2016). Effective arctic restoration is further challenged by limited knowledge on successional pathways, facilitative or inhibitive effects of common species, and the development of complex plant-fungal and/or plant-bacterial associations. Lastly, limited infrastructure necessitates the need for simple, cost-effective methods in these remote locations.

A common restoration method is whole turf or sod transplantation (*i.e.*, harvesting of intact vegetation and underlying soil layers and subsequent transplantation into disturbed sites). Turf transplants have been conducted in several arctic (Kidd *et al*., 2006; Cater *et al*., 2015) and alpine environments (Bay & Ebersole, 2006; Aradottir & Oskarsdottir, 2013; Mehlhoop *et al*., 2018), however, the efficacy of this technique has not yet been examined in low-arctic vegetative communities including upland heath tundra. Documented benefits of turf transplantation include: the use of native, functioning assemblages of vegetation (Cater *et al*., 2015), inclusion of soil resources such as organic matter, soil invertebrates, microbes, and root systems (Conlin & Ebersole, 2001; Klimeš *et al*., 2010), maintenance of species diversity (Aradottir, 2012; Aradottir & Oskarsdottir, 2013), and reduced shock to individuals within the transplants (Bay & Ebersole, 2006). Turfs also provide safe spaces for germination of seeds and spores (Mehlhoop *et al*., 2018), vegetative propagation within and adjacent to the turfs (Klimeš *et al*. 2010; Chapter 4.5.1, Chapter 4.5.2), and are simple to harvest and transplant. Similarly, there have been very few studies on the use of sod dumping, or the spreading of shredded vegetation and soils, within northern environments (Aradottir, 2012), a technique that is often used to restore non-vascular communities within peatland environments (Rochefort *et al*., 2003), and to transfer species within European meadows (Kiehl *et al*., 2010).

We investigated the use of both turf transplants and shredded tundra materials within two disturbed upland heath tundra sites. Our first objective was to determine the efficacy of these two restoration methods in maintaining vegetative cover (*i.e.*, the percentage of soil covered by vegetation), community composition (*i.e.*, the proportion of species and growth forms within the vegetation community), and species richness. We hypothesized that transplanted turfs would best maintain both vegetative cover and species richness. We hypothesized that the community composition of both turfs and shredded layers would change following transplantation, with turfs favoring the development of graminoids, and shredded layers promoting non-vascular colonization. Our second objective was to examine soil nutrient conditions in transplanted turfs. We hypothesized that nutrient concentrations would not change significantly two years post-transplantation. Our third objective was to investigate the recovery of vegetative cover within turf harvesting locations. We hypothesized that compared with other tundra vegetation, moss species would have the greatest vegetative cover within the harvesting plots two years post-harvest.

3.4 Methods

3.4.1 Study Location

Our study was located at the Agnico Eagle Mines Ltd. Meliadine site (63°01'22.9"N 92°11'41.1"W DMS) near Rankin Inlet, Nunavut, Canada (Fig. A.1). Situated in the Southern Arctic Ecozone, the area is characterized by cold winters (30-year climate normal, seasonal average temperature -24°C, Nov-Apr) and short, moderate summers (seasonal average temperature 7°C, June-Sept) with ~ 85 growing season days (Ecological Stratification Working Group 1995; Environment Canada 2021). Precipitation is low with an annual mean of 310 mm, roughly half falling during the summer months. Mean wind speeds are 23 km h^{-1}. During the two years of study, the site experienced both high summer rainfall in 2019, totaling 332 mm through June-Sept., and a dry early summer in 2020, with only 12.4 mm falling during June and July (Environment Canada 2021).

Soils are predominately Turbic Cryosols with Regosolic features, Turbic Cryosols, and Organics, with cryoturbation resulting in well-developed hummock-hollow complexes across the study site (Golder Associates, 2014). Mineral soils in the local area are composed of well-drained sand, silt, and gravel tills usually overlain by thin (10-12 cm) organic layers. Upland heath tundra, heath-lichen and lichen-rock communities associated with well-drained soils represent over half of the vegetative communities within the local area. Common species include the shrubs *Cassiope tetragona* (Arctic Mountain Heather), *Dryas integrifolia* (White Mountain-avens), *Salix reticulata* (Net-leaved Willow), and *Vaccinium uliginosum* (Bog Blueberry), forbs *Oxytropis maydelliana* (Maydell's Oxytrope), *Cardamine digitata* (Richardson's Bittercress), and *Stellaria longipes* (Long-stalked Starwort), the mosses *Aulacomnium turgidum* (Swollen Thread Moss), *Pohlia nutans* (Nodding Thread Moss) and *Hylocomium splendens* (Stairstep Moss), and lichens *Dactylina arctica* (Finger Lichen), *Thamnolia vermicularis* (White Worm Lichen), *Cetraria* spp., and *Cladonia* spp.

3.4.2 Experimental Design

Two gravel quarries (Q1 and Q2) were selected for the restoration sites (Fig. A.1). The quarries were selected following the identification of areas not expected to have further anthropogenic disturbance, areas not influenced from nearby roads (*i.e.*, sufficient distance from road dust), and flat, open areas with similar gravel substrates (Table A.1). Upland heath donor sites adjacent to the quarries were selected based upon similar community composition of both vascular and non-vascular vegetation, ease of harvesting and transport, and depth of both organic (~10 cm) and mineral (~5 cm) soil layers to ensure both soil layers were harvested.

At each restoration site, a 15 m^2 area was delineated and a backhoe-loader was used to excavate quarry substrates and deposit material into four, 15 m long by ~50 cm high rows, spaced ~1.5 m apart (Fig. A.2). The rows were re-contoured by hand to simulate the hummock-hollow microtopography characteristic of the surrounding tundra landscape. Hummocks were ~50 cm high, with the hummock ridges spaced one metre apart. Each row contained 10 treatment plots (0.5 by 1 m) separated by 0.75 m (Fig. A.2). Four treatments were applied within the treatment plots: a single 40 × 40 cm (approximately 10-15 cm thick) turf placed in the hollow (T), shredded turfs which were spread over the entire plot area (S), a combination of turfs and shredded material (TS), and a control plot with no material added (CT) (Table A.2). Treatments were placed in a randomized complete block design.

Harvesting of turfs and shredded material was conducted in upland heath communities located near each restoration site (119 m and 60 m distance for Q1 and Q2, respectively). A flat-head shovel was used to cut, lift and remove the turfs from the harvesting sites. The underlying material was predominantly organic matter; however, an effort was taken to ensure that harvested material also contained some of the underlying mineral layers (Table A.1). Harvesting plots were flagged, spatially referenced, and photographed. At all harvesting plots, the depth of each directional face was recorded for monitoring future vegetative encroachment or ground subsidence.

All turfs (n=40) were placed with the vegetative surface level flush with the surrounding substrate surface. Once placed, the surrounding substrate was pushed by hand against the turfs to ensure good turf-substrate contact. Three additional turfs were harvested for shredding. Shredded material was sieved through a 4 cm^2 metal mesh screen to homogenize before application. All treatments were covered by a jute-mesh erosion control blanket (Anti-wash GEOJUTE®, Belton Industries, Honea Path, SC), chosen due to high light penetration (45% - 60%), ability to protect from erosive forces, water storage capacity (425%) and degradation time of ~1-2 years, depending on climate.

3.4.3 Vegetation and Substrate Analysis

Immediately following transplanting (July 7th -12th, 2019), a 0.16 m^2 gridded quadrat (25, 8 cm × 8 cm sub-quadrats) was placed over the top of each T, TS, and S treatment. Values ranging from 1 to 4 (1 = <25%, 2 = 25-50%, 3 = 50-75%, 4 = >75%) were used to denote the cover of individual species in each individual grid. All vascular and non-vascular plants within harvested turfs were identified to species level whenever possible. Cover of bare ground (including rocks and stones), woody litter (*i.e.*, woody branches, stems, etc.), plant litter (*i.e.*, dead plant tissues, leaves, flowers, etc.), and presence or absence of flowers, per species, were also recorded within each individual grid. For T treatments only the center plot was surveyed, for TS and S treatments the center plot and the inside of both sides of the hummock were surveyed in the same fashion. A rapid visual survey was completed to ensure no vegetative cover in the CT plots. All treatments were resurveyed between July 7th -13th, 2021 in a similar manner, except for T and CT treatments where the inside of both sides of the hummocks were included. Aboveground expansion was assessed in each treatment by surveying an additional 24 cm on the non-treated sides of the central plot, to avoid inclusions of vegetation within shredded layers of S and TS treatments. Turf harvesting locations were surveyed between July 15th-17th, 2021 using the same method, except the 0.16 m^2 gridded quadrat had four 20 by 20 cm sub-quadrats. Several individuals were responsible for recording cover within treatment plots, however, the statistically similar communities within treatments suggests different individuals did not affect the recorded community compositions.

3.4.4 Soil Sampling and Analysis

In July 2019 soil samples were collected within 1 metre of ten randomly chosen turf harvesting locations at each restoration site. Both organic and mineral horizons were sampled (Table A.1). A composite sample of the substrate used to create the hummock-hollows at Q1 and Q2 was also sampled. The same harvesting locations were sampled between July 15th and 17th, 2021. On July 16th and 17th 2021 composite soil core samples were taken (n= 5 cored samples per turf, PN009 dry sampling tube, JMC, Iowa, USA) from all treatments that included a turf (n=40). Organic and mineral layers were sampled simultaneously, then separated into their respective layers, with the soil corer wiped clean by hand between each sample. All samples were transported to the University of Saskatchewan where they were air-dried at room temperature for roughly seven days, sieved (4 mm^2), and stored at -20°C for future analysis. Water extractions were conducted on all soil samples using a 1:4 ratio of soil to Milli-Q water, except for 8 (2019) and 23 (2021) samples that required higher water to soil ratios (*i.e.* 1:6 or 1:8) due to high amounts of organic matter. Extracts were then measured on a Dionex ICS-2000 Ion Chromatograph to determine concentrations of cations (Ca^{2+}, Mg^{2+}, K^+, Na^+, NH_4^+) and anions (F^-, NO_3^-, NO_2^-, SO_4^-, Cl^-, PO_4^{3-}). Soil pH was measured using a Mettler Toledo FiveEasy pH meter. Subsamples of each soil were ground in preparation for total carbon (TC), total inorganic carbon (TIC), and total nitrogen (TN). Total nitrogen was analyzed using a LECO TruMac CNS Analyzer, total inorganic nitrogen (TIN) was calculated from concentrations of NO_3^-, NO_2^-, and NH_4^+, and total organic nitrogen (TON) was calculated by the difference between TN and TIN. TIC was analyzed by an acetic acid pH standard curve (Goh & Mermut, 2008), TC was analyzed with a modified high-temperature combustion method (Skjemstad & Baldock, 2008), and total organic carbon (TOC) was calculated by the difference between TC and TIC. TC and TIC were analyzed by ALS Environmental (Saskatoon, Saskatchewan, Canada).

3.4.5 Statistical Analysis

Statistical analysis was conducted in R 4.1.2 (R Core Team, 2020). Vegetative percent cover data used in multivariate analyses were first converted into a Bray-Curtis distance matrix. Community composition of all treatments during 2021, and community composition of turfs (turfs of T and TS treatments) in 2019 and 2021 were visualized via non-metric dimensional scaling (NMDS) ordination plots using the ‘nmds’ function within the ‘vegan’ package (Oksanen *et al*., 2020). Differences in community composition between treatments in 2021 (with block nested within site), and between turf community composition in 2019 and 2021 (with each turf given a unique identifier “subject ID” to account for repeated measures) were tested using Permutational multivariate analysis of variation (PerMANOVA) using the ‘adonis’ function in vegan. Post-hoc comparisons between treatments were determined using ‘TukeyHSD’ in the ‘stats’ package (R Core Team, 2020), on the centroid location of each treatment.

The effects of treatment over the entire treatment plot (hummocks and hollow) on the response variables: species richness, cover of bare ground, litter, total vegetation, growth forms (deciduous and evergreen shrubs, forbs, graminoids, biological soil crusts, lichens, and mosses), and presence of flowers (*i.e.*, number of quadrat grids containing flowers, per species) in 2021 were assessed through linear mixed effect (LME) models [“lmer” functions within the “lmerTest” package (Kuznetsova, Brockhoff & Christensen, 2017)] and one-way analysis of variance (ANOVA) (“anova” within the “stats” package), using block as a random factor (each block at each site was given a unique identifier). All data were visually inspected using QQ-plots and residuals vs. fitted plots to ensure assumptions of ANOVA tests were met. When data did not meet assumptions, data were transformed (detailed transformation information can be found in Tables A.4, A.6, A.8, A.11, A.13, A.14, A.15). Changes in the above response variables were also examined within shredded materials (S and shredded layers of TS treatments) between 2019 and 2021 and within transplanted turfs (turfs of T and TS treatments) between 2019 and 2021 using the same approach. Differences between years, sites, and their interaction were tested with LME models and two-way ANOVA (with subject ID as a random factor). Expansion of vascular and non-vascular cover between treatments and distances (0-8, 8-16, 16-24 cm) from the treatment plot were assessed with LME models and two-way ANOVA tests, using block as a random factor. Expansion from turf treatments (bare ground, litter, vascular, and non-vascular cover) between distances (0-8, 8-16, 16-24 cm) were also assessed with linear models and one-way ANOVA.

Mineral and organic soil layer pH, cations, anions, TON, TIN, TOC, and TIC were compared between transplanted turfs and their harvesting locations in 2019 and 2021 for each site separately using linear models and one-way ANOVA tests. All data were inspected to ensure assumptions of ANOVA tests were met. When assumptions were not met, data were transformed (detailed transformation info can be found in Table A.18). Post hoc comparisons using the ‘lsmeans’ function within the ‘lsmeans’ package (Lenth, 2016) were used to determine statistical differences between groups in all linear models and ANOVA tests.

3.5 Results

3.5.1 Comparison of Treatment Plant Communities

Two years following turf transplantation, all four treatments had significant differences in community composition (df = 79, R^2 = 0.43, p = 0.001) (Fig. 3.1) (Table A.3). The community composition of all transplanted treatments was significantly different from the control (p = <0.001), with S treatments also significantly different from T and TS treatments (p = 0.009). T and TS treatments did not differ (p = 0.991). T and TS treatments contained the highest species richness, greatest vegetative cover, and greatest presence of flowers, whereas S treatments contained ~1/2 of the species richness, ~1/4 of the vegetative cover, and ~1/10 the flowers of T and TS treatments (Table 3.1, A.4- A.6). All growth forms, except for biological soil crust and lichen cover, were significantly higher in T and TS treatments compared to S or CT treatments (Table 3.1, A.4).

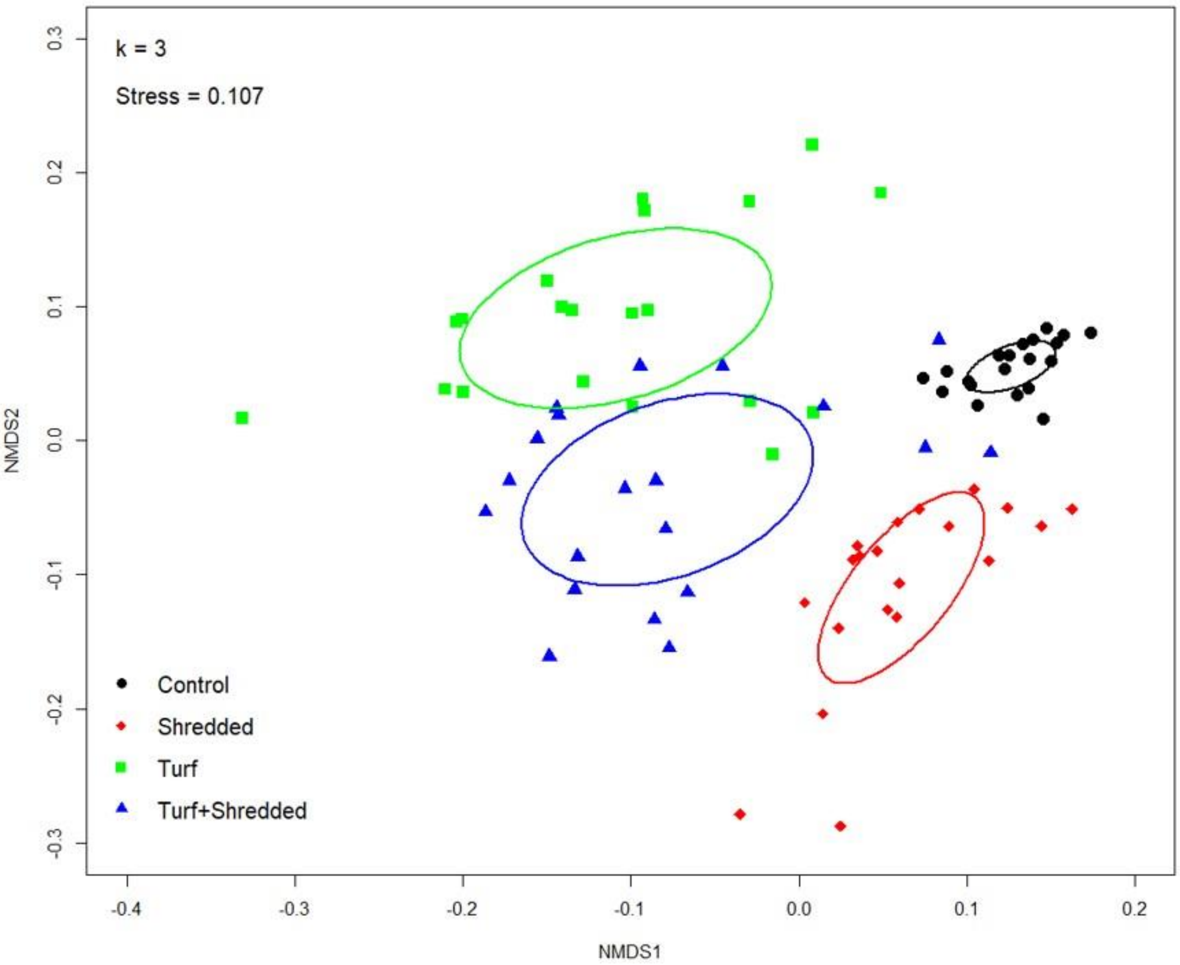

Figure 3.1. Non-metric dimensional scaling (NMDS) ordination plots of treatment community composition two years after turf transplantation. Treatments are represented by black circles (Control), green squares (Turf), blue triangles (Turfs + Shredded), and red diamonds (Shredded). Ellipses represent the standard error of the weighted average of scores.

Table 3.1. Means, standard deviations, and significant difference indicators for all treatments on species richness, and cover of bare ground features, total vegetation, litter, and growth forms. Different letters indicate significant differences between treatments for each variable determined through post hoc comparisons of linear mixed effect models using block as a random factor.

	Treatments			
Cover/Variable	**Turfs**	**Turfs + Shredded**	**Shredded**	**Control**
Species Richness	20.2 ± 6.1 ^{a}	23.2 ± 7.1 ^{a}	10.4 ± 6.1 ^{b}	3.7 ± 3.2 ^{c}
Bare ground	289 ± 26.6 ^{c}	320 ± 52.2 ^{bc}	349 ± 68.6 ^{ab}	366 ± 18.8 ^{a}
Total Vegetation	195 ± 75.4 ^{a}	210 ± 91.8 ^{a}	56.5 ± 42.0 ^{b}	12.1 ± 13.4 ^{c}
Litter	74.9 ± 28.7 ^{c}	163 ± 33.4 ^{b}	210 ± 51.9 ^{a}	46.8 ± 29.1 ^{d}
Deciduous Shrubs	41.4 ± 22.4 ^{a}	39.6 ± 26.5 ^{a}	0 ± 0 ^{b}	0 ± 0 ^{b}
Evergreen Shrubs	37.8 ± 21.3 ^{a}	35.7 ± 26.4 ^{a}	0.6 ± 1.3 ^{b}	0.1 ± 0.3 ^{b}
Forbs	21.4 ± 17.4 ^{a}	24.4 ± 17.9 ^{a}	9.7 ± 13.2 ^{b}	4.4 ± 5.3 ^{b}
Graminoids	23.8 ± 18.1 ^{a}	20.0 ± 15.6 ^{a}	2.6 ± 3.3 ^{b}	3.3 ± 6.5 ^{b}
Biological Soil Crusts	0.3 ± 0.6 ^{a}	0.1 ± 0.3 ^{a}	0 ± 0 ^{a}	0.1 ± 0.2 ^{a}
Lichens	51.1 ± 38.4 ^{ab}	69.4 ± 37.9 ^{a}	33.6 ± 26.6 ^{b}	2.3 ± 3.6 ^{c}
Mosses	16.2 ± 13.6 ^{a}	18.2 ± 16.7 ^{a}	6.1 ± 7.6 ^{b}	0.9 ± 1.8 ^{c}

3.5.2 Post-transplant Changes in Community Composition of Shredded Materials

Shredded layers (*i.e.*, S and shredded material from TS treatments) significantly decreased in species richness post-transplantation ($F_{(1,39)}$ = 9.91, p = 0.003) (Table A.7- A.8). Cover of total vegetation ($F_{(1, 39)}$ = 72.68, p = <0.001), bryophytes ($F_{(1, 39)}$ = 57.58, p = <0.001), and lichens ($F_{(1, 39)}$ = 70.92, p = <0.001), decreased by at least 50%, while evergreen ($F_{(1, 39)}$ = 167.69, p = <0.001) and deciduous shrub cover decreased at least 80% ($F_{(1, 39)}$ = 86.29, p = <0.001) (Table A.7- A.8). Both total litter ($F_{(1, 39)}$ = 226.96, p = <0.001) and bare ground ($F_{(1, 39)}$ = 167.1, p = <0.001) increased in cover over the two years. The only growth form to increase were forbs ($F_{(1, 39)}$ = 64.60, p = <0.001), which demonstrated a 18-fold increase in cover, likely due to the colonization of volunteer species (*e.g.*, *Descurainia sophioides* (Northern Tansy Mustard), *S. longipes* and several unidentified juvenile forbs). No evidence of biological soil crusts development was observed within shredded layers.

3.5.3 Post-transplant Changes in Community Composition of Turfs

Turfs (*i.e.*, T and central plot of TS treatments) maintained their species richness over two years (Table A.10- A.11). Although overall richness remained unchanged, plant community composition within the turfs changed significantly between 2019 and 2021 (df = 79, R^2 = 0.16, p = 0.001) (Fig. 3.2) (Table A.9). Significant differences were the result of increased cover of bare ground ($F_{(1, 39)}$ = 27.87, p = <0.001), litter ($F_{(1, 39)}$ = 248.6, p = <0.001), and forbs ($F_{1,39)}$ = 14.90, p = <0.001) along with decreases in total vegetation ($F_{(1,39)}$ = 13.19, p = <0.001) evergreen shrub ($F_{(1,39)}$ = 31.04, p = <0.001), lichen ($F_{(1, 39)}$ = 32.96, p = <0.001), and moss cover ($F_{(1,39)}$ = 7.04, p = 0.012) (Table A.10- A.11). Turfs between the two sites were similar in composition, except that Q1 had higher lichen cover in both 2019 and 2021 ($F_{(1.38)}$ = 11.24, p = 0.002), whereas Q2 had higher cover of deciduous shrubs in both years ($F_{(1,38)}$ = 22.78, p = <0.001) and higher cover of mosses, although only significantly in 2021 (F($_{1,38}$) = 11.13, p = 0.002) (Table A.12- A.13).

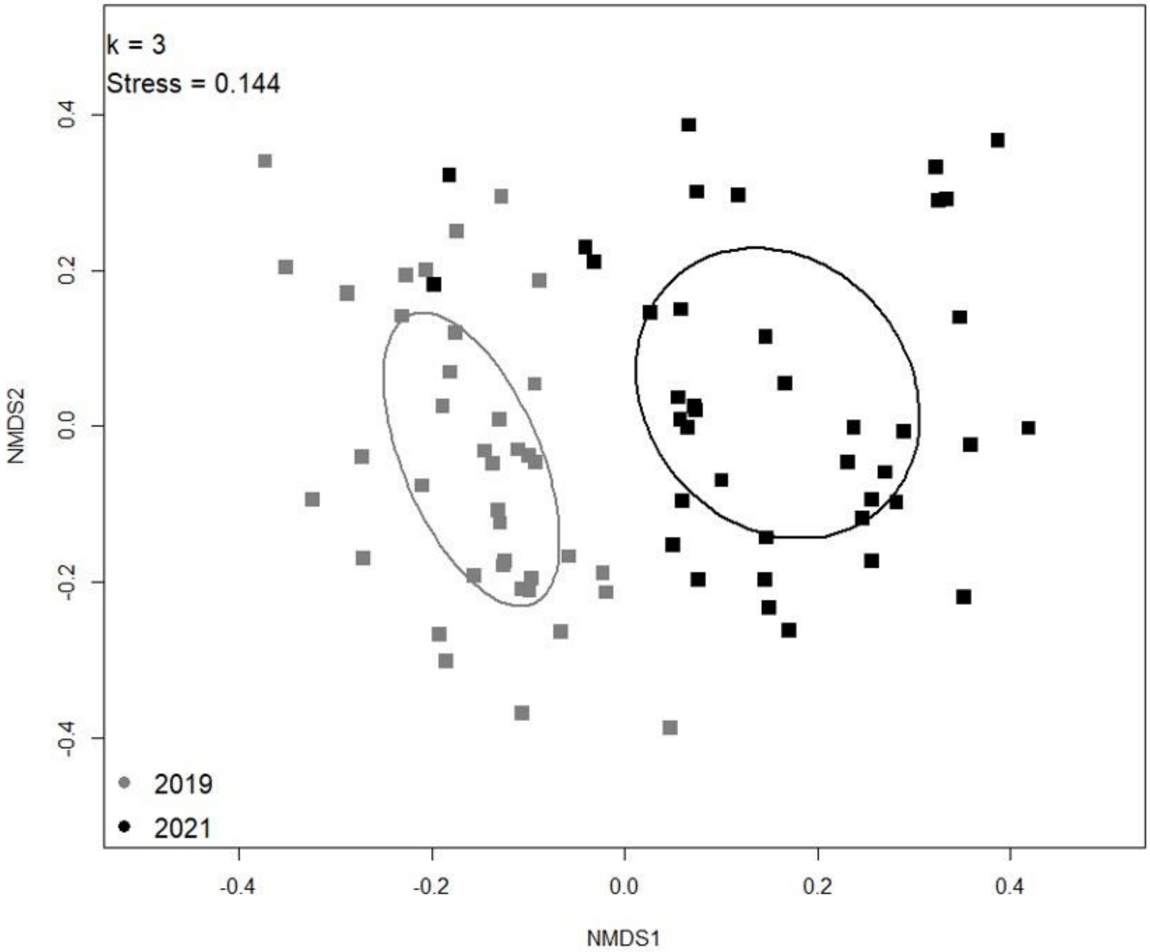

Figure 3.2. Non-metric Dimensional Scaling (NMDS) ordination plot of community composition of turfs over two years. Squares represent turfs from T treatments and the central turf from TS treatments. Years are represented by grey (2019) or black (2021) shading. Ellipses represent the standard error of the weighted average of scores.

Evergreen shrubs demonstrated the largest decrease in absolute cover over two years (-31%), and while all species declined in cover, *C. tetragona*, *Rhododendron lapponicum* (Lapland Rosebay), and *Rhododendron tomentosum* (Northern Labrador Tea) declined more than other species such as *D. integrifolia* and *Vaccinium vitis-idaea* (Lingonberry*)* (Table 3.2). Lichens had the second-largest decline in absolute cover (-30%), with all species including *T. vermicularis*, *Alectoria ochroleuc*a (Green Witch's Hair Lichen*)*, *D. arctica*, *Cetraria* spp., and *Cladonia* sp. declining by over 25% in absolute cover. Some moss species declined in cover (*e.g.*, *H. splendens* and *P. nutans*), while others increased [*A. turgidum* and *Bucklandiella microcarpa* (Small Fruited Rock Moss)]. Forbs and graminoids were the only growth forms to increase in cover (+83% and +63%, respectively), notably from increases of *Hedysarum alpinum* (Alpine Sweetvetch), *Bistorta vivipara* (Alpine Bistort), *C. digitata*, *O. maydelliana*, and *Carex bigelowii* (Bigelow's Sedge) (Table 3.2).

Table 3.2. Percent change of growth forms and species that represented at least 0.5% relative cover across both sites, either in 2019 or 2021, within the center plot of T and TS treatments. The percent difference of absolute cover is presented first, followed by relative cover in parenthesis. Absolute and relative cover are calculated as follows: absolute percent differences are ((species cover in 2021- species cover in 2019)/ species cover in 2019), and relative percent differences are ((species cover in 2021/total species cover in 2021) – (species cover in 2019/total species cover in 2019)). Asterisks denote significant differences between 2019 and 2021. Means and standard deviations of each growth form can be found in Table A.10.

Growth Form	**Percent Difference Absolute (Relative)**	**Species**	**Percent difference Absolute (Relative)**
Bare Ground*	+83% (+4%)	Bare Soil	+86% (+3%)
		Rocks and Stones	+110% (+1%)
Litter*	+321% (+13%)	Loose Organic Matter	+58% (+0.2%)
		Plant Litter	+75% (+2%)
		Woody Litter	+10464% (+11%)
Deciduous Shrubs	-1% (-2%)	*Arctous rubra*	-13% (-1%)
		Salix arctica	-16% (-0.4%)
		Salix reticulata	+25% (0.2%)
		Vaccinium uligonosum	-1% (-1%)
Evergreen Shrubs*	-31% (-8%)	*Cassiope tetragona*	-54% (-4 %)
		Dryas integrifolia	-0.4% (-1%)
		Empetrum nigrum	-12% (-0.3%)
		Rhododendron lapponicum	-61% (-2%)
		Rhododendron tomentosum	-43% (-1%)
		Vaccinium vitis-idaea	-8% (-0.3%)
Forbs*	+83% (+2%)	*Bistorta vivipara*	+208% (+1%)
		Cardamine digitata	+455% (+1%)
		Hedysarum alpinum	+58 % (+0.2%)
		Oxytropis arctica	-71% (-1%)
		Oxytropis maydelliana	+263% (+0.5%)
Graminoids*	+63% (+2%)	*Carex bigelowii*	+133% (+1%)
		Carex rupestris	+18% (+0.03%)
Lichens*	-30% (-12%)	*Alectoria ochroleuca*	-33% (-3%)

		Cetraria nivalis	-38% (-4%)
		Cetraria ericetorum	-34% (-0.2%)
		Cladonia sp.	-75% (-0.5%)
		Dactylina arctica	-26% (-0.4%)
		Gowardia nigricans	-9% (-1%)
		Thamnolia vermicularis	-43% (-3%)
		Unidentified crustose lichen	-16% (-0.2%)
Mosses*	-19% (-2%)	*Aulacomnium turgidum*	+23% (+0.1%)
		Hylocomium splendens	-66% (-1%)
		Pohlia nutans	-22% (-1%)
		Polytrichastum pallidisetum	-91% (-0.5%)
		Bucklandiella microcarpa	+35% (+0.1%)
		Rhytidium rugosum	-31% (-0.4%)
		Sanionia uncinata	-10% (-0.1%)

3.5.4 Post-transplant Vegetative Expansion

T and TS treatments had a significant expansion of vascular plants ($F_{(3,209)} = 20.79$, p = <0.001) within 8 cm of the turf, whereas little expansion was observed for S or CT treatments. Similarly, T and TS treatments demonstrated significantly higher non-vascular cover than CT treatments ($F_{(3,209)} = 10.60$, p = <0.001) with the cover of both metrics significantly decreasing after 8 cm (F = 32.26, p = <0.001; F = 8.41, p = <0.001) (Table A.14- A.15)(Fig. 3.3). Across both T and TS treatments, vascular and non-vascular species represented ~17% and ~6% of total cover within the first 8 cm, respectively, quickly dropping to 4% and 3% after 8 cm, with higher standard deviations indicating highly variable expansion, *i.e.*, small pockets of vegetation, or individual plants emerging near the boundaries of the turf (Fig. 3.3).

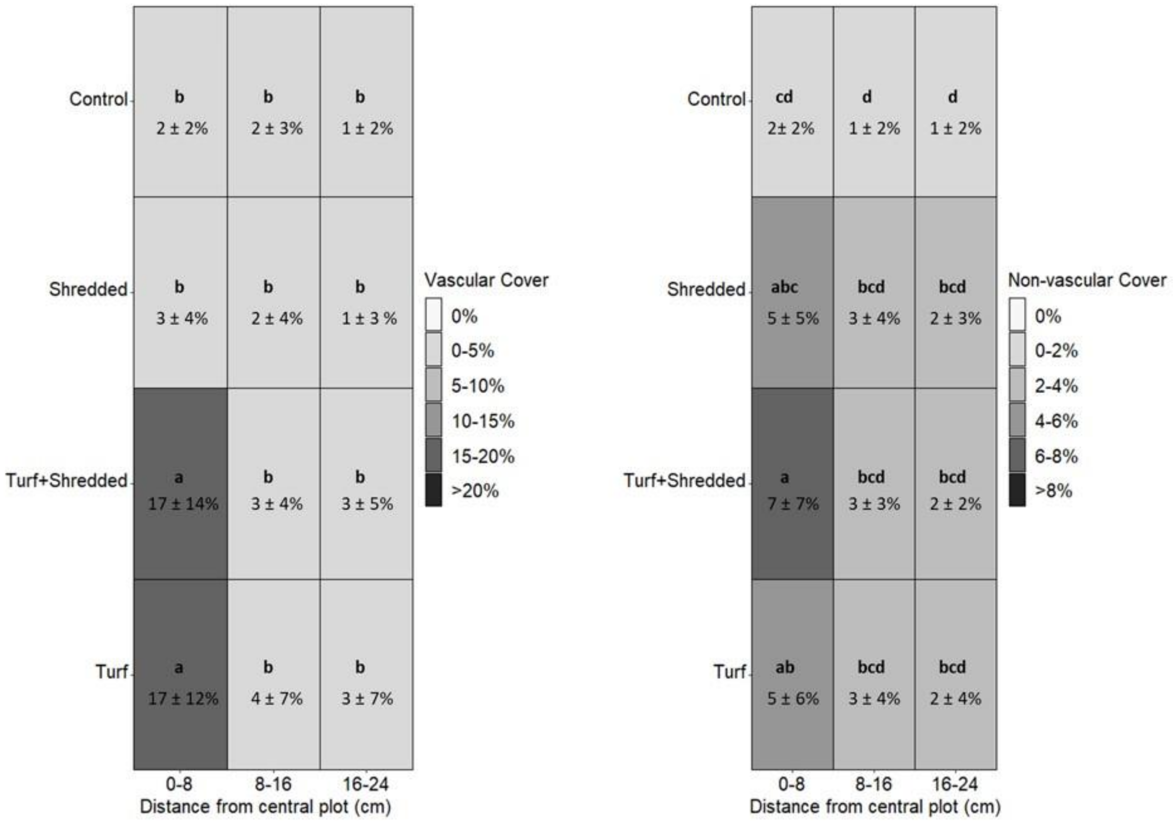

Figure 3.3. Heatmaps of mean relative vascular and non-vascular cover expanding from the central plot of each treatment, grouped by distance from the central plot. Different letters within boxes represent significant differences between treatments and between distances, with mean and standard deviations presented below.

Within 8 cm of the turf, the relative contribution of deciduous and evergreen shrubs, forbs, and graminoids was nearly equal, with each growth form contributing 27%, 27%, 26%, and 20% to vascular cover. *Vaccinium uligonosum* and *Empetrum nigrum* (Crowberry), made the highest contributions of 18% and 10% of the relative vascular cover immediately adjacent to the turfs, with all other species representing less than 7% (Fig. 3.4). Beyond 8 cm, forbs and graminoids dominated the expanding vascular cover, with 46% and 40% relative cover, respectively. *S. longipes*, juvenile unidentified forbs, *C. bigelowii*, and *Poa arctica* (Arctic Bluegrass) contributed the most to the cover of vascular species beyond 8 cm (Fig. 3.4). Deciduous shrubs represented 11% of the vascular cover beyond 8 cm, with evergreen shrubs only 3%. A similar pattern was observed with non-vascular cover, with lichens dominating the cover within the first 8 cm, representing 63% of non-vascular cover, and mosses representing 36%. *Alectoria ochroleuca*, *Cetraria nivalis* (Crinkled Snow Lichen), *Gowardia nigricans* (Grey

Witch's Hair Lichen), *T. vermicularis,* and unidentified crustose lichen growth represented 13%, 20%, 10%, 13%, and 11% of non-vascular cover immediately adjacent to the turfs, respectively (Fig. 3.4). Beyond 8 cm, the relative contribution to non-vascular cover switched with mosses representing 63%, and lichens 36%. Further from the turfs, juvenile mosses comprised nearly half of all non-vascular cover, although *T. vermicularis* and crustose lichen growths still contributed >10% each.

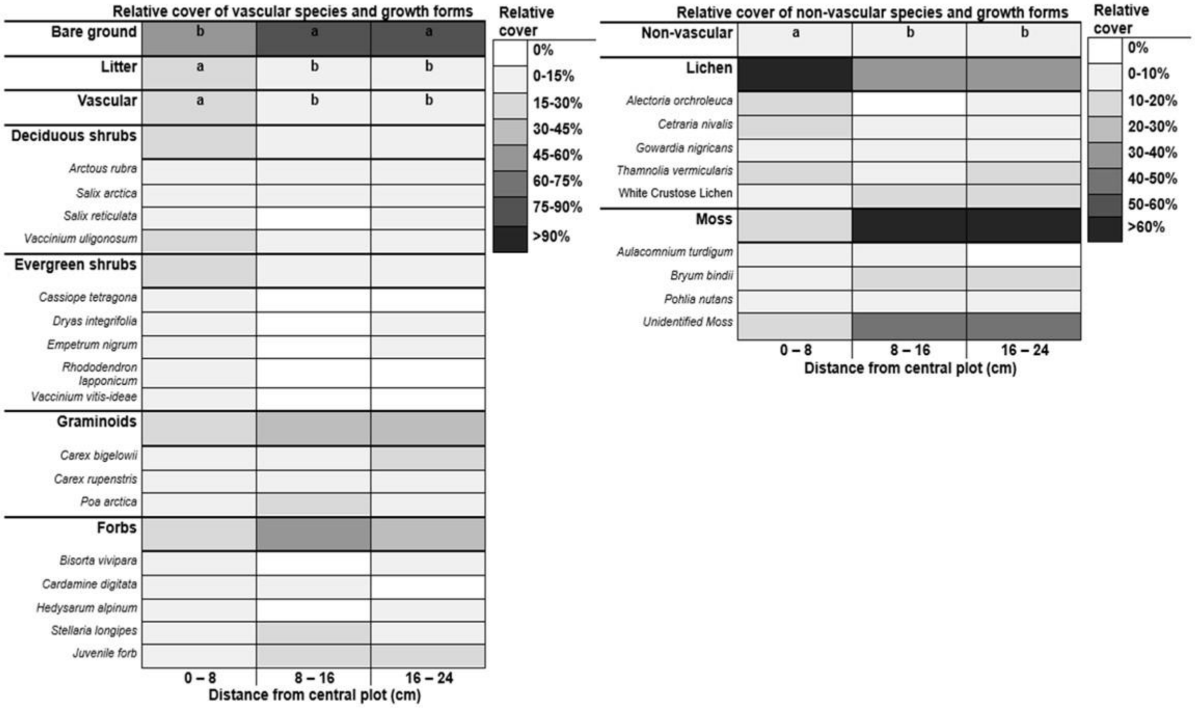

Figure 3.4. Heatmaps of vascular and non-vascular species expanding from the turfs that were observed in at least 10% of the T and TS treatment plots at either Quarry #1 or #2. Darker shades represent a greater relative contribution to the expanding communities. Bare ground, litter, vascular and non-vascular cover represent cover relative to all species, while deciduous shrubs, evergreen shrubs, graminoids, and forb growth forms represent cover relative to vascular cover, and lichen and moss growth forms represent cover relative to non-vascular cover. Different letters within the bare ground, litter, vascular, and non-vascular cover columns represent significant differences between distances.

3.5.5 Recovery of Turf Harvest Sites

After two years, vegetative establishment within harvest plots was minimal (i.e., very little vegetation cover, with harvest plots predominately composed of bare ground). The vegetation cover present was dominated by non-vascular lichens and mosses that had likely blown into the depressions left from harvest, accounting for 29% and 11% of the relative vegetation cover respectively (Table A.16). Vascular plant growth originated primarily from the sidewalls of the harvest plot (*i.e.*, new growth from cut and exposed belowground structures) and was composed of evergreen shrubs, deciduous shrubs, forbs, and graminoids at 14%, 12%, 9%, and 6% relative cover respectively (Table A.16). There was evidence of slumping from the sides of some harvest plots, with the average depth of harvest plots decreasing by roughly one centimetre over the two years since harvest (*i.e.*, an average of 10.29 cm to 9.13 cm).

3.5.6 Post-transplant Changes in Soil Nutrients

Two years following transplantation, the turfs within T and TS center plots demonstrated significant differences in nutrient concentrations from their respective harvesting sites, in both organic and mineral layers. Within organic layers, both sites demonstrated significant increases, approximately 15-fold (Q1) and 3-fold (Q2), in concentration of Cl^- (F = 17.80, p = <0.001; F = 11.21, p = <0.001) and approximately 19-fold (Q1) and 5-fold (Q2) in Na^+ (F = 16.59, p = <0.001; F = 24.96, p = <0.001). Q1 demonstrated significantly decreased TON (F = 6.57, p = <0.01), whereas Q2 demonstrated 9-fold increase in SO_4^{2+} (F = 82.46, p = <0.001), roughly twice the concentration of Ca^{2+} (F = 10.36, p = <0.001), and significant decreases in NO_3^- (F = 22.19, p = <0.001) (Table A.17).

Differences between transplanted turf and harvest sites within mineral soils were far more frequent than organic layers, with both sites demonstrating significant increases in pH (F = 48.75, p = <0.001; F = 114.09, p = <0.001), Cl^- (F = 74.52, *p* = <0.001; F = 20.29, p = <0.001), Na^+ (F = 40.77, p = <0.001; F = 73.31, p = <0.001), Ca^{2+} (F = 22.95, p = <0.001; F = 9.86, p = <0.001), SO_4^{2+} (F = 40.32, p = <0.001; F = 40.32, p = <0.001) and TIC (F = 23.92, p = <0.001; F = 23.47, p = <0.001) (Table A.17). Mineral soils of turfs at Q1 also increased in NO_3^- (F = 31.64, p = <0.001), K^+ (F = 13.62, p = <0.001), Mg^{2+} (F = 17.93, p = <0.001) and TIN (F = 21.49, p = <0.001) compared with harvest sites. Substrates used to create the hummock-hollow complexes had higher pH and greater concentrations of Cl^-, SO_4^{2+}, NO_3^-, Ca^{2+}, and Na^+ relative to harvest sites mineral soils (Table A.17). Mineral soils at both sites demonstrated 2-fold and 10-fold increases in Ca^{2+}, and SO_4^{2+}, respectively. Q1 mineral soils demonstrated 24-fold and 18-fold increases in Cl^- and Na^+, respectively, compared to a 3-fold and 4-fold increase at Q2.

3.6 Discussion

We examined the revegetation of disturbed low-arctic sites through transplantation of upland heath whole turfs and shredded layers and found that whole turf transplants were most effective at maintaining species diversity, community composition, and vegetative cover of all vascular growth forms. Shredded treatments performed poorly, and while some development of non-vascular communities was evident, greater protection from external forces such as wind, water, and desiccation is likely required for spreading shredded material to be effective. Changes in community composition occurred for both turfs and shredded treatments and our hypothesis of increasing cover of graminoids in turfs and non-vascular species in shredded layers was supported. In contradiction to our hypothesis, nearly all metrics regarding available nutrients changed significantly within transplanted turfs, however, these changes were primarily driven by an influx of soluble salts from quarry substrates. We hypothesized that mosses would make the greatest contribution to vegetative cover in the early stages of turf harvest plot recovery, however, instead, lichens provided the most cover. Regardless, recovery of the turf harvesting plots were minimal two years post-transplantation, suggesting that the use of turf transplants should be pre-emptively planned to reduce additional disturbance when employing this technique for restoration of low-arctic sites.

3.6.1 Intact vs. Shredded Turfs Require Different Management

Turf transplants performed well, maintaining high degrees of vegetative cover (85% of the average absolute vegetative cover recorded in 2019) and species richness (91% of average), along with nearly nine times the flower production of the S and CT treatments. In contrast, shredded materials only maintained 43% of the vegetative cover and 79% of the species richness of the original material placed in 2019. Unusually high precipitation during 2019 resulted in the flooding of several treatment plots. The flooding may have caused movement of shredded material (evidenced by ~50% reduction in organic material within treatment plots) and the dry early summer of 2020 may have desiccated whatever remained within treatment plots, with further movement of shredded material caused by the high wind characteristic of the study site. Despite the use of an erosion control mat, shredded materials will likely require even greater protection to ensure optimal growth, especially considering that extreme precipitation events in northeastern Canada are likely to become more frequent with the increasing effects of climate change in arctic environments (Walsh *et al*., 2020; Christensen *et al*., 2021). Whole turfs in our study also experienced these same conditions, indicating they are likely to endure transplantation and extreme weather with greater success than shredded materials.

3.6.2 Post-transplant Changes in Shredded Community Composition

The establishment of non-vascular species from the shredded treatment observed in this study is similar to the findings of others, specifically that shredded materials demonstrate non-vascular development (Aradottir, 2012) and transfer of vascular and non-vascular species from donor sites (Kiehl *et al*., 2010). Previous experiments have used large quantities of shredded material (up to 50 cm depths, compared to the 2 cm depths of this study). We had expected far greater non-vascular colonization, however, shredded materials in this study were derived from whole turfs including the underlying organic and mineral soils, effectively diluting potential non-vascular propagules that are constrained to the upper ~ 5 cm of tundra mats. The relatively thin layer of shredded materials applied with limited non-vascular materials in combination with the sensitivity of these newly deposited materials to desiccation (Rochefort *et al*., 2003) may have limited the effectiveness of our treatment. It must be noted that comparisons between shredded material and whole turfs represent plot-by-plot comparisons and do not consider differences in the amount of material used for each treatment. One shredded turf provided enough material to cover the same surface area of five whole turfs. If data is extrapolated to account for this difference, the application of shredded material could be considered comparable to whole turfs at promoting non-vascular cover over a wider area.

Transfer of vascular species via shredded materials was very limited, indicating that physical shredding resulted in significant damage to vascular structures such as rhizomes. While forb cover did increase within shredded materials, this may have been due to volunteer colonization. The two forb species with the greatest cover within shredded treatments were *D. sophioides* and *S. longipes*. *Descurainia sophioides* is a ruderal native annual and was most frequently observed in CT plots, and likely originated from the seed bank or seed rain. Similarly, turf treatments had roughly twice the cover of *S. longipes* than that found in S and CT plots, mimicking research demonstrating *S. longipes* readily expanded from transplanted turfs (Chapter 4.5.3). An additional 53% of the forb cover in S treatments was unidentified juvenile plants. Juvenile forbs had twice as much cover within shredded layers compared to control treatments or turfs, suggesting that shredded layers may have acted as effective seed traps or improved germination microsites. Overall, our results suggest that the use of shredded materials may be effective for development of primarily non-vascular communities. The application of non-vascular rich shredded materials (*i.e.*, harvesting only the top 5-10 cm of the tundra), spreading at greater depth, and with enhanced protection from wind and water erosion may improve establishment.

3.6.3 Post-transplant Changes in Turf Community Composition

Changes in the community composition of turfs post-transplantation confirmed our second hypothesis and were similar to those found in other turf transplant studies within both arctic (Kidd *et al*. 2006, Cater *et al*. 2015) and alpine environments (Aradottir, 2012; Bay & Ebersole, 2006). In a growth chamber experiment using upland heath communities from the same study site, we observed increases in forbs and graminoids and decreased lichen cover (Chapter 4.5.1). Increases in graminoid growth forms in turfs are frequently reported (Bay & Ebersole, 2006; Cole & Spildie, 2006), and it has been theorized that graminoid expansion occurs to take advantage of increasing inorganic N from mineralization associated with harvesting disturbance (Aradottir & Oskarsdottir, 2013). One of the graminoid species that increased in cover, *C. bigelowii*, has been shown to preferentially utilize NO_3^-N (Kelley & Epstein, 2009), which significantly increased within Q1 organic layers.

Others have reported reductions in species adapted to low-nutrient conditions, such as evergreen shrubs, due to increasing nutrient availability (Zamin *et al*., 2014; Gu & Grogan, 2020). Increasing pH may also have led to the significant decline in evergreen shrubs, and simultaneous increase in forb cover, with previous investigations finding evergreens, and associated fungal-dominated microbial communities are adapted to low-pH systems (Eskelinen *et al*., 2009). Previous turf transplants have also shown decreases in evergreen shrubs with turfs ≤ 30 cm^2 (Aradottir, 2012), suggesting that evergreens generally do not weather separation from the surrounding tundra well, potentially damaging root systems and fungal associations. Natural colonization studies indicate late-stage species such as *D. integrifolia* usually establish after ~60 years, with *C. tetragona* establishing after ~100-150 years, and only with acidification of underlying substrate and significant build-up of organic material (Hodkinson *et al*., 2003). Therefore, late-successional evergreen shrubs are likely not good candidates for restoration, even via intact turfs.

All lichen species decreased in cover within the transplanted turfs. In addition to shading from taller stature graminoids and forbs (Dormann & Woodin, 2002), high winds may have removed lichens from the turfs. This is supported by the presence of lichen species in comparable quantities to forbs and graminoids within CT plots and the large contribution of lichens within turf harvesting plots, indicating significant aeolian movement of lichen and moss fragments across the tundra. The decrease of lichen and moss cover may have been, in part, due to movement of the hummock-hollow complexes, with several turfs at Q2 demonstrating partial burial from loose rocks and stones.

3.6.4 Post-transplant Changes in Soil Nutrients

Nearly all nutrients demonstrated significant differences from harvest sites, often demonstrating much higher concentrations, notably of Cl^-, SO_4^{2-}, NO_3^-, Mg^{2+}, Ca^{2+}, and Na^+, leading us to reject our third hypothesis. Initial investigations of metal concentrations within local soils demonstrated high concentrations of various ions (Golder Associates, 2014), suggesting that the observed increases were likely due to the movement of these ions in soil solution across the quarries, especially during the heavy rainfall event in 2019. At each site, turfs with noticeable plant death also had higher concentrations of Na^+ and Cl^- (~100 to >2500 mg kg^{-1} and ~100 to >4000 mg kg^{-1}, respectively), with total turf death coinciding with the greatest concentrations observed. However, the turfs with the greatest concentrations were also noted to have flooded, another potential cause of turf mortality. Turf soils did not demonstrate any significant changes in C concentrations compared to the harvest sites and changes in N were inconsistent between sites. Overall, changes in the nutrient status of transplanted turfs were driven by nutrient conditions of the surrounding quarry substrate. Changes in the mineral layer were likely influenced by the inclusion of some underlying substrate during turf soil sampling. Overall, the ability of whole turfs to withstand an influx of soluble salts from the surrounding substrate suggests that turfs may support vegetative establishment even under poor soil conditions.

3.6.5 Vegetative Expansion

Expansion of vascular species from the treatment plots was largely confined to T and TS treatments and was minimal, with less than 25% of the immediate surroundings having any vegetative cover. Previous investigations of turf expansion have found 74% increases in turf area in Alaskan tundra (Shirazi *et al*., 1998) and 55% in Icelandic alpine heath (Aradottir, 2012), over two years. Our expansion was less than reported by these studies (*i.e.*, only ~ 17% vascular cover within 8 cm), however, an investigation into expansion using the same upland heath communities found limited aboveground development originating from transplanted low-Arctic turfs, with over 90% of expanding biomass contained within belowground structures (Chapter 4.5.4). Early expansion from these transplanted upland health communities may be driven by belowground expansion, and further studies that incorporate both above and belowground expansion are needed.

Vegetative expansion may have also been limited due to the conditions of the surrounding substrates, generally composed of coarse gravel, rocks, and stones with limited soil organic matter (SOM) content. Previous investigations into vegetative establishment within Arctic and alpine environments have stressed both the importance of fine particles and SOM to ensure establishment (Naeth & Wilkinson, 2014; Mehlhoop *et al*., 2018). High concentrations of soluble salts originating from the substrates may have had a significant impact on expansion, as many of the most severely impacted turfs demonstrated a significant reduction in plant cover, suggesting that amelioration of surrounding substrates may be required to ensure turf survival and subsequent expansion.

Due to the observed changes in turf community composition, the observed expansion, and results from previous research, we recommend that restoration practitioners working in upland heath environments focus on transplantation of forb and graminoid-dominated communities, particularly species from the *Fabaceae* family, notably *Astragalus alpinus* (Alpine Milkvetch), *H. alpinum*, and *O. maydelliana* for their survival rates, N-fixation capacities, and observed expansion in growth chambers, as well as species of *Carex*. *Cardamine digitata* and *B. vivipara* demonstrated significant survival and growth, along with *S. longipes*, known as a highly polymorphic species and aggressive colonizer. A few select shrub species are also recommended, such as the deciduous *S. reticulata*, the only shrub with increased growth within the turfs, and the evergreen *D. integrifolia*, which had negligible decreases in cover. Both shrub species have shown high survival rates and were the predominate shrubs responsible for belowground expansion under growth chamber conditions (Chapter 4.6.4). If a seed source could be obtained for the native annual *D. sophioides*, this may be an excellent candidate for quick establishment of vegetative cover, development of organic layers, and organic matter enrichment of the underlying soils, over non-native species that carry risks of spreading to surrounding tundra.

3.6.6 Harvest Recovery

Harvest recovery was very limited across all harvesting locations, with the greatest vegetative cover represented by lichens that had likely blown into the harvesting plots and vascular ingrowth from the surrounding tundra. Mosses represented a minor component of vegetative cover within harvesting plots, rejecting our last hypothesis. Since we used categorical cover values (*i.e.*, 1 through 4, representing increments of 25% cover) specific cover values for species with cover <25% are not available. However, no vascular species had a cover value over 1 (representing <25% cover), and in only three of the 176 harvesting grids did any non-vascular species have a cover value of 2 (representing 25-50% cover). Due to the limited recovery, we recommend that transplanted turfs not be harvested from undisturbed areas, rather, turfs should be harvested pro-actively from areas planned for future disturbance (Aradottir & Oskarsdottir, 2013; Cater *et al.*, 2015). Turf transplants are usually done as soon as possible following harvesting, and to our knowledge, there have been no investigations of the effects of stockpiling or temporary transplanting on the success of turf transplants in northern environments.

Overall, our two-year study demonstrated that turf transplants can maintain vegetative cover and species richness after two years, and can demonstrate resilience to flooding, drought, and an influx of less-than-desirable concentrations of soluble salts, potentially allowing their use in areas expected to be affected by extreme abiotic conditions, areas with poor substrate conditions (low organic matter and/or high concentration of soluble salts), and areas with uncertain weather patterns amidst a changing climate. Successful transplanting combined with the limited resources, time, and effort required to transplant turfs suggests this technique may be an effective means of re-establishing native vegetative communities onto disturbed sites within low-arctic tundra, without the reliance on non-native species or fertilization. Importantly, while the community composition of turfs may change and therefore are not identical to pre-disturbance conditions, most species transferred are likely to survive transplantation, providing disturbed areas with the necessary species required to regenerate self-organizing assemblages of native vegetation. Shredded turfs, while demonstrating comparable establishment of non-vascular species over a wide area, demand greater protection from environmental conditions, limiting the sites within low-arctic environments where this technique is effective, especially considering expected climatic alterations and increased frequency of extreme weather events. However, targeting non-vascular propagules for shredded treatments, increasing the depth of applied materials, and providing surface protection may increase the feasibility of this approach. Due to the slow recovery of turf harvesting locations, turfs used for transplantation should not be harvested from undisturbed sites, rather, plans should be made proactively to ensure that turf materials are harvested from areas slated for development.

4. VEGETATIVE GROWTH AND BELOWGROUND EXPANSION FROM TRANSPLANTED LOW-ARCTIC TUNDRA TURFS[2]

4.1 Preface

Turf transplants show promise for the establishment of native intact arctic plant-soil communities. However, investigation of transplant success and expansion has been limited to aboveground characteristics, ignoring development and expansion of vegetation below the soil surface. In this chapter, the effects of fertilization of the surrounding substrate on turf community composition, and above and belowground expansion are investigated. DNA metagenomics is utilized to characterize the belowground community composition of expanding roots and provide species identification.

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[2] This chapter was accepted for publication on April 29th, 2022 to *Restoration Ecology* as Hnatowich, I., Lamb, E.G., & Stewart, K.J. (2021). Vegetative growth and belowground expansion from transplanted low-Arctic tundra turfs. https://doi.org/10.1111/rec.13716. Dr. Lamb and Dr. Stewart assisted in developing the experimental design, data collection, data analysis, and editorial input. I performed the experiment, collected data, completed the data analysis, and completed the initial manuscript draft. Funding for this research was provided by NSERC's Collaborative Research and Development grant (CRDPJ 518281-17) in collaboration with Agnico Eagle Mines Ltd.

4.2 Abstract

Whole turf transplantation is a restoration method used to restore plant communities within disturbed arctic environments. Transplant expansion and restoration success is often determined based on aboveground characteristics, and to our knowledge, this is the first investigation of belowground expansion from transplanted turfs. In this growth chamber experiment, turfs harvested from undisturbed tundra near Rankin Inlet, Nunavut, Canada, were exposed to fertilized and unfertilized substrates to determine the effect of adjacent nutrient-enrichment on plant community composition within the turfs and substrates, as well as above and belowground biomass and expansion. Next-generation sequencing was used to determine the species identity of expanding roots. Our results show that fertilization of substrates surrounding tundra transplants did not alter the community composition of the turfs, but did increase biomass and expansion, as well as biological soil crust cover on the adjacent substrate. Belowground biomass far exceeded aboveground revealing the importance of evaluating belowground roots and rhizomes that dominate the vegetative biomass within arctic ecosystems. Investigation of belowground development is likely to provide holistic interpretations of restoration success and should not be ignored in future transplantation studies.

4.3 Introduction

Low-arctic environments are characterized by extreme conditions that result in slow rates of ecosystem recovery following disturbance. Therefore, restoring these environments can be particularly challenging. Short growing seasons and harsh climates (*i.e.*, low temperatures and low rates of precipitation) inhibit decomposition, nutrient cycling, seed production and seed maturation, plant colonization, and reduce the number of colonizing species (Naeth & Wilkinson, 2014; Mehlhoop *et al*., 2018). Due to these environmental conditions, timeframes for natural colonization and ecosystem recovery are long, ranging from decades to hundreds of years following disturbance (Forbes & Jefferies, 1999; Bay & Ebersole, 2006; Johnstone & Kokelj, 2009). Revegetated ecosystems may not recover to a similar functional state equivalent to pre-disturbance conditions (Fenger-Nielsen *et al*., 2019). Regardless of the environmental conditions, the remote locations of disturbed low-arctic environments, the limited knowledge on plant recovery in differing arctic ecotypes, along with the lack of local plant materials offer additional challenges to restoration practitioners.

Through previous restoration attempts and studies in arctic and alpine environments, important lessons have been learned, including the importance of soil organic matter (SOM) for plant establishment and development of successional stages (Hodkinson *et al*., 2003; Naeth & Wilkinson, 2014; Mehlhoop *et al*., 2018), and the response of many arctic growth forms to fertilization and increased nutrient availability (Gough & Hobbie, 2003; Gretarsdottir, Aradottir, Vandvik, Heegaard & Birks, 2004; Kelley & Epstein, 2009). Fertilization of low-arctic tundra in conjunction with restoration and revegetation has been recommended to promote plant community development in wet-sedge meadows (Kidd *et al*. 2006), and significantly decrease the expected timeframes for plant colonization (Naeth & Wilkinson 2014). Increased biological soil crust (BSC) development on fertilized soils in the high-Arctic and Icelandic alpine have also been observed (Hodkinson *et al*. 2003; Gretarsdottir *et al*. 2004). In contrast, fertilization in these nutrient-limited environments can result in significant changes in community composition and structure (Kelley & Epstein, 2009; Zamin *et al*., 2014; Gu & Grogan, 2020), limiting its use for restoration of natural assemblages of tundra vegetation. However, fertilization may still prove an effective means of promoting functionally similar species assemblages to pre-disturbance conditions.

Whole turf or sod transplantation (*i.e.*, harvesting and transplanting intact soil-plant plugs) for restoration has been examined in several arctic and alpine environments, ranging from alpine grassland and dry alpine heaths (Bay & Ebersole, 2006; Aradottir & Oskarsdottir, 2013; Mehlhoop *et al*., 2018), w*et al*pine (Conlin & Ebersole, 2001) and arctic sedge meadows (Kidd *et al*., 2006; Cater *et al*., 2015). Differences in the efficacy of turf transplants between arctic environments are likely due to differences in vegetation types, nutrient availability, and soil conditions. There is still a paucity of turf transplant studies overall, and specifically in arctic upland-heath tundra. However, known benefits of turf transplants include initiating restoration efforts with established and functional assemblages of native plant species (Cater *et al*., 2015), development of safe sites for seed and spore germination, the inclusion of soil resources (*i.e.*, SOM, plant propagules, soil invertebrates, and microbial communities) (Conlin & Ebersole, 2001; Klimeš *et al*., 2010), maintenance of high species diversity and presence of rare species (Aradottir, 2012; Aradottir & Oskarsdottir, 2013), and reduced shock to individuals within the transplants (Bay & Ebersole, 2006). In remote areas, whole turf transplants can be carried out with limited equipment.

To our knowledge, no studies have examined active BSC (*i.e.*, soil surface assemblages of mosses, liverworts, lichens, bacteria, cyanobacteria, and fungi) restoration or transplanting in an arctic environment. Examinations of BSC restoration and function are frequently conducted in warmer, drier environments (Antoninka *et al*., 2020; Bowker *et al*., 2020). However, studies in low-arctic tundra have found BSC communities contribute significantly to landscape-scale nitrogen (N) fixation (Stewart, *et al*., 2011a), aid in the germination and establishment of seedlings in alpine tundra (Gold *et al*., 2001), and can facilitate the development of early and mid-successional plant communities in high-Arctic glacial forelands (Breen & Lévesque, 2006).

Previous transplant efforts have been successful in restoring degraded arctic and alpine sites; however, restoration success from turf transplants, to our knowledge, has only been based on the assessment of aboveground growth and expansion, neglecting to examine the corresponding belowground growth and expansion. In almost all arctic environments plant roots represent the majority of biomass, contribute to soil carbon (C) sequestration and organic matter, and act as long-lasting sources of nutrients for microbial communities (Iversen *et al*., 2015). Assessing restoration success based only on aboveground growth ignores both the substantial development of belowground structures and considerable contribution to nutrient cycling regimes within arctic environments (Iversen *et al*., 2015).

Due to the dominance and importance of belowground biomass, and the limited knowledge on belowground expansion, we investigated both above and belowground expansion of low-arctic upland-heath tundra turf transplants adjacent to fertilized or unfertilized bare substrates. The first objective of this study was to determine the effect of fertilization on both the turf and substrate community composition. We hypothesized that fertilization would alter the community composition of both the turfs and substrates. Secondly, we examined the effect of fertilization on biomass expanding from transplanted turfs and hypothesized that fertilization would encourage greater above and belowground expansion into nutrient-enriched substrates. Our final objective was to determine the identity of species responsible for expansion, with graminoid species hypothesized to dominate the expanding communities.

4.4 Methods

4.4.1 Study Site

Plant-soil turfs were harvested from an area of undisturbed upland tundra heath located on Agnico Eagle Mines Meliadine site (63°01'22.9"N 92°11'41.1"W DMS) near Rankin Inlet, Nunavut, Canada. Meliadine lays within the Maguse River Upland Ecoregion of the Southern Arctic Ecozone, characterized by long, cold winters (seasonal temperature -24°C, Nov-Apr) with short, cool summers (seasonal temperature 7°C, June-Sept), and an average precipitation of 250 to 400 mm per year (Ecological Stratification Working Group, 1995; Environment Canada, 2021). Soils are well-drained sand, silt, and gravel tills, usually overlain with thin organic horizons. Due to cryoturbation, extensive and well-developed hummock-hollow microtopography is present. The dominant soil classes are Turbic Cryosols with Regosolic features, followed by Organic and Turbic Cryosols (Golder Associates, 2014). Local vegetation is dominated by upland heath communities associated with well-drained soils. Heath tundra, heath-lichen, and lichen-rock vegetation comprise over half of the local plant communities. Common species include dwarf shrubs *Cassiope tetragona* (Arctic Mountain Heather), *Dryas integrifolia* (White Mountain-avens), *Salix reticulata* (Net Veined Willow), low-lying forbs *Astragalus alpinus* (Alpine Milkvetch), *Pedicularis flammea* (Red-tipped Lousewort), *Cardamine digitata* (Richardson's Bittercress), mosses *Aulacomnium turgidum* (Swollen Thread Moss), *Polytrichum pallidisetum* (Mountain Haircap Moss), *Hylocomium splendens* (Stairstep Moss), and lichens *Dactylina arctica* (Finger Lichen), *Thamnolia vermicularis* (White Worm Lichen), *Cetraria spp.,* and *Cladonia spp.* (Golder Associates, 2014).

4.4.2 Turf Harvesting

Plant-soil turfs were harvested within upland-heath tundra in early July 2019. A spiral search was used to identify eight suitable turfs, with at least 5 metres of separation between each harvested turf. Suitable turfs occurred on the upper part of large hummocks which had relatively level topography and nearly continuous vascular plant cover. Turfs approximately 46 cm long, 28 cm wide, and 10 cm deep were dug out using a flathead shovel and lifted into plastic bins. Turf soils were composed of approximately 8-10 cm of organic soil with two centimetres of mineral soil below. Plant species cover was measured for each turf on-site, within three days of harvest, using a 0.096 m^2 gridded quadrat (15 grids, 0.0064 m^2 each). Values ranging from 1 to 4 (1 = <25%, 2 = 25-50%, 3 = 50-75%, 4 = >75%) were used to estimate each species' cover in each individual grid square. Turfs were transported to the University of Saskatchewan and placed in growth chambers [18.5hrs light (15°C) and 5.5hrs dark (5°C) cycle at ~400 µmol, relative humidity 65%] for seven months. Plant species cover was re-measured on each turf before experimental treatment began to determine any changes in the plant community since harvesting. Significant changes in the community composition that were detected since the time of harvest were decreased cover of lichens and deciduous shrubs and increased cover of plant litter, evergreen shrubs, and mosses (Fig. B.1, Tables B.1- B.2). These changes were likely due to optimal growing conditions present within growth chambers.

4.4.3 Experimental Design

Plastic containers (50 cm long x 30 cm wide x 15 cm deep) with drainage holes were lined with a thin layer of cheesecloth. Turfs were scored lengthwise using a flathead shovel and separated by hand to prevent damage to each half. Separated turfs were then placed against one side of a container (individual containers for each turf), with a mesh screen (12 cm^2 openings) placed underneath the turf, and across the growing front. In a paired design, one half of each of the eight turfs was placed in contact with a fertilized soil substrate (10 g m^{-2} N (NH_4NO_3), 5 g m^{-2} P (P_2O_5)), and the other half with unfertilized soil substrate (Fig. B.2). Soil substrates (hereby referred to as “substrate”) were sourced from a local distributor (Rock Bros, Saskatoon, SK), and fertilized through dry mixing of powdered material before being placed alongside the fresh growing front. The substrate was added until the surface of the turf and substrate were level (*i.e.*, ~ 10-12 cm depth). The texture of the substrate varied from sand to sand/loam, and the particle size distribution (clay 0%, silt 12-17%, sand 83-88% (Horiba LA950 Particle Size Analyzer, Japan)) was similar to substrates encountered at Rankin Inlet. Each container (turf and substrate) was watered twice per week using reverse osmosis water (643 mL/container/watering) to simulate the natural precipitation regime of Rankin Inlet during the growing season and were kept in growth chambers for a total of 140 days. Turfs were randomly repositioned and photographed weekly within the growth chambers.

After the 140-day incubation period, the turfs were photographed using a multi-spectral camera (Parrot Sequoia, U.S.A) to generate normalized-difference vegetative index (NDVI) values for each container, used as a general measurement of quantity and health of vegetation present within each container. Images representing the near-infrared and red spectral reflectance were superimposed over each other in ArcMap 10.7.1 (ESRI, 2011), and the raster calculator function used to calculate the NDVI values for each container (ESRI, 2021b). NDVI images were then separated into turf and substrate sides by splitting the raster image along the boundary of the mesh screen, allowing assessment of the quantity of healthy vegetation within each section. Raster data was extracted (ESRI, 2021a), and weighted mean NDVI values for each image were calculated by taking the sum product of all values and their corresponding pixel counts, divided by the total number of pixels in the image.

To assess community composition, plant growth, and species richness on the turfs and on the substrate, plant species cover across the entire container was measured using a 0.15 m^2 gridded quadrat (25 grids delineating sampling cells 0.006 m^2 each, 10 grids overtop the vegetated turfs, 15 grids overtop the substrate) after the 140-day incubation (Fig. B.2). All vascular plants, mosses, and lichens were identified to species and percent cover determined in each grid cell. The percent cover of additional biological soil crust components was recorded as crusted surfaces (*i.e.*, black crust and white crustose lichens), cyanobacterial colonies (*i.e.*, cyanobacterial mats and *Nostoc* spp.) and fungal fruiting bodies in each grid. All non-vascular species and BSC components were intermingled and considered part of the crust community, except when determining cover differences in growth forms, where moss, lichen and BSC components were examined separately (*e.g.*, Table 4.1). Cover of rocks and stones, woody litter (*i.e.*, woody branches, stems, etc.) and plant litter (*i.e.*, dead plant tissues, leaves, flowers, etc.) were also measured in grid cell.

All vascular plants emerging above the substrate (*i.e.*, individual vascular plant stems, hereby referred to as “emergents”) were counted in each grid and the height, lateral spread (*i.e.*, the horizontal distance of emergents), and distance of each vascular emergent from the turf (*i.e.* the mesh screen) were measured. Emergents were harvested at ground level after percent cover measurements, dried for 72 hours at 40^{o}C, and weighed to determine biomass. The side of each container farthest from the turf was cut away, and the substrate in each cell (*i.e.*, grid surface area delineated a cell sampled to the depth of the substrate; total n = 15 cells in each container x 16 containers = 240 cells) was carefully washed out to harvest the roots in each cell. Collected roots were then dried for 72 hours at 40^{o}C, and subsequently freeze-dried to ensure all moisture was removed. Dried roots were then weighed, before being ground with a ball mill (Retsch MM-400, Germany).

The identity of roots extending into the new substrate was determined using DNA barcoding (Lamb *et al*., 2016). Root DNA was extracted from ~0.7 g of ground material using a PowerPlant Pro kit (Qiagen, Germany). All protocols from the kit were followed except for minor deviations (detailed sequencing information can be found in Appendix B). Of the 240 cells sampled for belowground biomass, 14 cells contained insufficient biomass to successfully extract DNA, yielding a total of 226 root DNA samples. Sequencing was performed at the Toxicology Centre at the University of Saskatchewan, using a 300-cycle MiSeq v2 kit (Illumina, San Diego, USA).

TrnL bioinformatics was performed within mothur 1.40.5 (Schloss *et al*., 2009) following a modified version of the bioinformatics pipeline developed by Lamb *et al*., (2016). Forward and reverse reads were merged and trimmed to the primer region then aligned to a template created by Lamb *et al*., (2016). Sequences were quality filtered, chimeras removed, clustered into 97% similar Operational Taxonomic Units (OTU) and classified using a custom local database. The local database included species that had been observed in the plant-soil turfs during any recording event (original harvesting, before separation, after treatment implementation, and before root harvesting), with reference sequences obtained from the National Center for Biotechnology Information’s nucleotide database. Sequences were classified based on at least 90% similarity to reference database sequences. Additional filtering to reduce noise included removing any sequence within a given sample that contained ≤ 10 reads, as well as removing any species that contained less than 100 reads across all samples. All OTU’s detected were identified to species level, except for an unidentified *Salix* and *Fabaceae* sp., which we were unable to identify beyond family level.

4.4.4 Statistical Analysis

All statistical analysis was conducted in R 3.6.1 (R Core Team, 2020). Percent cover data were first separated to create two datasets, one containing only plant cover on the turfs, and the other with plant cover for the substrates, allowing us to examine changes in community composition within the harvested turf and on the newly colonized substrate, respectively. Plant cover data for both datasets were converted into a Bray-Curtis distance matrix. Two non-metric dimensional scaling (NMDS) ordination plots using the ‘nmds’ function with the package ‘vegan’ (Oksanen *et al*., 2020) were used to visualize the community composition data, and permutational multivariate analysis of variance (PerMANOVA) was performed to test for statistically significant differences in plant communities; i) between the time of harvesting and the start of the experiment, and ii) between turf community composition at the start and end of the experiment, and iii) between fertilized and unfertilized treatments at the end of the experiment, using the “adonis” function in the vegan package. When differences were encountered, similarity percentage analysis (SIMPER) was performed to identify influential species responsible for community dissimilarities, using the function ‘simper’ within the vegan package. To avoid the contribution of vascular overgrowth from the vegetated turfs influencing the substrate community composition data, the first six centimetres of the substrate community data were excluded from analysis, with all statistical analysis of community composition performed on the remainder of the substrate (*i.e.*, 6-18 cm from the turfs). All statistical analysis of cover was conducted using absolute cover values.

Paired Student’s t-tests were used to test for treatment differences (fertilized versus unfertilized) between mean weighted turf and substrate NDVI, total above and belowground biomass, as well as the number of emergents, mean height, mean distance from the turfs, and mean lateral spread of emergents, using the function ‘t.test’ within the ‘stats’ package (R Core Team, 2020). Paired turfs represented two halves of the same original turf and therefore were not considered independent. All data was inspected to ensure that assumptions of paired Student’s t-tests were not violated. When data did not meet assumptions, data were transformed as follows: number and height of emergent were log-transformed, and substrate NDVI were reciprocal transformed. Two-way within-subjects analysis of variance (ANOVA) were performed to test effects of treatment and distance, as well as interactions between treatment and distance, for above and belowground biomass, biological soil crust cover, and species richness, using the ‘ezANOVA’ function within the ‘ez’ package (Lawrence, 2016). All data were inspected to ensure the assumptions of within-subjects ANOVA were met. When data did not meet assumptions, data were transformed as follows: BSC cover, species richness, and belowground species richness were square-root transformed and belowground biomass was reciprocal transformed. Neither aboveground biomass nor species richness met model assumptions. Post-hoc pairwise comparisons using the ‘pairwise.t.test’ within the stats package were used to determine statistical differences between groups.

4.5 Results

4.5.1 Aboveground Community Composition and Growth

Fertilization of the adjacent substrate did not affect the community composition of the turfs after 140 days of growth (df = 31, R^2 = 0.04, p = 0.25), while the composition of the turfs did change over time (df = 31, R^2 = 0.16, p = 0.001). Significant interactions between treatment and time were not detected (Fig. 4.1) (Table B.3). Over time we observed decreases in lichen and plant litter cover and increases in evergreen shrub, moss, and graminoid cover (Table 4.1). Plant community composition on the fertilized substrate was significantly different from the unfertilized substrate (df = 15, R^2 = 0.42, p = 0.001) (Fig. 4.2). Mean NDVI of the turfs was not significantly different between fertilized and unfertilized treatments (df = 7, t = 0.64, p = 0.5), however, the mean NDVI of the fertilized substrates was significantly higher than unfertilized substrates (df = 7, t = -3.5, p = <0.01) (Fig. 4.3).

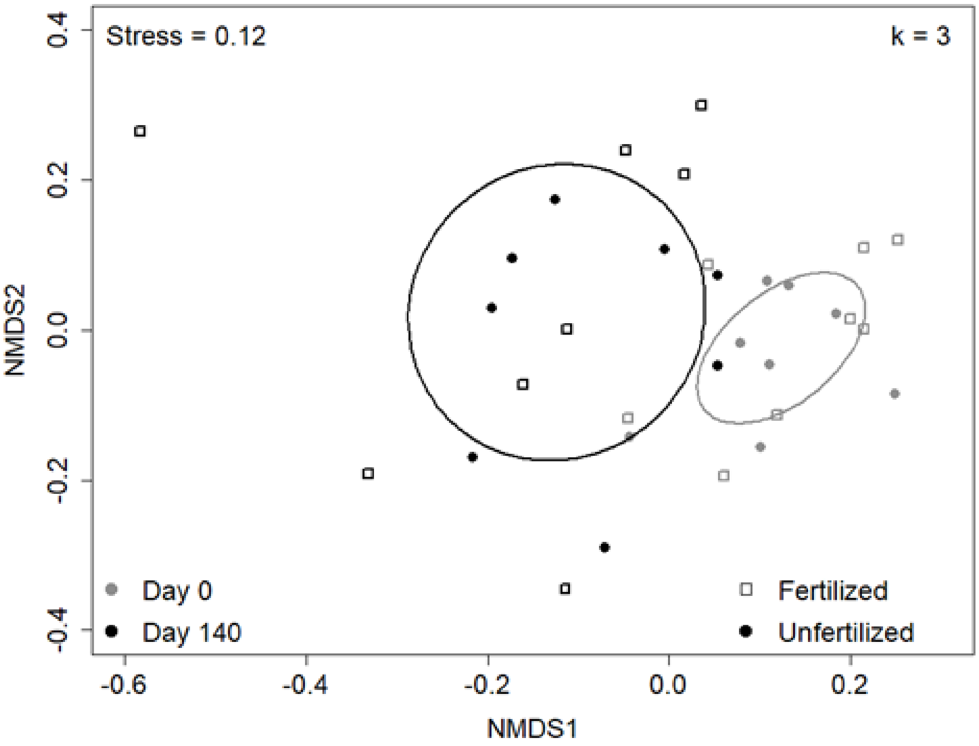

Figure 4.1. A non-metric multidimensional scaling ordination of aboveground turf community composition placed adjacent to fertilized (hollow square) and unfertilized (filled circle) substrates, at the start of the experiment (Day 0- grey) and end of the experiment (Day 140- black). Ellipses represent the standard error of the weighted average of scores.

Table 4.1. Growth forms, species, litter, and bare ground that contributed to >70% of the overall dissimilarity between Day 0 and Day 140 for turfs and between fertilized and unfertilized substrates on Day 140. Absolute and relative percent differences between Day 0 and Day 140 turfs and between fertilized and unfertilized substrates were calculated. Absolute percent differences are (species cover Day 140 or fertilized – species cover Day 0 or unfertilized)/species cover Day 0 or species cover unfertilized), respectively. Relative percent differences are ((species cover Day 140 or species cover fertilizer/total species cover Day 140 or total species cover fertilizer) - (species cover Day 0 or species cover unfertilized/total species cover Day 0 or total species cover unfertilized)), respectively. Species that were not recorded at Day 0 or within unfertilized treatments but recorded at Day 140 or within fertilized treatments are recorded as 100% increase in absolute cover.

Section	**Growth Form**	**Dissimilarity Contribution (SIMPER)**	**Absolute Percent Difference (Relative)**	**Species**	**Absolute Percent Difference (Relative)**
Turf	Lichen	25%	-51% (-14%)	*Thamnolia vermicularis*	-66% (-4%)
				Cetraria nivalis	-44% (-4%)
				Gowardia nigricans	-54% (-3%)
				Alectoria ochroleuca	-40% (-1%)
	Litter	16%	-52% (-10%)	Plant Litter	-51% (-5%)
				Woody Litter	-58% (-6%)
	Moss	13%	+43% (+8%)	*Aulacomnium turgidum*	+42% (+3%)
				Dicranaceae sp.	-27% (-2%)
	Evergreen Shrubs	13%	+15% (+3%)	*Dryas integrifolia*	+47% (+3%)
				Cassiope tetragona	-16% (-1%)
	Graminoids	10%	+95% (+8%)	*Carex sp.*	+100% (+3%)
				Poa flexuosa	+100% (+4%)
Substrate	Bare Ground	37%	-55% (-40%)	Bare Soil	-70% (-30%)
				Rocks and Stones	-31% (-11%)
	Moss	27%	+123% (+18%)	*Pohlia nutans*	+160% (+7%)
				Dicranoweisia crispula	+103% (+5%)
				Aulacomnium turgidum	+3% (-1%)
	Biological Soil Crusts	18%	+169% (+12%)	*Nostoc* spp.	+258% (+7%)
				Cyanobacterial mats	+100% (+4%)

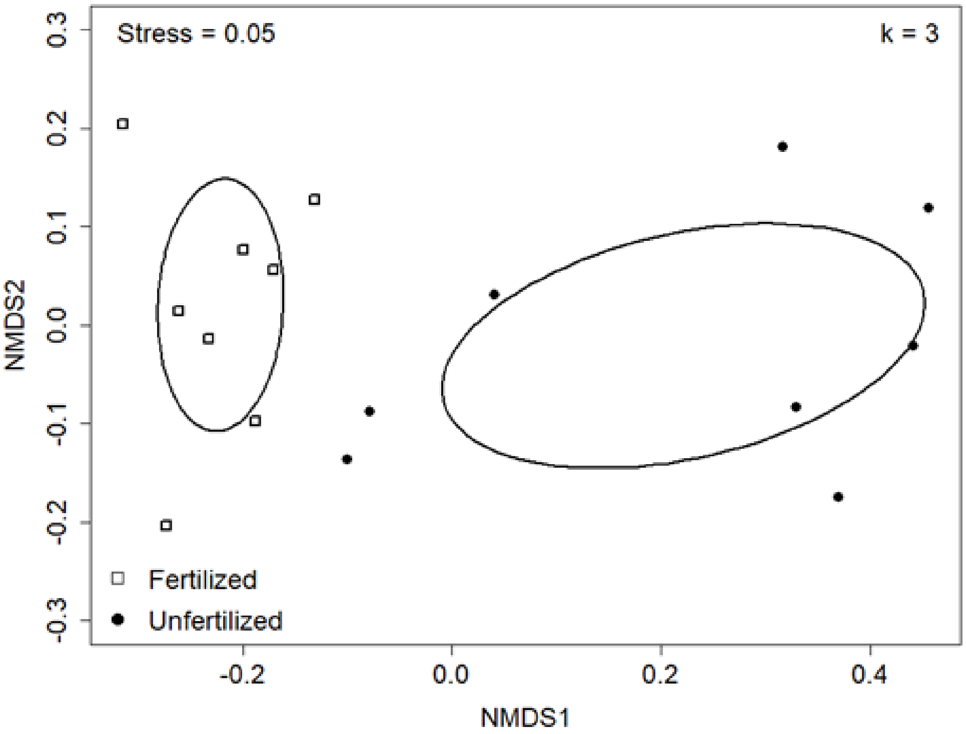

Figure 4.2. A non-metric multidimensional scaling analysis of aboveground substrate community composition of fertilized (squares) and unfertilized (circles) substrates. Ellipses represent the standard error of the weighted average of scores.

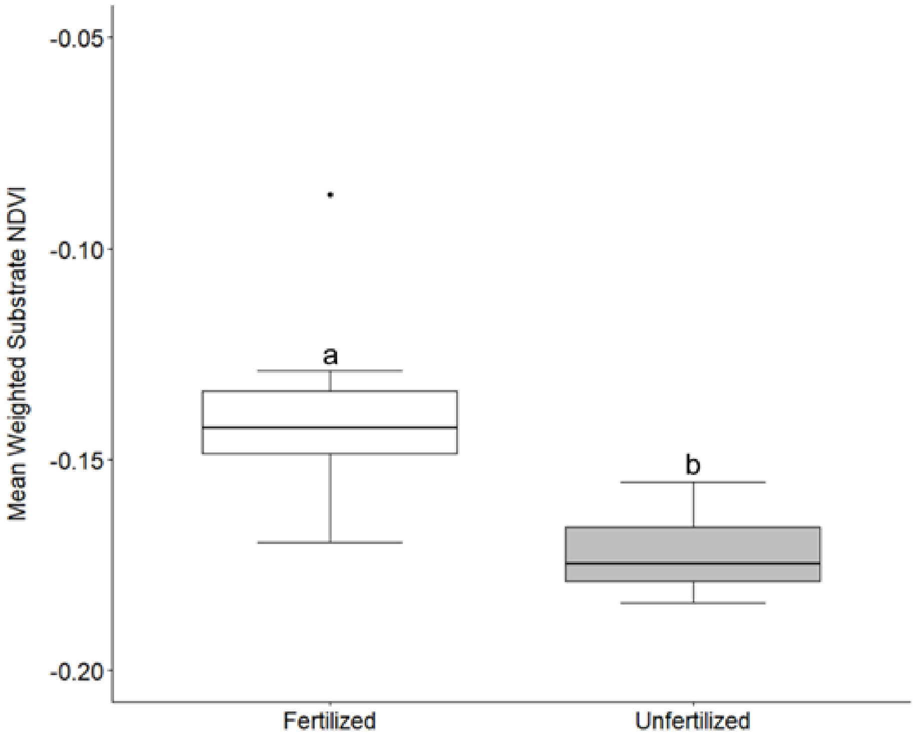

Figure 4.3. Mean weighted substrate NDVI values between fertilized and unfertilized substrates. Boxes encompass 25-75% percentiles of the data, while whiskers encompass 5-95%. The median is indicated by the black horizontal line, with dots representing points outside of the 95th percentile. Different letters indicate significantly different NDVI values (Paired t-test, t-stat = -3.5, p = <0.01).

4.5.2 Biological Soil Crusts and Species Richness

Biological soil crust cover and species richness on fertilized substrates were significantly higher ($F(_{1,7}) = 25$, $p = < 0.01$; $F(_{1,\ 7}) = 15$, $p = < 0.01$), with both metrics approximately double that of unfertilized substrates (Table 4.1). Biological soil crust cover and richness were not significantly different with distance from the turf and the fertilizer × distance interaction was not significant (Fig. 4.4A) (Table B.4). Cover on both substrates was dominated by moss (~63% of total cover), with cyanobacterial mats and lichen species comprising 28% and 8% of total cover, respectively. *Pohlia nutans* (Copper Wire Moss), *Dicranoweisia crispula* (Mountain Pincushion), and *A. turgidum* were the most common moss species, while cyanobacterial components included *Nostoc* spp. Most BSC species occurred on both fertilized and unfertilized substrates, except for *Bryum blindii*, which only occurred on fertilized substrates (Fig. 4.5A) (Fig. B.3).

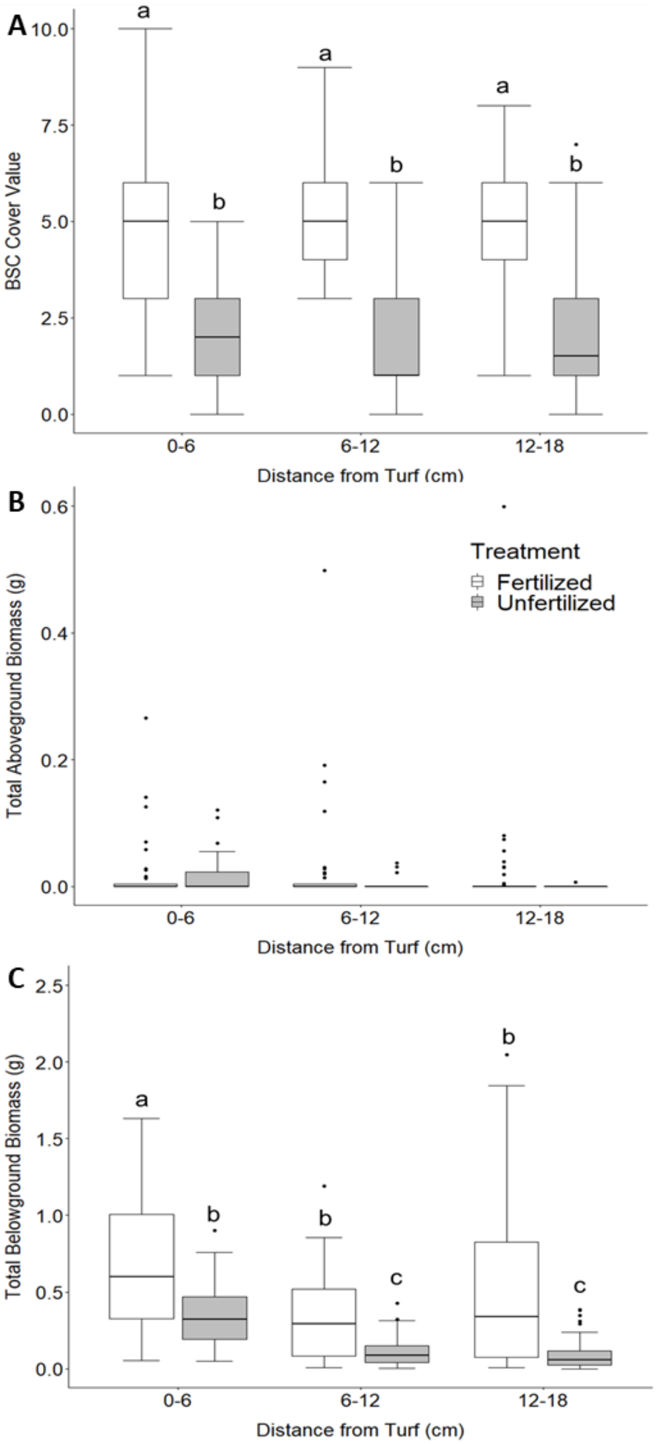

Figure 4.4. Comparative box plots (untransformed data) of biological soil crust cover (A), aboveground vascular biomass (B), and belowground biomass (C), at each distance category (0-6, 6-12 and 12-18 cm). White boxplots represent fertilized treatments, grey boxplots represent unfertilized treatments. Lower-case letters on boxplots represent significant differences or similarities between groups. The boxplot of biological soil crust cover uses the raw values recorded in each grid.

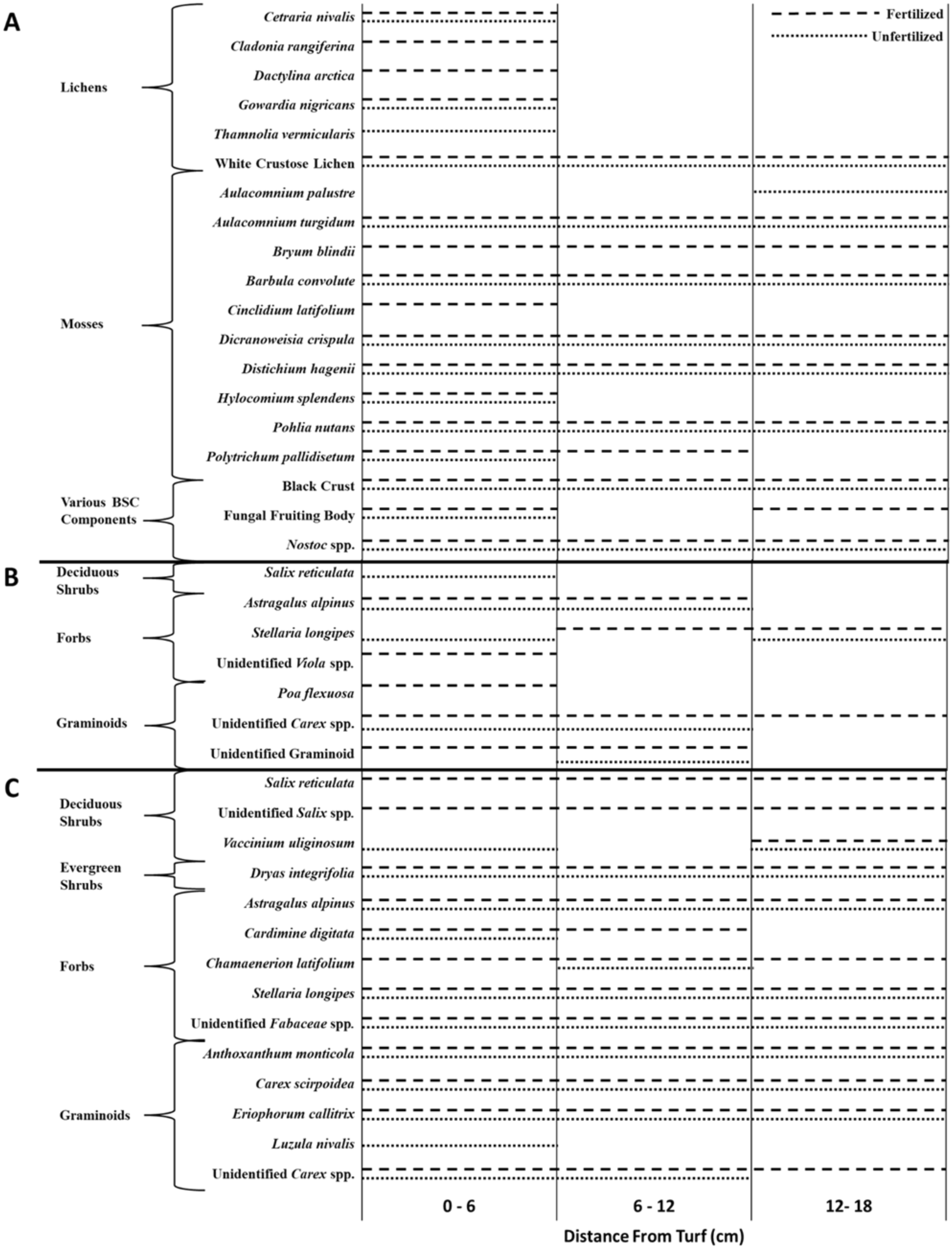
A
Lichens
Cetraria nivalis
Cladonia rangiferina
Dactylina arctica
Gowardia nigricans
Thamnolia vermicularis
White Crustose Lichen
Mosses
Aulacomnium palustre
Aulacomnium turgidum
Bryum blindii
Barbula convolute
Cinclidium latifolium
Dicranoweisia crispula
Distichium hagenii
Hylocomium splendens
Pohlia nutans
Polytrichum pallidisetum
Various BSC Components
Black Crust
Fungal Fruiting Body
Nostoc spp.
Fertilized
Unfertilized
B
Deciduous Shrubs
Salix reticulata
Forbs
Astragalus alpinus
Stellaria longipes
Unidentified Viola spp.
Graminoids
Poa flexuosa
Unidentified Carex spp.
Unidentified Graminoid
C
Deciduous Shrubs
Salix reticulata
Unidentified Salix spp.
Vaccinium uliginosum
Evergreen Shrubs
Dryas integrifolia
Forbs
Astragalus alpinus
Cardimine digitata
Chamaenerion latifolium
Stellaria longipes
Unidentified Fabaceae spp.
Graminoids
Anthoxanthum monticola
Carex scirpoidea
Eriophorum callitrix
Luzula nivalis
Unidentified Carex spp.
0 - 6
6 - 12
12- 18
Distance From Turf (cm)

Figure 4.5. Presence of individual species on/in fertilized/unfertilized substrates at each distance category (0 - 6, 6 - 12, and 12 - 18 cm), separated into biological soil crust (A), aboveground species (B), and belowground species (C). The presence of species within fertilized substrates is represented by dashed lines, and unfertilized substrates are represented by dotted lines.

4.5.3 Vascular Plants

We were unable to model vascular species richness, however, we observed similar species richness immediately adjacent to the turfs (0 - 6 cm) in fertilized and unfertilized substrates (0.33 ± 0.32 and 0.43 ± 0.36, respectively), while at the furthest distance assessed (12 - 18 cm) fertilized substrates had higher mean vascular richness than unfertilized (0.28 ± 0.41 and 0.03 ± 0.07 respectively). Both fertilized and unfertilized substrates contained similar assemblages of vascular species, namely *A. alpinus*, *Stellaria longipes* (Longstalk Starwort), and *Carex* spp. (Fig. 4.5B) (Fig. B.3). *S. longipes* demonstrated the farthest growth from the turf, often emerging against the farthest wall of the container, and was more prevalent in the fertilized compared to the unfertilized substrates (46 vs 3 emergents, respectively). Mean aboveground vascular biomass was higher in the fertilized substrates than the unfertilized substrates, with approximately three times more emergents, although differences between treatments were not significant. No significant effect of fertilizer treatment was detected for any other aboveground measurements (*i.e.*, distance from the turf, mean height, mean lateral spread).

4.5.4 Belowground Community Composition and Growth

Belowground species richness was not significantly different with fertilization, distance, or their interaction. Both fertilized and unfertilized substrates contained the same species, except for *Luzula nivalis* (Arctic Wood Rush), which occurred only within unfertilized substrates. Nine vascular species were detected in greater prevalence within fertilized substrates, most notably *Anthoxanthum monticola* (Alpine Sweetgrass), *S. longipes*, and *S.reticulata*, which were detected in 56%, 31%, and 18% of fertilized cells, compared to 37%, 13%, and 1% cell within unfertilized substrates respectively (Fig. B.3). Unfertilized substrates had a greater prevalence of four species, *D. integrifolia*, *A. alpinus*, *Eriophorum callitrix* (Arctic Cottongrass), and an unidentified *Fabaceae spp*., being detected in 51%, 20%, 10%, and 14% unfertilized cells, yet only detected in 22%, 8%, 7%, and 11% fertilized cells, respectively. It should be noted that while these species seemingly favored unfertilized substrates, each was found expanding at least 12 cm from the turf in fertilized substrates as well (Fig. 4.5C).

Belowground biomass was significantly higher in the fertilized substrate than the unfertilized substrates ($F_{(1, 7)} = 6.3$, $p < 0.05$). Fertilized substrates had more belowground biomass than unfertilized at each distance increment (0 - 6 cm, df = 39, t = -3.68, p = <0.001; 6 - 12 cm, df = 39, t = -5.98, p = <0.001; 12 – 18 cm, df = 39, t = -4.87, p = <0.001). The belowground biomass was significantly higher at 0 - 6 cm compared with 6 - 12 cm and 12 - 18 cm in both substrates ($F_{(2, 14)} = 17$, $p < 0.001$) (Fig. 4.4C). A significant fertilizer × distance interaction was detected ($F_{(2, 14)} = 3.9$, p = <0.05). The increase in biomass observed within the fertilized substrates at 12 to 18 cm was due to belowground biomass found pressed against the farthest wall of the container, indicating that container size limited belowground expansion in the fertilized treatment, and likely caused the significant fertilizer × distance interaction. Overall, total belowground biomass was 22-fold (fertilized) and 12-fold (unfertilized) greater than aboveground biomass.

4.6 Discussion

4.6.1 Community Composition and Growth within Harvested Turfs

The observed alterations to turf community composition over time were expected, based on the results from previous turf transplantation in alpine grasslands (Bay & Ebersole, 2006), alpine heath (Aradottir & Oskarsdottir, 2013), and arctic wet sedge meadows (Kidd *et al*., 2006; Cater *et al*., 2015) all noting significant increases of graminoid species, along with decreases in non-vascular cover. Counter to our first hypothesis, turf community composition was not significantly different when placed adjacent to fertilized or unfertilized substrates. In contrast to many direct fertilization or artificial warming experiments conducted *in situ* within arctic and alpine environments, the similarity of both turf community composition and NDVI values between treatments is interesting, as direct fertilization and greater nutrient availability in nutrient-limited tundra environments often results in significant community alterations (Gough & Hobbie, 2003; Kelley & Epstein, 2009). These changes are often characterized by an increase in graminoid, forb, and deciduous growth forms (which are known to respond favorably to nutrient-additions either directly through fertilization or indirectly through mineralization resulting from harvesting disturbance), as well as decreases in species adapted to low-nutrient conditions, or low-stature species, such as evergreen shrubs and non-vascular species (Chapin, Shaver, Giblin, Nadelhoffer & Laundre, 1995; Zamin *et al*., 2014; Gu & Grogan, 2020).

The cover of evergreen shrubs and mosses also increased during the experiment. Fertilization of surrounding substrates may have a limited impact on nutrient availability within turfs, allowing for the continued presence of species with lower nutrient requirements, as well as maintaining the community composition of soil biota (*i.e.*, bacteria, fungi and invertebrates), as has been suggested by previous research (Eskelinen *et al*., 2009). Due to the alteration to community composition, regardless of fertilizer influence, there is an argument that turf transplants may not necessarily constitute restoration, as the communities are no longer identical to the pre-disturbance conditions, but rather a form of rehabilitation. However, longer duration of study would be required to determine if the community alterations stabilize over time, revert to the original community compositions, or develop new community assemblies.

4.6.2 Plant Expansion from Harvested Turfs

Vegetation expansion in both treatments was up to 18 cm in distance and turfs increased ~150% in area (*i.e.* 900 cm^2 bare substrate colonized from 600-cm^2 turf), although constrained by the container size in fertilized treatments, indicating potential increases to distance of expansion. Graminoids were predominantly responsible for expansion, confirming our third hypothesis. This expansion is consistent with previous turf transplantations, with turfs demonstrating an increase in total transplant area of 240% over two years in Iceland (Aradottir, 2012) and 10 to 50-fold increases in mean turf size at Prudhoe Bay, Alaska over seven years (Kidd *et al*., 2006). In contrast, others reported far less expansion in Prudhoe Bay (*i.e.*, 74% over two years) (Shirazi *et al*., 1998) and in disturbed alpine environments (only 77% over four years) (Cole & Spildie, 2006). Reasons for the observed differences in expansion, such as local climate, moisture regimes, rates of fertilization, or turf plant communities, have not been fully elucidated, further underscoring the need for additional studies across the circumpolar Arctic. Results from our growth chamber trial, which used an extended growing period without vernalization, are likely to differ from *in situ* results, particularly regarding time frames of expansion.

4.6.3 Aboveground Responses

Greater development of BSC communities on fertilized substrates was expected based on results of previous fertilizer-addition studies, which found increased moss and BSC development on fertilized soil in upland tundra (Naeth & Wilkinson, 2014), high-arctic glacial forelands (Hodkinson *et al*., 2003), and alpine environments (Gold *et al*., 2001; Gretarsdottir *et al*., 2004).

These studies and others in alpine tundra (Gold *et al*. 2001) and the high-Arctic (Breen & Lévesque, 2006) found increased BSC development was important for soil stabilization, fertility, and moisture, as well as facilitating the development of vascular plant communities through the capture of wind-blown seeds and increased availability of favorable microsites for seed germination. Our study found that transplanted turfs are likely to act as sources of BSC propagules, and adjacent fertilization can decrease the time frames of BSC colonization, found to be ~10 years in high-arctic proglacial chronosequences (Breen & Lévesque, 2006).

Fertilization of surrounding substrates also encouraged the expansion of fast-growing vascular growth forms [*i.e.*, typically graminoids, forbs, and deciduous shrubs (Henry, Freedman & Svoboda, 1986)] into the enriched substrates. The expansion of these labile species versus the expansion of evergreen woody species with recalcitrant tissues may be desirable, due to the direct impact of increased litter and organic residue deposition, and consequent indirect effects including the development of organic soil layers and nutrient cycling regimes. Fertilization did not have a significant effect on the number of vascular species expanding from the turf, similar to previous research demonstrating fertilization did not affect aboveground vascular biomass within Alaskan tundra (Kelley & Epstein, 2009). However, fertilization resulted in a greater prevalence of most of the vascular species that expanded into substrates, indicating that fertilization promoted the expansive potential of those species.

4.6.4 Belowground Responses

We hypothesized that fertilization of the adjacent substrate would increase above and belowground biomass. Belowground biomass was roughly three-fold greater within fertilized substrates, whereas there was no significant difference between fertilized and unfertilized substrates for aboveground biomass of emergents. Therefore, expansion from the turfs was primarily represented by belowground expansion that was influenced by fertilizer addition. Belowground biomass represented 96% and 93% of total biomass exiting the turfs within fertilized and unfertilized substrates, respectively. This ratio is consistent with previous investigations of belowground biomass within arctic tundra, where researchers have estimated 90% of total biomass may be contained within belowground structures (Iversen *et al*., 2015). Higher belowground biomass facilitates the development of SOM from root death, decomposition, and exudation (Iversen *et al*., 2015) and is critical to ecosystem recovery.

Graminoid species were largely responsible for belowground expansion, being detected within 86% and 64% of fertilized and unfertilized cells, respectively. *Anthoxanthum monticola* was the most important, representing over half of all graminoids detected in both treatments. The response of this species is congruent with research on fertilization of dry-heath tundra communities, which found *A. monticola* dominated within N and phosphorus enriched communities (Gough, Wookey & Shaver, 2002). *D. integrifolia* and both *Salix* species each represented roughly half of the shrub species detected within fertilized substrates, however, *D. integrifolia* dominated the expansion of shrubs within unfertilized substrates, representing ~ 95% of all shrubs detected. *D. integrifolia* is known to respond negatively to fertilizer additions through decreased cover and growth (Henry *et al*., 1986). Conversely, *Salix* species are known to respond positively to increased nutrient availability, potentially explaining the significant increase of *Salix* within fertilized treatments, as well as the lessening dominance of *D. integrifolia*.

The forbs, *S. longipes*, *A. alpinus*, and an unidentified *Fabaceae* species had the greatest expansion in both fertilized and unfertilized substrates. *S. longipes* is known as an aggressive colonizer and highly polymorphic species, favorable to have within transplanted turf communities for its rapid expansive potential (Aiken *et al*., 2007). Both N-fixing species, *A. alpinus* and the unidentified *Fabaceae* species, were more prevalent within unfertilized substrates. Fertilization may reduce the expansion of N-fixing species and/or inhibit nodulation and N fixation (Graham, 2005). However, including these N-fixers in harvested turfs may promote expansion into unfertilized substrates and facilitate the growth of other species through N addition (Olsen, Sandvik & Totland, 2013).

Belowground biomass was significantly lower beyond six centimetres from the turf in both fertilized and unfertilized substrates; however, relatively high biomass within fertilized treatments 12 to 18 cm from the turf suggests that fertilization also is likely to increase the distance of expansion if not constrained by the container size. An increased distance of expansion would allow for increased resource acquisition by deep-rooted graminoids as well as shallow-rooted shrub species (Wang *et al*., 2016), and increase the likelihood of developing belowground exchanges (*i.e.*, two-way transfers of energy and organisms, through common mycorrhizal networks) with external systems or other transplants (He, Critchley & Bledsoe, 2003). Increased distance of expansion may allow for greater spacing between transplants or fewer transplants to be used, reducing both the impact on donor sites and treatment costs.

4.6.5 Turf Transplants for Arctic Restoration

In our study, belowground biomass far exceeded the biomass aboveground, therefore assessing restoration success based solely on aboveground characteristics ignores the primary component of turf expansion, potentially leading researchers and/or restoration practitioners to inappropriate conclusions. Investigating belowground expansion is critical in an arctic environment where 90% of vegetative biomass is belowground (Iversen *et al*., 2015) and can provide valuable information and a holistic assessment of whole turf restoration approaches. While fertilization of adjacent substrates did not alter the turf community as seen in direct fertilization experiments (Gough & Hobbie, 2003; Kelley & Epstein, 2009), the greater BSC development and belowground biomass that occurred in fertilized substrates suggests fertilization may decrease timeframes of natural colonization in the turf's immediate surroundings and may support the development of later-stage plant communities (Breen & Lévesque, 2006). Conversely, the reduced *Fabaceae* prevalence within fertilized substrate may indicate inhibited expansion of arctic N-fixing vascular species. Turfs in our growth chamber experiment expanded ~150% in total area, with graminoid species predominately responsible for this expansion, similar to results of turf transplants within Alaskan tundra (Shirazi *et al*., 1998; Kidd *et al*., 2006). Our *ex situ* results will likely differ from real-world conditions, however, ongoing research of *in situ* turf transplants near the turf donor sites will provide further insight into arctic turf expansion under natural conditions within a low-arctic upland-health tundra. There is a significant knowledge gap on how turf transplants will respond in different arctic ecosystems. Moisture regimes, climates, and plant communities will likely have a significant effect on the survivability, growth, and expansion of turfs, necessitating turf transplant studies across arctic ecosystems. Maintenance of soil biota, development of nutrient cycling regimes, identification of species to be targeted, and rates of fertilization, specifically in real-world settings, are needed to optimize whole turf transplanting techniques and increase the use of this technique by restoration practitioners.

5. SYNTHESIS AND CONCLUSIONS

To investigate the efficacy of whole turf transplantation within low-arctic upland heath tundra, we examined the survival of species, alterations to turf community composition, above and belowground vegetative expansion, maintenance of soil nutrient concentrations, and recovery of harvest plots two years post-transplantation. To investigate these aspects of turf transplantation, we completed both an *in situ* and *ex situ* study using turfs harvested from undisturbed tundra located on the Agnico Eagle Mines Meliadine mine site.

The *in situ* study investigated the efficacy of transplanted turfs and shredded tundra material in maintaining vegetative cover, community composition, and species richness post-transplantation into a disused gravel quarry located on the Meliadine mine site. We examined the maintenance of soil nutrient concentrations within the transplanted turfs and investigated the recovery of plant cover within turf harvesting locations. The *ex situ* study investigated the effects of turf-adjacent substrate fertilization on turf and substrate community composition under growth chamber conditions. We investigated the effect of fertilization on the expansion of vegetation and determined the identities of the expanding species.

In both studies, species survival was high, with the only species not surviving transplantation representing less than 0.5% of the relative cover across all transplanted turfs. In both studies, turfs maintained high degrees of vegetative cover and species richness, however, levels by which both were maintained differed. While total vegetative cover and species richness increased slightly in the growth chamber, the field trials demonstrated slight decreases in species richness and significant decreases in vegetative cover. Alterations to turf community composition also differed between the two studies. Under optimal growth chamber conditions, evergreen shrub and moss cover increased, while both growth forms significantly decreased within the field. Under both growth chamber and field conditions, lichen cover decreased, and graminoid cover increased. Shredded turf treatments used in the field study performed poorly in maintaining both vegetative cover and species richness. However, shredded turfs may represent an effective method of transferring non-vascular cover over a wide area, although they require far greater protection from environmental stressors to achieve appreciable development of non-vascular communities. The majority of soil nutrient concentrations changed significantly two years post-transplantation; however, these changes were driven by influences from surrounding and underlying substrates. The minimal vegetative cover within harvesting plots suggests that turfs should only be harvested from areas expected to be disturbed in the future.

Fertilization of substrates surrounding turf transplants increased the development of belowground structures, with roughly three times more belowground biomass compared to unfertilized substrates. Fertilization did not have any detectable effect on turf community composition, but significantly increased the development of biological soil crust communities on the bare substrates. *Ex situ* expansion from the transplanted turfs was predominantly belowground, with over 90% of the biomass exiting the turfs occurring belowground. Roots expanded up to 18 cm from the turfs in both fertilized and unfertilized substrates, with evidence suggesting fertilization may increase the distance of expansion. Species responsible for expansion were mainly graminoids, although forbs and shrubs also contributed. Fertilization of turf-adjacent substrates did not have any detectable effect on belowground species richness; however, several species were found in greater frequencies within fertilized substrates, indicating species specific preference to, or inhibition of growth from, increased nutrient availability. *In situ* expansion from turfs was assessed through aboveground characteristics, and similar to the finding from the growth chamber, most aboveground vegetation was found in the turf's immediate surroundings, *i.e.*, within 6-8 cm from the turf.

5.1 Turf Resilience

The upland health turfs used in this research were resilient to both harvesting and transplantation, as well as environmental stressors, including flooding, drought, and influx of soluble salts. While several turfs were found to significantly decline in species richness and vegetative cover, as a collective, the considerable growth of other turfs largely counteracted the deleterious effects of excess salt concentrations and flooding that led to turf mortality (Chapter 3.6.4). The resilience to unfavorable conditions demonstrated by these upland heath turfs suggests whole turfs will continue to survive and actively grow even when subjected to adverse conditions. As gravel quarries and gravel pads represent a significant proportion of the disturbances from industrial mining, the survival and growth of transplanted turfs suggest this may be an appropriate restoration method for a large proportion of disturbed sites, provided turfs are widely available for transplantation. Waste rock piles, another disturbance feature of mining activity, are also frequently associated with course substrates, high electrical conductivities, elevated metal concentrations, and low organic matter (Miller *et al*., 2021). The resilience of transplanted turfs to these conditions suggests turfs may also be appropriate for the restoration of these common disturbances, further increasing the applicability of transplanting turfs for restoration across many mining-related disturbances.

Based on our study, as well as previous research investigating appropriate sizes of turf transplants, we would recommend the use of turfs ≥ 40 cm^2. Smaller turf sizes are likely to result in reduced plant establishment, particularly of shrub species (Aradottir, 2012). Smaller turfs may also demonstrate reduced resilience to environmental forces due to lower interior:edge ratios. Increasing turf size, while potentially providing greater resilience, is likely to require greater implementation of heavy machinery to cut, transport, and deposit turfs onto disturbed areas (Cater *et al*., 2015). The increased use of machinery would likely increase the costs of restoration efforts, however, Cater *et al*. (2015) has demonstrated that simple mechanical additions to excavators and bucket loaders can easily transport turfs larger than 2.5 m^2. Similar modifications could be made to machinery already present on mining sites, such as the Meliadine site, allowing the use of significantly larger turfs without requiring shipment of additional equipment to site.

Shredded tundra turfs are unlikely to share similar resilience, demonstrating high susceptibility to extreme climatic events and high winds characteristic of low-arctic environments. Modifications to the technique used may improve the establishment of non-vascular communities. Application of shredded tundra materials could be further targeted towards the establishment of early successional non-vascular species by selectively harvesting surface organic layers (*i.e.*, upper ~ 5-10 cm) with higher concentrations of non-vascular propagules. The application of a greater volume of shredded material per area, finer-textured mineral surface soils, and incorporation of shredded material into substrates may promote greater non-vascular cover. Even with improvements to the quality and quantity of shredded material, these layers will likely still require considerable protection (*e.g.*, erosion control mats, windbreaks, etc.) to develop non-vascular communities. Selective harvesting and heavier application rates would result in a larger disturbance footprint at donor sites to gain the desired volume of shredded materials. However, selective harvesting of surface layers rich in non-vascular species would reduce the depth of disturbance and allow donor sites to retain organic soil layers. Recovery of vegetative cover may occur more rapidly on organic soil layers than that observed on bare mineral soil, and therefore may represent less overall disturbance, however, further investigation in the field would be required to confirm this hypothesis.

If restoration of primarily non-vascular communities is sought after by restoration practitioners, the use of shredded materials may be appropriate if the site is well protected. Surface fertilization in addition to the application of shredded material may increase treatment efficacy. Increased non-vascular and BSC development from surface fertilization has been observed in both natural colonization studies (Hodkinson *et al*., 2003), and restoration and/or reclamation studies (Gretarsdottir *et al*., 2004; Naeth & Wilkinson, 2014). However, surface fertilization has also shown the potential to inhibit the development of native N-fixing species (Chapter 4.6.4), and long-term fertilization studies often demonstrate a significant decline in moss species, although this is due to the increased growth of taller-stature vascular species (Kelley & Epstein, 2009). One-time fertilization effects are theorized to largely dissipate after several years of vegetative development (Jorgenson & Joyce, 1994; Cater *et al*., 2015), indicating that single fertilization events may be beneficial to kick-start the development of non-vascular communities, without significantly impacting future successional development.

5.2 Alterations to Community Composition

Significant alterations to turf community composition are frequently reported, with turfs in arctic and alpine environments consistently demonstrating increased cover of graminoid growth forms (Shirazi *et al*., 1998; Conlin & Ebersole, 2001; Bay & Ebersole, 2006; Kidd *et al*., 2006; Aradottir, 2012; Aradottir & Oskarsdottir, 2013; Cole, 2013; Cater *et al*., 2015), and less frequently the loss of non-vascular cover (Shirazi *et al*., 1998; Kidd *et al*., 2006; Aradottir & Oskarsdottir, 2013). While differences regarding growth form responses existed between the growth chamber and field studies, specifically between evergreen shrubs and mosses, these were most likely due to the lack of environmental stressors within growth chambers. Specifically, the consistent watering regime likely promoted the development of moss communities within the growth chamber turfs, while the dry early summer of 2021 may have exacerbated moss decline within the field trial.

Regarding evergreen shrubs, the differing observations between the growth chamber and field trial may be due to analyzing changes in growth forms, as individual species within the evergreen shrub group demonstrated similar responses between studies. Specifically, the increase of evergreen shrubs within the growth chambers were entirely due to the increased absolute cover of *D. integrefolia,* with all other species decreasing in cover. In the field trial, all evergreen species decreased in absolute cover, although *D. integrifolia* demonstrated the smallest reduction at -0.4% less absolute cover. This suggests *D. integrifolia* may represent the best evergreen species to target for transplantation, both for its survival, growth, and the observed expansion in the growth chambers (Chapter 4.6.4).

Both studies demonstrated significant decreases in lichen cover, potentially due to shading by taller vascular species within the growth chamber, along with high winds and partial burial observed in the field trial. Regardless of the forces from neighbouring plants and/or environmental stressors, we were unlikely to observe increased lichen cover in either study due to the characteristically slow growth rate of many lichen species (Abdulmanova & Ektova, 2015; Gauslaa, Alam & Solhaug, 2016), and known difficulties of lichen propagation within growth chambers (Cooper, Smith & Wookey, 2001; Hansen, 2010). Despite the consistent decline of lichen cover within both studies, lichen species still comprised at least 20% of the vegetative cover present within turfs across both experiments. Determining how best to prevent lichen decline within transplanted turfs may be particularly important within arctic environments. Although not as heavily studied as vascular plants, there are more lichen species within northern environments than vascular species, demonstrating a full suite of ecological roles and functionality. This includes land-scape scale N fixation, accelerated mineral weathering, generation of microhabitats, nutrient capture and nutrient-rich litter deposition, and altered surface albedo (Asplund & Wardle, 2017). Lichens act as a significant food resource for a plethora of organisms, ranging from micro and meso-organisms such as nematodes and tardigrades, to mega-fauna such as rodents and ungulates (Asplund & Wardle, 2017). Indeed, lichens are well known as an important food source for caribou, as lichens comprise over 50% of their winter food intake (Joly, Jandt & Klein, 2009; Asplund & Wardle, 2017). Given the growing body of literature promoting the importance of lichens, ensuring lichen survival and growth within restoration treatments should be of particular importance for future arctic restoration studies.

Across both studies, the cover of forb and graminoid species increased. Forb and graminoid species that demonstrated decreased cover represented negligible components of overall community composition (*i.e.*, >0.5% relative cover). Both experiments demonstrated similar responses of individual species except for *A. alpinus* and *O. arctica*, both of which increased in cover within the growth chamber and decreased within the field trial. The similarity of growth form responses between the two studies indicates that upland heath communities may demonstrate relatively consistent responses to transplantation, even when subjected to widely differing environmental conditions, such as the optimal conditions offered by the growth chambers and the extreme weather events observed in the field trial. Overall, both experiments began to demonstrate shifts from shrub-dominated communities to forb and graminoid-dominated communities.

5.3 Future Vegetative Development

If these transplanted communities continue to increase in graminoid and forb cover, previous research suggests the developing vegetative communities may begin to alter the underlying soil conditions to favor the continued development of forbs and graminoids (Eskelinen *et al*., 2009). The greater nutrient availability characteristic of forb and graminoid communities may potentially inhibit future encroachment of evergreen shrubs adapted to low-nutrient conditions, resulting in significantly different vegetative communities to pre-disturbance conditions. However, the observed shift in community composition still represents the re-vegetation of communities entirely composed of native species, presenting a significant ecological advantage over seeding with non-natives or use of fertilizers. While this may not represent true restoration of pre-disturbance or historical conditions, the increased cover of forbs and graminoids, and the limited change in deciduous shrub cover may represent restoration of similar communities to contemporary, climate-change mediated, vegetative communities currently emerging across the low-arctic (*i.e.*, increasing dominance of forbs, graminoids, and deciduous shrubs, over evergreen species) (Elmendorf, Henry, Hollister, Björk, Bjorkman, *et al*., 2012a; Elmendorf, Henry, Hollister, Björk, Boulanger-Lapointe, *et al*., 2012b; Zamin *et al*., 2014; Myers-Smith *et al*., 2019).

The development of forb-graminoid-deciduous shrub communities within turfs may present greater benefits to underlying and adjacent soils than that of evergreen shrub-dominated communities. Forbs, graminoids, and deciduous shrubs all deposit labile, N-rich organic residues on the soil surface, at significantly greater rates than evergreen species (Eskelinen *et al*., 2009; Chu & Grogan, 2010; Sundqvist *et al*., 2014). This allows greater rates of organic layer development within the turfs, as well as kick-starting the development of organic layers immediately surrounding the turfs. Further modifications to soil conditions may include lower C:N ratios, and higher net-N mineralization rates (Eskelinen *et al*., 2009), potentially accelerating rates of primary production and organic litter deposition, and therefore greater SOM incorporation.

Greater graminoid cover may also accelerate rates of expansion from transplanted turfs into their immediate surroundings (Chapter 4.6.4), and therefore develop greater potential for increased connectivity and two-way exchanges of energy and soil organisms, either between turfs and nearby undisturbed areas, or between other transplanted turfs. The significant decline in cover of almost all evergreen species indicates they are unlikely to survive transplantation well, potentially due to damages to root systems and associated fungal networks or the significant change in soil pH. Regardless of the mechanism of evergreen decline, future evergreen development is unlikely without considerable development of organic soil layers, and acidification of underlying and adjacent mineral soils (Hodkinson *et al*., 2003). Both organic layer development and acidification of adjacent or underlying substrates was absent from the turf surroundings, suggesting that evergreen establishment will be significantly hindered during the initial stages of recolonization, potentially for several years, possibly even decades.

Despite the shift to forb- and graminoid-dominated communities, the continued survival and growth of several shrub species may be particularly important for the facilitation of future evergreen regeneration. In particular, the deciduous shrubs *S. arctica, S. reticulata, and V. uligonosum*, all demonstrated survival and expansion in both trials, similar to the evergreen shrubs *D. integrifolia* and *V. vitis-idaea.* The deposition of woody tissues from these species would be primarily responsible for the incorporation of recalcitrant organic material, which has been shown to demonstrate significant alterations to underlying soil conditions, such as high C:N ratios, and lower soil pH─both of which favor the development of soil fungal communities (Eskelinen *et al.*, 2009). Fungal communities within Arctic environments are known to be particularly important to many shrub species for increased nutrient acquisition (Timling *et al*., 2012), with species such as *C. tetragona* observed developing ericoid, endophytic, and ectomycorrhizal associations (Fujimura & Egger, 2012).

Current evidence suggests that fungal communities are far more taxonomically rich than previously assumed, often demonstrating compatibility with a wide range of potential hosts (Fujimura & Egger, 2012; Timling *et al*., 2012). The species richness in fungal communities appears to be predominately affected by soil pH, indicating soil acidification from recalcitrant organic deposition may be critical for future development of characteristic fungal communities, and thus consequent evergreen establishment and regeneration. Future studies investigating alterations to fungal communities, and community development following turf transplantation, would be needed to determine if fungal communities offer facilitative effects to evergreen shrub development. If evergreen species continue to decline in cover within transplanted turfs, potential combinations with individual transplants may assist with evergreen recolonization. Recent evidence suggests several evergreen species can develop root structures from cuttings obtained in undisturbed tundra (Ficko & Naeth, 2021), indicating these cuttings may be used to propagate evergreen species within the established turf transplants.

5.4 Recommendations for Restoration Practitioners

Through this research, several important insights into transplantation of upland heath vegetative communities have been identified. First and foremost, is the recommendation to avoid transplantation of evergreen shrub-dominated communities. Nearly all evergreen species declined considerably in absolute cover, across both studies, except for *D. integrifolia* which was observed to increase in the growth chamber, and marginally decrease in the field trial. In contrast, nearly all forbs and graminoids, along with select deciduous shrubs demonstrated consistent increases in absolute cover, indicating a tendency to survive the transplantation process and subsequently grow, both within the turfs and into the surrounding substrates. As many species were observed both increasing in cover or expanding from the turfs, we have compiled a list of species recommended for transplantation, along with the specific reasons for each species (Table 5.1).

Table 5.1. Species recommended for transplantation, based on alterations to cover, evidence of expansion, or other notable features that would be beneficial within transplanted turfs.

Growth Form	**Species**	**Reason for Recommendation**
Evergreen Shrubs	*Dryas integrifolia*	Minimal decrease in absolute cover, observed expanding on both studies. Predominately responsible for shrub expansion in the growth chamber.
	Vaccinium vitis-idaea	Minimal decreases in absolute cover, observed expanding in the field trial.
Deciduous Shrubs	*Salix arctica*	Regarded as both pioneer colonizer and member of late-stage communities, observed expanding in the field trial.
	Salix reticulata	Increased in absolute cover, observed expanding in both studies.
	Vaccinium uligonosum	Minimal decreases in absolute cover, observed expanding in both studies.
Forbs	*Astragalus alpinus*	Increased in absolute cover and observed expanding in the growth chamber. Nitrogen fixer.
	Bistorta vivipara	Increased in absolute cover and observed expanding in the field trial.
	Cardamine digitata	Increased in absolute cover in both studies, observed expanding in the growth chamber.
	Chamaenerion latifolium	Increased in absolute cover and observed expanding in the growth chamber. One of the few vascular species observed directly colonizing harvest sites.
	Hedysarum alpinum	Increased in absolute cover and observed expanding in the field trial. Nitrogen fixer.
	Stellaria longipes	Increased in cover in the growth chamber. Observed expanding in both studies. Prolific and aggressive volunteer colonizer in the both the field trial, and across untreated gravel quarries.
	Oxytropis arctica	Increased in cover in the growth chamber. Nitrogen fixer.
	Oxytropis maydelliana	Increased in cover in the growth chamber. Nitrogen fixer.
Graminoids	*Anthoxanthum monticola*	Increased in absolute cover and predominately responsible for expansion in the growth chamber.
	Carex bigelowii	Increased in absolute cover and observed expanding in the field trial.
	Carex rupenstris	Increased in absolute cover and observed expanding in the field trial.
	Eriophorum callitrix	Observed expanding in growth chamber.
Lichens*	*Gowardia nigricans*	The least reduced lichen species within the field trial
Mosses	*Aulacomnium turgidum*	Increased in absolute cover and observed colonizing surrounding substrate in both studies.
	Byrum bindii	Increased in absolute cover in the growth chamber. Observed colonizing surrounding substrate in both studies.
	Pohlia nutans	Increased in absolute cover in the growth chamber. Observed colonizing surrounding substrate in both studies.
	Bucklandellia macrocarpa	Increased in absolute cover in the field trial.

* All lichen species declined in cover, although provide multiple ecological services and functions within arctic environments. Transfer of lichens would be recommended, although similar to evergreen shrubs, are likely to demonstrate decreased cover post-transplantation.

Regardless of the community transplanted, we would also recommend the transfer of turfs ≥ 40 cm^2. Greater turf sizes, if possible, are recommended not only for reductions in interior:edge ratios which would provide greater insulation from external influences, but also for the increased likelihood of harvesting a wider range of vascular plants, including rare species (Aradottir, 2012), and a wider range of soil organisms. Larger turfs are also likely to present a larger transfer of common mycorrhizal networks, potentially providing individuals within the turf greater access to soil nutrients. Along with turf sizes, turfs with significant organic layer depth would be preferable, not only for maintaining a source of continual decomposition and subsequent nutrient release but for maintaining thermal insulation properties (Shirazi *et al*., 1998). Due to the minimal vegetative recovery of turf harvesting locations, we strongly recommend turfs are harvested only from areas expected to be disturbed, otherwise turf harvesting would represent additional, unnecessary disturbance (Chapter 3.6.6).

The observed turf mortality, either from flooding or excess salt influx presents a considerable challenge for restoration practitioners. However, both may be overcome by effective management of local topography, *i.e.*, the creation of hummock-hollow microtopography to develop multiple niches and favorable microsites (Miller *et al*., 2021) and creating drainage systems to manage floodwater. The hummock-hollow complexes created for the *in situ* study were linear features designed for data collection, and as such did not accurately represent the relatively disorganized development of local microtopography. Therefore the creation of intentionally disorganized microtopographic features are likely to result in greater availability of favorable microsites within practical restoration initiatives. Several authors have also recommended the amelioration of local conditions to promote vegetative success. Naeth and Wilkinson (2014) recommended the use of organic matter deposition to increase germination and plant establishment. Sewage sludge, commonly available on large mine sites, was found to be as effective as salvaged soil in promoting seedling germination and significantly lowered both soil pH and electrical conductivity five years after deposition. Sewage in combination with turf transplants has yet to be investigated but may provide greater amelioration of soil conditions in addition to nutrients.

5.5 Future Directions

Both studies contained significant limitations, with the optimal conditions within growth chambers potentially resulting in unrealistic alterations to vegetative communities and subsequent expansion from the turfs. However, the frequent similarities between the growth chamber and field trial, in terms of community alteration and aboveground expansion, give greater confidence in the results of Chapter 4 applying to real-world conditions. Regarding the *in situ* restoration trial, the major limitation of the experiment is study duration. Vegetative development and successional advancement in arctic environments is well-known to be a lengthy process, with proglacial sequences in the high-Arctic needing decades to centuries before the establishment of late-stage vegetative communities is achieved (Hodkinson *et al*., 2003; Jones & Henry, 2003; Breen & Lévesque, 2006). Previous research has found significant differences in plant responses and treatment efficacy between short-term and long-term observations (Gretarsdottir *et al*., 2004; Cole, 2013; Naeth & Wilkinson, 2014). This suggests that the results obtained after two years are unlikely to reveal the full extent of community alterations, species survival, nor turf expansion. Significantly longer timeframes are needed to conclusively determine the efficacy of turf transplants in restoring disturbed low-arctic sites. Drilling waste sites nearby the *in situ* restoration trial have been observed developing similar vegetative communities to nearby undisturbed tundra after ~20 to 25 years. However, recovery of these small-scale disturbances likely benefits from their small disturbance areas embedded within a matrix of undisturbed tundra, providing opportunity for natural recolonization. Similar rates of recovery are unlikely to carry over to larger disturbances, further underscoring the need for long-term investigation of disturbances such as the gravel quarries.

Further avenues of research, aside from longer study durations, directly relate to alterations of turf transplantation methods. The assumption that larger turfs may present greater resilience to external forces could be tested similarly to Aradottir (2012), analyzing the survival of species and growth forms within progressively larger turfs. Selective soil sampling at progressively greater distances from the center of turfs may allow for the determination of the effective distance of influence from external soils (*i.e.*, the degree of change in pH, moisture content, or soil temperatures from the edge of the turf to the center). Greater characterization of root expansion from transplanted turfs, specifically *in situ*, is likely to provide greater information regarding which species are responsible for expansion and to what distances, further elucidating aspects such as optimal turf spacing. Lastly, turf transplants combined with soil amelioration or fertilization should be conducted *in situ* to test the validity of findings regarding the effect of fertilization on turf and substrate community composition (Chapter 4.5.1).

Further research into changing nutrient availability, particularly into alterations to N cycling or N mineralization rates, may assist researchers in determining future community composition changes within turfs. Similarly, investigations of soil organisms including bacteria, fungi, and invertebrates, and how these communities may change post-transplantation, may assist researchers in determining the most appropriate vegetative communities, and associated soil resources to transplant, *i.e.*, pH, SOM content, C:N ratios, or N and/or P availability (Fujimura & Egger, 2012; Timling *et al*., 2012). Lastly, turf transplants also need to be conducted across a wider variety of vegetative communities, as large discrepancies between rates of vegetative development and vegetative expansion exist between published literature on turf transplants within arctic and alpine environments (Chapter 4.6.2). The exact reasons for these discrepancies, whether due to local climates, soil conditions or vegetative communities, are still largely unknown, necessitating the need for further investigation.

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Appendix A CHAPTER 3 SUPPLEMENTAL INFORMATION

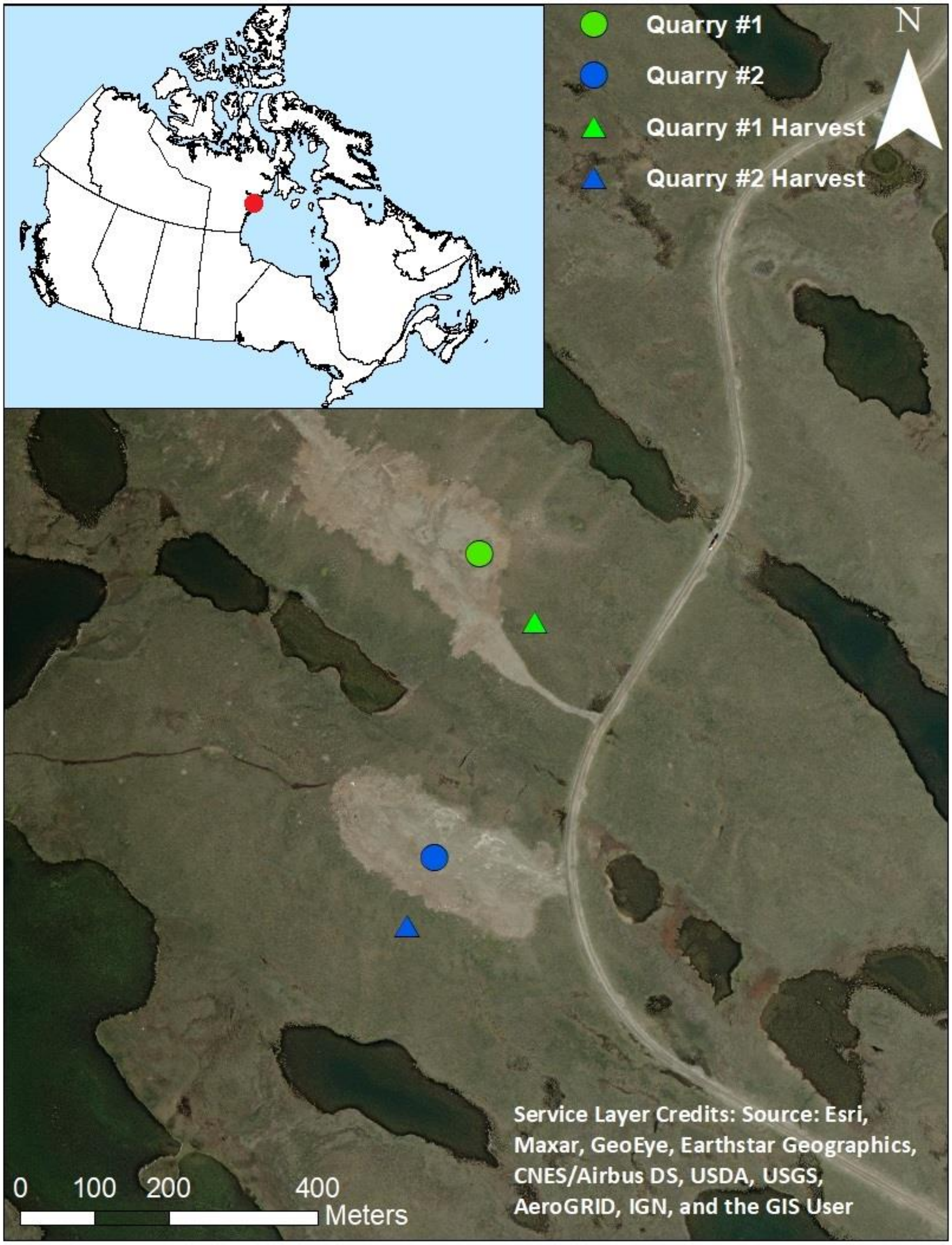

Figure A.1. Location of the two restoration trials established in 2019, and associated harvesting/reference systems. Study sites are located near the coast of Hudson Bay, approximately 25 km northwest of Rankin Inlet, Nunavut, Canada (red circle).

Table A.1 Depth values of organic and mineral layers across all turfs used at each restoration trial, and soil particle size fractions expressed as a percentage of weight, with contribution of soil separates expressed in parentheses. Soil separate fractions were analyzed using a Horiba LA950 Particle Size Analyzer (Japan). Depth of organic and mineral layer values are means with standard deviation (n= 20 turfs per site).

Site	**Organic Depth (cm)**	**Mineral Depth (cm)**	**Clay**	**Silt**	**Sand**	**Gravel**
Q1	7.0 ± 4.2	3.08 ± 3.52	0% (0%)	9% (12%)	64% (88%)	27%
Q2	9.1 ± 3.2	1.46 ± 2.10	0% (0%)	8% (16%)	45% (84%)	47%

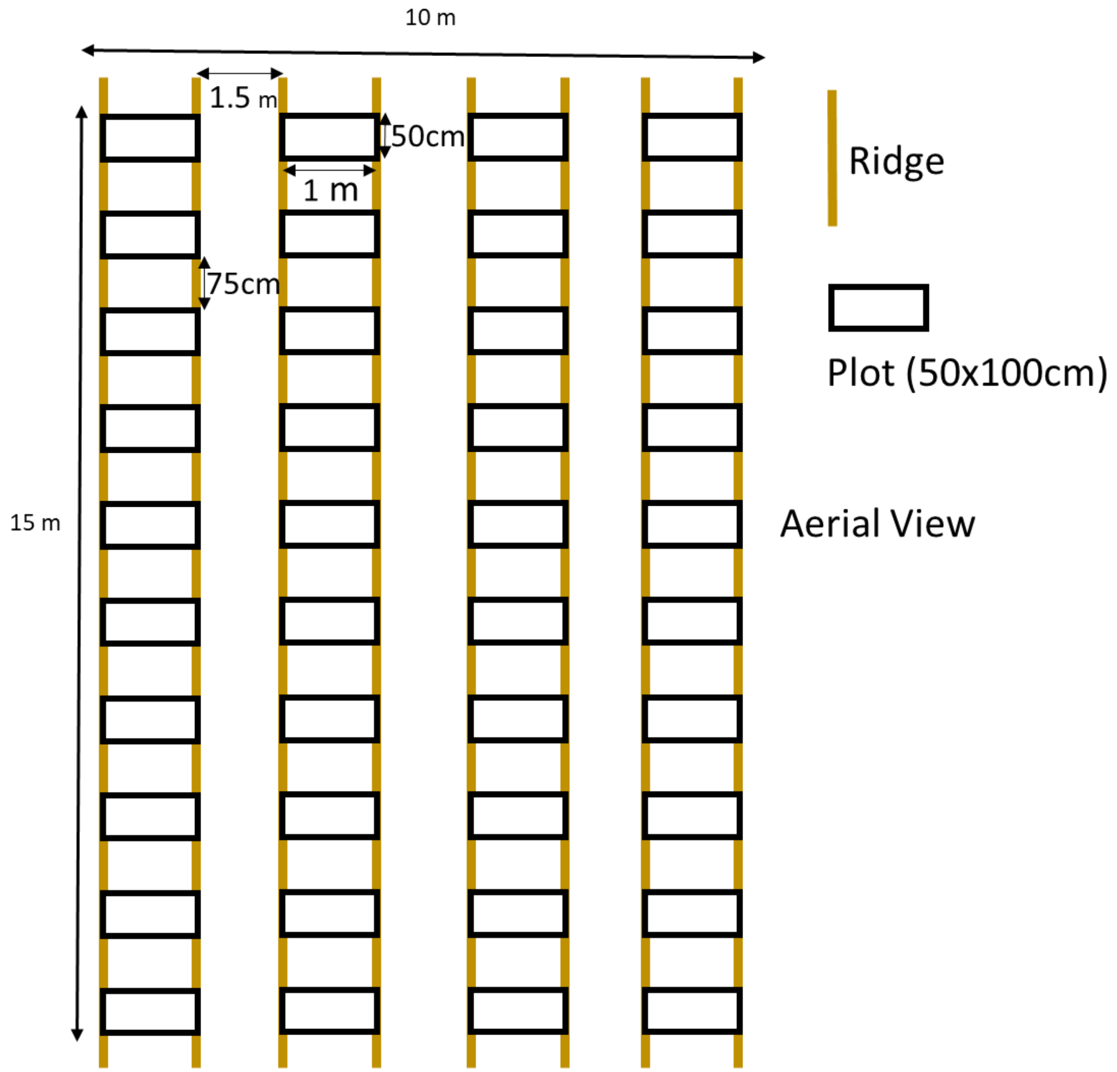

Figure A.2. Conceptual diagram of the restoration trials created at Q1 and Q2. Each row contained ten plots (50 x 100 cm), separated by 0.75 m, for a total of 40 plots per site. Rows were separated by 1.5 m, with each hummock roughly 50 cm high. Treatments were placed in a randomized complete block design,

Table A.2. Description, dimensions (length, width, depth) and diagram of the four treatments applied within the hollow-hummock plot.

Treatment	Description	Dimensions	Diagram
Turf (T)	Intact vegetative sod placed in bottom (center) of hollow	40 x 40 x ~10-15 cm	
Shredded (S)	Vegetative sods shredded and homogenized, spread over the entire plot	3000 cm^3 shredded material spread across plot at 2 cm depth	
Turfs + Shredded (TS)	Intact vegetative sod placed in bottom (center) of hollow and shredded material spread on hummock sides	40 x 40 x ~10-15 cm and 1800 cm^3 shredded material spread at a 2 cm depth	
Control (C)	Control plot	No material added	

Table A.3. Results of Permutational Multivariate Analysis of Variation (PerMANOVA) analysis on all treatment's (Control, Turfs, Turfs + Shredded, and Shredded) community compositions following two years after treatment implementation, using block nested within site as random factors. Boldface indicates significance (p=<0.05).

	Variable	**df**	**Sum of Squares**	**Mean Squares**	**Pseudo-F**	**R^2**	**p-value**
Treatment Community Composition	Treatment	3	1.75	0.58	19.5	0.43	**0.001**
	Residuals	76	2.28	0.03		0.57	
	Total	79	4.03			1.00	

Table A.4. Summary output of linear mixed effect models including all treatments (test as a function of treatment, with block as a random factor) on species richness and cover of bare ground, vegetation, litter, and functional vegetative groups. Cover of deciduous and evergreen shrubs, forbs, graminoids, lichens and mosses were log transformed and vegetative cover square root transformed to meet statistical assumptions. Block was removed from the random error structure of biological soil crust (BSC) cover, due to singular fit errors. NumDf represents the numerator degrees of freedom, and DenDf represents the denominator degrees of freedom. Boldface indicates significance (p<0.05).

Cover/Variable	**NumDf**	**DenDF**	**Sum of Squares**	**Mean Square**	**F-statistic**	**p-value**
Species Richness	3	57	4890	1630	80.84	**<0.001**
Bare ground	3	57	68903	22968	12.43	**<0.001**
Total Vegetation	3	57	1698	566.1	143.3	**<0.001**
Litter	3	57	354120	115040	117.48	**<0.001**
Deciduous Shrubs	3	57	221.4	73.81	115.3	**<0.001**
Evergreen Shrubs	3	57	192.6	64.19	118.6	**<0.001**
Forbs	3	57	36.67	12.22	21.04	**<0.001**
Graminoids	3	57	77.19	25.73	38.64	**<0.001**
BSCs	3	N/A	0.70	0.23	1.69	0.177
Lichens	3	57	124.5	41.50	59.93	**<0.001**
Mosses	3	57	56.79	18.93	21.96	**<0.001**

Table A.5. Means, standard deviations and significant difference indicators for all treatments on presence of flowers (*i.e.*, number of quadrat cells containing flowering structures, per species). Significance groups determined through post-hoc comparisons of linear mixed effect models using block as a random factor.

		Treatments		
Variable	**Turfs**	**Turfs + Shredded**	**Shredded**	**Control**
Flower Density	13.8 ± 11.1 ^{a}	11.2 ± 16.3 ^{a}	1.2 ± 2.95 ^{b}	1.75 ± 4.76 ^{b}

Table A.6. Summary output of linear mixed effect models including all treatments (test as a function of treatment, with block as a random factor) on flowering density. Flowering data was inverse transformed to meet statistical assumptions. NumDf represents the numerator degrees of freedom, and DenDf represents the denominator degrees of freedom. Boldface indicates significance (p<0.05).

Variable	**NumDf**	**DenDF**	**Sum of Squares**	**Mean Square**	**F-statistic**	**p-value**
Flower Density	3	57	7.06	2.35	26.05	**<0.001**

Table A.7. Means, standard deviations and significant differences on species richness, and cover of bare ground, vegetation, litter, and functional vegetative groups for shredded treatments and shredded layers within turf + shredded treatments only. The term “bryophytes” was used in lieu of mosses, as all moss fragments were labelled as bryophytes during 2019. Significance groups determined through post-hoc comparisons of linear mixed effect models, using subject ID as a random factor.

Cover/Variable	**2019**	**2021**
Species Richness	11.6 ± 2.27 ^{a}	9.18 ± 4.88 ^{b}
Bare ground	96.0 ± 30.9 ^{b}	206 ± 51.8 ^{a}
Total Vegetation	97.6 ± 39.0 ^{a}	42.5 ± 29.3 ^{b}
Litter	220 ± 60.7 ^{a}	128 ± 51.5 ^{b}
Deciduous Shrub	7.5 ± 4.89 ^{a}	1.42 ± 3.31 ^{b}
Evergreen Shrub	15.6 ± 8.07 ^{a}	1.5 ± 3.18^{b}
Forb	0.40 ± 0.74 ^{b}	7.35 ± 8.63 ^{a}
Graminoid	3.05 ± 3.71 ^{a}	2.88 ± 2.67 ^{a}
Bryophyte	17.2 ± 13.5 ^{a}	6.3 ± 8.86 ^{b}
Lichen	53.8 ± 22.3 ^{a}	23.0 ± 18.5 ^{b}

Table A.8. Summary output of linear mixed effect models (test as a function of year, with subject ID as a random factor) on species richness, and cover of bare ground, vegetation, litter, and functional vegetative groups within shredded treatments and shredded layers within turf + shredded treatments only. Cover of evergreen shrubs, deciduous shrubs, and bryophytes were log transformed, and forb cover inverse transformed to meet statistical assumptions. Subject ID was removed from the random error structure of graminoid cover, due to singular fit errors. NumDf represents the numerator degrees of freedom, and DenDf represents the denominator degrees of freedom. Boldface indicates significance (p<0.05).

Cover/Variable	**dfNum**	**dfDen**	**Sum of Squares**	**Mean Square**	**F-statistic**	**p-value**
Species Richness	1	39	117.61	117.61	9.91	**0.003**
Bare ground	1	39	239915	239915	167.1	**<0.001**
Total Vegetation	1	39	60720	60720	72.68	**<0.001**
Litter	1	39	172051	172051	226.96	**<0.001**
Deciduous Shrub	1	39	44.56	44.56	86.29	**<0.001**
Evergreen Shrub	1	39	89.03	89.03	167.69	**<0.001**
Forb	1	39	5.70	5.70	64.60	**<0.001**
Graminoid	1	N/A	0.61	0.61	0.06	0.810
Bryophyte	1	39	25.17	25.17	57.58	**<0.001**
Lichen	1	39	18911	18911	70.92	**<0.001**

Table A.9. Permutational Multivariate Analysis of Variation (PerMANOVA) analysis on community composition of turfs only (including turfs of turf and turf + shredded treatments) two years after treatment implementation, using a unique subject ID for each plot as a random factor. Boldface indicates significance ($p<0.05$).

	Variable	**df**	**Sum of Squares**	**Mean Squares**	**Pseudo-F**	**R^2**	**p-value**
Plot	Year	1	2.00	2.00	14.97	0.16	**0.001**
Community	Residuals	76	10.44	0.13		0.83	
Composition	Total	79	12.44			1.00	

Table A.10. Means, standard deviations and significant differences on species richness, and cover of bare ground, vegetation, litter, and functional vegetative groups for Turfs only (including Turfs of Turf and Turf + shredded treatments). Significance groups determined through post-hoc comparisons of linear mixed effect models, using subject ID as a random factor.

Cover/Variable	**2019**	**2021**
Species Richness	19.6 ± 3.4 ^{a}	18.0 ± 4.9 ^{a}
Bare ground	15.8 ± 11.4 ^{b}	28.8 ± 18.9 ^{a}
Total Vegetation	180 ± 24.7 ^{a}	153 ± 54.1 ^{b}
Litter	11.3 ± 7.5 ^{b}	47.6 ± 17.5 ^{a}
Deciduous Shrub	34.5 ± 12.9 ^{a}	34.0 ± 18.9 ^{a}
Evergreen Shrub	45.6 ± 14.4 ^{a}	31.4 ± 19.4 ^{b}
Forb	7.02 ± 7.27 ^{b}	12.9 ± 12.1 ^{a}
Graminoid	8.8 ± 8.8 ^{b}	14.3 ± 11.4 ^{a}
BSC	0.9 ± 1.9 ^{a}	0.28 ± 0.78 ^{a}
Lichen	66.2 ± 24.9 ^{a}	47.0 ± 31.6 ^{b}
Moss	16.6 ± 13.6 ^{a}	13.4 ± 12.9 ^{b}

Table A.11. Summary output of linear mixed effect models (test as a function of treatment, with subject ID as a random factor) on species richness, and cover of bare ground, vegetation, litter, and functional vegetative groups for turfs only (including turfs of Turf and Turf + shredded treatments). Cover of bare ground, litter, BSC, forbs and graminoids were square root transformed and total vegetation log transformed to meet statistical assumptions. Subject ID was removed from the BSC cover random error structure due to singular fit errors. NumDf represents the numerator degrees of freedom, and DenDf represents the denominator degrees of freedom. Boldface indicates significance (p<0.05).

Cover/Variable	**dfNum**	**dfDen**	**Sum of Sq**	**Mean Sq**	**F-statistic**	**p-value**
Species Richness	1	39	51.2	51.2	3.87	0.056
Bare ground	1	39	35.98	35.98	27.87	**<0.001**
Total Vegetation	1	39	1.09	1.09	12.63	**<0.001**
Litter	1	39	248.6	248.6	202.28	**<0.001**
Deciduous Shrub	1	39	5.00	5.00	0.05	0.832
Evergreen Shrub	1	39	4047	4047	31.04	**<0.001**
Forb	1	39	14.03	14.03	14.90	**<0.001**
Graminoid	1	39	10.90	10.90	24.22	**<0.001**
BSC	1	N/A	0.56	0.56	3.24	0.076
Lichen	1	39	7334	7334	32.96	**<0.001**
Moss	1	39	201.61	201.61	7.04	**0.012**

Table A.12. Means, standard deviations and significant difference indicators for both sites and both years on species richness, and cover of bare ground, vegetation, litter, and functional vegetative groups for Turfs only (including turfs of turf and turf + shredded treatments). Significance groups determined through post-hoc comparisons of linear mixed effect models, using Subject ID as a random factor.

Year	**2019**		**2021**	
Site	**Quarry 1**	**Quarry 2**	**Quarry 1**	**Quarry 2**
Species Richness	18.4 ± 3.9 ^{a}	20.7 ± 2.4 ^{a}	17.3 ± 5.9 ^{a}	18.6 ± 3.6 ^{a}
Bare ground	16.0 ± 12.7 ^{bc}	15.5 ± 10.3 ^{c}	31.0 ± 21.1 ^{a}	26.6 ± 16.7 ^{ab}
Total Vegetation	175 ± 26.0 ^{a}	184 ± 23.1 ^{a}	145 ± 67.4 ^{b}	161 ± 36.5 ^{ab}
Litter	8.8 ± 6.5 ^{b}	13.8 ± 7.7 ^{b}	47.2 ± 21.5 ^{a}	47.9 ± 12.9 ^{a}
Deciduous Shrub	27.8 ± 11.9 ^{b}	41.2 ± 10.3 ^{a}	23.3 ± 18.3 ^{b}	44.7 ± 12.4 ^{a}
Evergreen Shrub	43.8 ± 14.9 ^{ab}	47.5 ± 14.0 ^{a}	28.0 ± 21.3 ^{c}	34.8 ± 17.1 ^{bc}
Forb	4.1 ± 5.1 ^{b}	10.0 ± 8.0 ^{ab}	11.4 ± 13.8 ^{a}	14.3 ± 10.3 ^{a}
Graminoid	8.2 ± 7.9 ^{b}	9.4 ± 7.9 ^{ab}	16.0 ± 13.7 ^{a}	12.6 ± 8.36 ^{ab}
BSC	1.1 ± 2.3 ^{a}	0.7 ± 1.4 ^{a}	1.35 ± 2.6 ^{a}	0.9 ± 1.68 ^{a}
Lichen	78.5 ± 24.5 ^{a}	53.8 ± 18.7 ^{b}	59.6± 30.9 ^{b}	34.4 ± 27.4 ^{c}
Moss	11.8 ± 11.3 ^{a}	21.5 ± 14.2 ^{a}	6.4 ± 9.8 ^{b}	20.5 ± 11.9 ^{a}

Table A.13. Summary output of linear mixed effect models (test as a function of year, site and their interaction, using Subject ID as a random factor) on species richness, and cover of bare ground, litter, and functional vegetative groups for turfs only (including turfs of turf and turf + shredded treatments). Cover of bare ground, litter, BSC, forbs and graminoids were square root transformed and total vegetation log transformed to meet statistical assumptions. Subject ID was removed from the BSC cover random error structure due to singular fit errors. dfNum indicates degrees of freedom numerator, dfDen indicates degrees of freedom denominator. Sum of Sq indicates sum of squares, Mean sq indicates the mean squares. Boldface indicates significant differences (p<0.05).

Cover/Variable	Factor	dfNum	dfDen	Sum of Squares	Mean Square	F-statistic	p-value
Species Richness	Year	1	38	51.20	51.20	3.80	0.059
	Site	1	38	41.86	41.86	3.11	0.086
	Year:Site	1	38	4.05	4.05	0.30	0.587
Bare ground	Year	1	38	35.98	35.98	27.64	**<0.001**
	Site	1	38	0.19	0.19	0.15	0.705
	Year:Site	1	38	0.87	0.87	0.67	0.418
Total Vegetation	Year	1	38	1.09	1.09	13.48	**<0.001**
	Site	1	38	0.21	0.21	2.59	0.116
	Year:Site	1	38	0.15	0.15	1.84	0.184
Litter	Year	1	38	248.56	248.56	205.11	**<0.001**
	Site	1	38	2.81	2.81	2.32	0.136
	Year:Site	1	38	1.87	1.87	1.55	0.221
Deciduous Shrub	Year	1	38	5.00	5.00	0.05	0.827
	Site	1	38	2360.07	2360.07	22.77	**<0.001**
	Year:Site	1	38	328.05	328.05	3.17	0.083
Evergreen Shrub	Year	1	38	4047.00	4047.00	30.54	**<0.001**
	Site	1	38	163.80	163.80	1.24	0.273
	Year:Site	1	38	49.60	49.60	0.37	0.544
Forb	Year	1	38	14.03	14.03	15.07	**<0.001**
	Site	1	38	4.23	4.23	4.54	**0.040**
	Year:Site	1	38	1.36	1.36	1.47	0.233
Graminoid	Year	1	38	10.90	10.90	25.04	**<0.001**
	Site	1	38	0.01	0.01	0.01	0.930
	Year:Site	1	38	1.01	1.01	2.32	0.136
BSC	Year	1	N/A	0.56	0.56	3.19	0.078
	Site	1	N/A	0.18	0.18	1.00	0.321
	Year:Site	1	N/A	0.01	0.01	0.04	0.827
Lichen	Year	1	38	7334.40	7334.40	32.12	**<0.001**
	Site	1	38	2797.80	2797.80	11.37	**0.002**
	Year:Site	1	38	1.80	1.80	0.01	0.930
Moss	Year	1	38	201.61	201.61	7.49	**0.009**
	Site	1	38	299.30	299.30	11.13	**0.002**
	Year:Site	1	38	94.61	94.61	3.52	0.068

Table A.14. Summary output of linear mixed effect models (test as a function of treatment, distance and the interaction, with block as random factors) on vascular and non-vascular cover expanding from the central plot of all treatments. Both vascular and non-vascular cover were log-transformed to meet statistical assumptions. NumDf represents the numerator degrees of freedom, and DenDf represents the denominator degrees of freedom. Boldface indicates significance (p<0.05).

Cover	Variable	dfNum	dfDen	Sum of Squares	Mean Square	F-statistic	p-value
Vascular	Treatment	3	209	28.16	9.39	20.79	**<0.001**
	Distance	2	209	37.47	18.74	41.50	**<0.001**
	Treatment: Distance	6	209	20.79	3.47	7.67	**<0.001**
Non-vascular	Treatment	3	209	11.25	3.75	10.60	**<0.001**
	Distance	2	209	11.66	5.83	16.50	**<0.001**
	Treatment: Distance	6	209	2.22	0.37	1.05	0.397

Table A.15. Summary output of linear models (test as a function of distance) on bare ground, litter, vascular and non-vascular cover and non-vascular cover expanding from the central plot of turf and turf + shredded treatments. Litter cover was log transformed and vascular and non-vascular cover was square root transformed to meet statistical assumptions. Boldface indicates significance (p<0.05).

Cover	Variable	df	Sum of Squares	Mean Squares	F-Statistic	p-value
Bare ground	Distance	2	465.62	232.81	10.25	**<0.001**
	Residuals	117	2657.85	22.72		
Litter	Distance	2	18.73	9.36	11.00	**<0.001**
	Residuals	117	99.61	0.85		
Vascular	Distance	2	56.29	28.15	32.36	**<0.001**
	Residuals	117	101.77	0.87		
Non-vascular	Distance	2	11.04	5.52	8.41	**<0.001**
	Residuals	117	76.83	0.66		

Table A.16. Mean relative cover and standard deviation of functional vegetative groups across all harvest plots (n = 46) in 2021. Relative cover was calculated as: cover of functional group/total cover of all functional groups.

Cover	Relative cover and standard deviation
Deciduous Shrub	11.7% ± 5.9%
Evergreen Shrub	14.3% ± 7.5%
Forb	9.4% ± 5.6%
Graminoid	6.0% ± 5.0%
BSC	0.2% ± 1.4%
Lichen	28.9% ± 12.1%
Moss	10.7% ± 9.8%

Table A.17. Means, standard deviations, and significance indicators for pH, and concentrations of chloride, sulfate, nitrate, sodium, ammonium, potassium, magnesium, calcium, total organic nitrogen, total inorganic nitrogen, total organic carbon, and total inorganic carbon within the turf harvesting locations (in 2019 and 2021), turfs within turf and turf + shredded center plots, and substrates used at each site, separated by soil layer and site. TON, TOC, and TIC are presented as g kg^{-1}, with all other values aside from pH presented as mg kg^{-1}. Significance indicators are presented as differences between harvesting locations and turfs, within each soil layer, and within each site. Unless stated within parentheses, sample numbers were n=10 for all harvesting sites (except for the mineral layer in Quarry #2, where only 9 samples were obtained in 2019), and n=20 for all turfs in 2021. Total organic and inorganic carbon each contained n=5 samples for each harvest site and turf. Only one composite sample of Quarry #1's substrate was obtained in 2019, and two composite samples for Quarry #2, due to different materials used to create the hummock-hollow microtopography.

Soil Layer	**Site**	**Treatment**	**pH**	**Cl$^-$**	**SO_4^{2-}**	**NO_3^-**	**Na^+**	**NH_4^+**	**K^+**	**Mg^{2+}**	**Ca^{2+}**	**TON**	**TIN**	**TOC**	**TIC**
Organic	Q1	H-19	5.8 ± 0.4 ^{ab}	35 ± 22.^{b}	86 ± 93^{a}	2.5 ± 1.4 (9) ^{ab}	23 ± 12 ^{b}	4.2 ± 4.0 ^{a}	27 ± 13 ^{a}	15 ± 12^{a}	77± 73^{b}	11 ± 7.3 ^{a}	6.5 ± 3.9 ^{a}	171 ± 52 ^{a}	1.3 ± 0.67 ^{a}
		H-21	5.3 ± 1.0 ^{b}	52 ± 42 ^{b}	100 ± 114 ^{a}	0.29 ± 0.59 ^{b}	13 ± 11 ^{b}	9.2 ± 9.2 ^{a}	63 ± 43 ^{a}	16 ± 13 ^{a}	108 ± 85 ^{ab}	12 ± 7.2 ^{a}	9.5 ± 9.2 ^{a}	254 ± 112 ^{a}	1.8 ± 1.5 ^{a}
		T-21	6.3 ± 0.5 ^{a}	813 ± 1031 ^{a}	430 ± 478 ^{a}	15 ± 20 ^{a}	439 ± 670 ^{a}	6.7 ± 5.5 (16) ^{a}	52 ± 36 ^{a}	58 ± 91 ^{a}	291 ± 280 ^{a}	4.9 ± 2.6 ^{b}	27 ± 31 ^{a}	123 ± 65 ^{a}	1.2 ± 0.49 ^{a}
	Q2	H-19	6.2 ± 0.3 ^{b}	32 ± 15 ^{b}	53 ± 31 ^{b}	6.1 ± 2.8 (9) ^{a}	24 ± 7.4 ^{b}	4.9 ± 9.4 ^{b}	57 ± 42 ^{a}	24 ± 11 ^{a}	152 ± 65 ^{b}	15 ± 6.3 ^{a}	11 ± 11 ^{b}	250 ± 38 ^{a}	2.6 ± 0.55 ^{a}
		H-21	6.4 ± 0.3 ^{ab}	53 ± 40 ^{b}	45 ± 16 ^{b}	3.1 ± 4.1 ^{b}	35 ± 71 ^{b}	18 ± 11 (8) ^{a}	51 ± 18 ^{a}	12 ± 4.2 ^{b}	119 ± 32 ^{b}	17 ± 5.2 ^{a}	18 ± 11 ^{ab}	334 ± 73 ^{a}	3.0 ± 1.2 ^{a}
		T-21	6.5 ± 0.3 ^{a}	151 ± 215 ^{a}	492 ± 280 ^{a}	0.38 ± 1.2 ^{c}	152 ± 124 ^{a}	20 ± 8.7 ^{a}	69 ± 28 ^{a}	30 ± 15 ^{a}	245 ± 141 ^{a}	13 ± 3.5 ^{a}	21 ± 8.5 ^{a}	279 ± 70 ^{a}	2.8 ± 0.84 ^{a}
Mineral	Q1	H-19	6.5 ± 0.4 ^{b}	4.2 ± 1.6 ^{b}	11 ± 11 ^{b}	0.61 ± 0.30 (7) ^{b}	5.5 ± 1.7 ^{b}	0.27 ± 0.57 ^{b}	0.69 ± 0.94 ^{b}	3.7 ± 2.0 ^{b}	22 ± 12 ^{b}	0.70 ± 0.36 ^{a}	0.70 ± 0.63 ^{b}	11 ± 5.3 ^{a}	0 ± 0 ^{b}
		H-21	5.4 ± 0.6 ^{c}	6.5 ± 6.7 ^{b}	14 ± 13 ^{b}	0.41 ± 1.3 ^{b}	0.30 ± 0.81 ^{c}	1.4 ± 1.2 ^{a}	5.2 ± 7.6 ^{b}	2.5 ± 1.2 ^{b}	21 ± 13 ^{b}	1.6 ± 1.4 ^{a}	1.8 ± 2.2 ^{b}	16 ± 18 ^{a}	0.12 ± 0.27 ^{b}
		T-21	7.2 ± 0.4 ^{a}	156 ± 113 ^{a}	143 ± 93 ^{a}	8.8 ± 6.6 ^{a}	101 ± 83 ^{a}	0.31 ± 0.09 (3) ^{ab}	9.3 ± 4.6 ^{a}	13 ± 8.2 (19) ^{a}	82 ± 34 (19) ^{a}	0.8 ± 0.4 ^{a}	8.9 ± 6.6 ^{a}	14 ± 8.5 ^{a}	1.5 ± 0.61 ^{a}
		Q1	7.0	10	20	17	10	0	1.6	5.1	41	0.95	17	17	0.90
	Q2	H-19	6.8 ± 0.2 ^{b}	2.9 ± 2.1 ^{c}	10 ± 4.8 ^{b}	1.3 ± 1.1 (5) ^{a}	8.1 ± 1.5 ^{b}	0.02 ± 0.05 ^{c}	0.25 ± 0.74 ^{b}	4.9 ± 1.7 ^{ab}	36 ± 11 ^{b}	0.83 ± 0.43 ^{ab}	0.73 ± 1.1 ^{b}	13 ± 7.4 ^{b}	0.10 ± 0.22 ^{b}
		H-21	6.2 ± 0.4 ^{c}	8.1 ± 6.5 ^{b}	11 ± 7.9 ^{b}	0.99 ± 1.7 ^{a}	1.1 ± 2.2 ^{c}	2.3 ± 2.0 (6) ^{a}	4.6 ± 6.9 ^{ab}	2.7 ± 1.4 ^{b}	30 ± 15 ^{b}	0.17 ± 0.15 ^{a}	2.4 ± 2.9 ^{ab}	44 ± 30 ^{a}	0.48 ± 0.48 ^{b}
		T-21	7.7 ± 0.2 ^{a}	26 ± 27 ^{a}	111 ± 125 ^{a}	3.3 ± 3.5 ^{a}	34 ± 25 ^{a}	0.72 ± 0.49 (17) ^{b}	9.9 ± 9.2 ^{a}	8.1 ± 4.4 ^{a}	89 ± 60 ^{a}	0.51 ± 0.32 ^{b}	3.9 ± 3.5 ^{a}	8.0 ± 3.1 ^{b}	1.8 ± 0.48 ^{a}
		Q2	7.6 ± 0.4	19 ± 27	111 ± 149	4.8 ± 3.5	26 ± 33	0.80 ± 1.1	0 ± 0	9.0 ± 5.6	79 ± 60	0.23 ± 0.32	2.0 ± 2.8	4.7 ± 1.3	1.8 ± 1.6

Table A.18. Summary output of linear models on pH, concentrations of chloride, sulfate, nitrate, sodium, ammonium, potassium, magnesium, calcium, total organic nitrogen, total inorganic nitrogen, total organic carbon, and total inorganic carbon within the turf harvesting locations (in 2019 and 2021) and turfs within turf and turf + shredded center plots, separated by soil layer and site. Cl^-, SO_4^-, NO_3^-, Na^+, NH_4^- (mineral, Q1 organic), Mg^{2+} (organic), Ca^{2+} (organic), TIN (mineral), TON (mineral, Q1 organic), and TOC (mineral) were log-transformed, and K^+, Mg^{2+} (mineral), and TON (Q1 organic) were square root-transformed to meet statistical assumptions. Boldface indicates significance ($p=<0.05$)

Nutrient/ ion	**Soil Layer**	**Site**	**Variable**	**df**	**Sum of Squares**	**Mean Squares**	**F- Statistic**	**p-value**
pH	Organic	Q1	Treatment	2	5.71	2.85	7.23	**0.002**
			Residuals	37	14.61	0.39		
		Q2	Treatment	2	0.59	0.29	4.36	**0.020**
			Residuals	37	2.50	0.07		
	Mineral	Q1	Treatment	2	21.02	10.51	48.75	**<0.001**
			Residuals	37	7.98	0.21		
		Q2	Treatment	2	16.07	8.04	114.09	**<0.001**
			Residuals	37	2.54	0.07		
Cl^-	Organic	Q1	Treatment	2	58.18	29.09	17.80	**<0.001**
			Residuals	37	60.47	1.63		
		Q2	Treatment	2	11.39	5.69	11.21	**<0.001**
			Residuals	37	18.80	0.51		
	Mineral	Q1	Treatment	2	99.56	49.78	74.52	**<0.001**
			Residuals	37	24.72	0.67		
		Q2	Treatment	2	21.83	10.91	20.29	**<0.001**
			Residuals	36	19.36	0.54		
SO_4^{2-}	Organic	Q1	Treatment	2	11.64	5.82	1.84	0.172
			Residuals	37	116.50	3.15		
		Q2	Treatment	2	50.81	25.40	82.46	**<0.001**
			Residuals	37	11.40	0.31		
	Mineral	Q1	Treatment	2	63.36	31.68	30.27	**<0.001**
			Residuals	37	38.72	1.05		
		Q2	Treatment	2	45.62	22.81	40.32	**<0.001**
			Residuals	36	20.37	0.57		
NO_3^-	Organic	Q1	Treatment	2	18.58	9.29	6.92	**0.003**
			Residuals	36	46.34	1.34		
		Q2	Treatment	2	18.69	9.35	22.19	**<0.001**
			Residuals	36	15.16	0.42		
	Mineral	Q1	Treatment	2	28.20	14.10	31.64	**<0.001**
			Residuals	34	15.15	0.45		
		Q2	Treatment	2	3.26	1.63	2.86	0.072
			Residuals	32	18.23	0.57		
Na^+	Organic	Q1	Treatment	2	56.39	28.20	16.59	**<0.001**
			Residuals	37	62.90	1.70		

		Q2	Treatment	2	30.61	15.31	24.96	**<0.001**
			Residuals	37	22.70	0.61		
	Mineral	Q1	Treatment	2	104.17	52.08	40.77	**<0.001**
			Residuals	37	47.27	1.28		
		Q2	Treatment	2	55.65	27.83	73.31	**<0.001**
			Residuals	36	13.66	0.38		
NH_4^+	Organic	Q1	Treatment	2	2.87	1.44	2.08	0.141
			Residuals	33	22.81	0.69		
		Q2	Treatment	2	1564	782.2	8.69	**<0.001**
			Residuals	35	3150	90.00		
	Mineral	Q1	Treatment	2	2.02	1.01	7.04	**0.005**
			Residuals	20	2.87	0.14		
		Q2	Treatment	2	3.72	1.86	17.18	**<0.001**
			Residuals	29	3.14	0.11		
K^+	Organic	Q1	Treatment	2	29.18	14.59	2.57	0.091
			Residuals	37	210.45	5.69		
		Q2	Treatment	2	2390	1195	1.33	0.277
			Residuals	37	33271	899.2		
	Mineral	Q1	Treatment	2	23.39	11.70	13.62	**<0.001**
			Residuals	37	31.78	0.86		
		Q2	Treatment	2	25.51	12.75	10.62	**<0.001**
			Residuals	36	43.23	1.20		
Mg^{2+}	Organic	Q1	Treatment	2	6.42	3.21	2.82	0.073
			Residuals	37	42.19	1.14		
		Q2	Treatment	2	4.69	2.34	15.61	**<0.001**
			Residuals	37	5.56	0.15		
	Mineral	Q1	Treatment	2	27.08	13.54	17.93	**<0.001**
			Residuals	36	27.19	0.76		
		Q2	Treatment	2	7.34	3.67	9.64	**<0.001**
			Residuals	36	13.71	0.38		
Ca^{2+}	Organic	Q1	Treatment	2	12.74	6.37	6.08	**0.005**
			Residuals	37	38.80	1.05		
		Q2	Treatment	2	3.29	1.65	10.36	**<0.001**
			Residuals	37	5.88	0.16		
	Mineral	Q1	Treatment	2	174.3	87.15	22.96	**<0.001**
			Residuals	36	136.7	3.80		
		Q2	Treatment	2	110.2	55.08	9.86	**<0.001**
			Residuals	36	201.2	5.59		
TON	Organic	Q1	Treatment	2	11566	5783	6.57	**0.011**
			Residuals	37	32557	879.9		
		Q2	Treatment	2	$8.30\text{x}10^7$	$4.15\text{x}10^7$	1.83	0.175
			Residuals	37	$8.40\text{x}10^8$	$2.27\text{x}10^7$		
	Mineral	Q1	Treatment	2	1.84	0.92	2.06	0.142
			Residuals	37	16.49	0.45		
		Q2	Treatment	2	7.62	3.81	6.10	**0.005**

			Residuals	36	22.48	0.62		
TIN	Organic	Q1	Treatment	2	5.36	2.68	2.41	0.104
			Residuals	37	41.24	1.12		
		Q2	Treatment	2	742.5	371.3	3.94	**0.028**
			Residuals	37	3487	94.24		
	Mineral	Q1	Treatment	2	19.43	9.71	21.49	**<0.001**
			Residuals	37	16.72	0.45		
		Q2	Treatment	2	5.43	2.71	5.31	**0.010**
			Residuals	36	18.41	0.51		
TOC	Organic	Q1	Treatment	2	4.34x10^{10}	2.17x10^{10}	3.37	0.069
			Residuals	12	7.72x10^{10}	6.44x10^{9}		
		Q2	Treatment	2	1.79x10^{10}	8.94x10^{9}	2.29	0.144
			Residuals	12	4.69x10^{10}	3.91x10^{9}		
	Mineral	Q1	Treatment	2	0.14	0.07	0.15	0.861
			Residuals	12	5.68	0.47		
		Q2	Treatment	2	6.26	3.13	8.01	**0.006**
			Residuals	12	4.69	0.39		
TIC	Organic	Q1	Treatment	2	1.03x10^{6}	5.17x10^{5}	0.54	0.599
			Residuals	12	1.16x10^{7}	9.65x10^{5}		
		Q2	Treatment	2	4.00x10^{5}	2.00x10^{5}	0.24	0.790
			Residuals	12	1.00x10^{7}	8.33x10^{5}		
	Mineral	Q1	Treatment	2	7.03x10^{6}	3.51x10^{6}	23.92	**<0.001**
			Residuals	12	1.76x10^{6}	1.47x10^{5}		
		Q2	Treatment	2	7.92x10^{6}	3.96x10^{6}	23.47	**<0.001**
			Residuals	12	2.02x10^{6}	1.68x10^{5}		

Appendix B CHAPTER 4 SUPPLEMENTAL INFORMATION

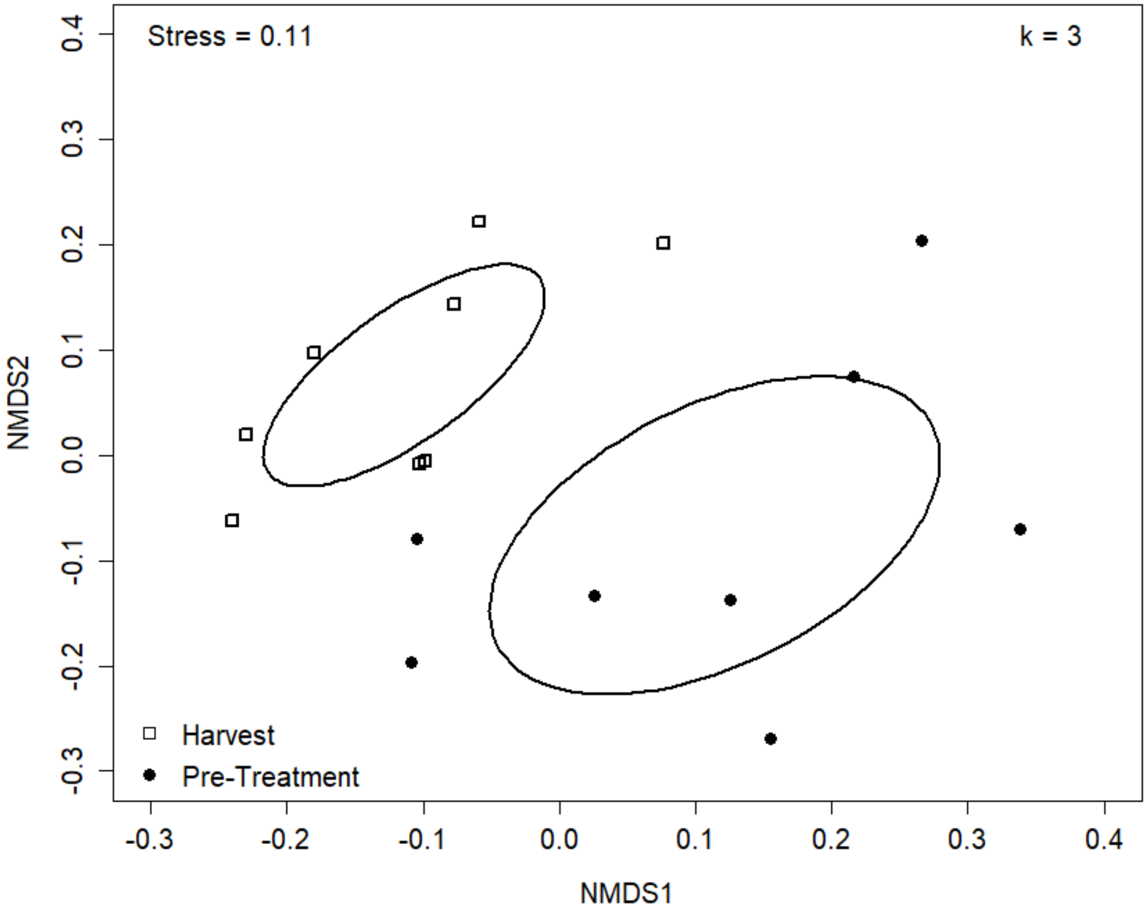

Figure B.1. A non-metric multidimensional scaling analysis of turf community composition recorded at the time of harvest (squares) and before implementing experimental treatments (circles), roughly seven months after harvest.

Table B.1. Results of Permutational Multivariate Analysis of Variation (PerMANOVA) analysis on turf community composition between recording at Rankin Inlet and recording at Saskatoon. Boldface indicates significance ($p<0.05$).

	Variable	**df**	**Sum of Squares**	**Mean Squares**	**Pseudo-F**	**R^2**	**Significance**
Turf Community Composition	Date	1	0.28	0.28	3.0	0.18	**0.002**
	Residuals	14	1.3	0.09		0.82	
	Total	15	1.6	0.04		1.00	

Table B.2. Growth forms, species, litter, and bare ground that contributed to ≥70% of the overall dissimilarity between recording at harvest and recording before turf separation. The dissimilarity contribution of species was identified using Bray-Curtis dissimilarities in a SIMPER analysis. Absolute[3] and relative[4] percent differences between harvest and before turf separation were calculated. Species that were not recorded at harvest but recorded before turf separation are simply recorded as a 100% increase in absolute cover.

Growth Form	**Dissimilarity Contribution (SIMPER)**	**Absolute Percent Difference from Harvest (Relative)**	**Species**	**Absolute Percent Difference from Harvest (Relative)**
Lichen	*26%*	-21% (-8%)	*T. vermicularis*	-46% (-4%)
			G. nigricans	-26% (-2%)
			A. ochroleuca	-3% (0%)
			C. nivalis	-14% (-1%)
			D. arctica	+43% (+1%)
Plant Litter	*17%*	+83% (+9%)	Plant Litter	+25% (+2%)
			Woody Litter	+1167% (+6%)
Evergreen Shrubs	*15%*	+4% (+1%)	*D. integrifolia*	+61% (+2%)
			C. tetragona	+9% (0%)
			V. vitis-idaea	-45% (-1%)
Moss	*12%*	+23% (+3%)	*D. crispula*	+100% (+6%)
			A. turgidum	-33% (-3%)
Deciduous Shrubs	*11%*	-28% (-4%)	*V. uliginosum*	-37% (-4%)

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[3] Absolute percent differences = $(\mathit{preseparation\ species\ cover} - \mathit{harvest\ species\ cover}) \div \mathit{harvest\ species\ cover}$

[4] Relative percent differences = $((\mathit{preseparation\ species\ cover} \div \mathit{total\ preseparation\ species\ cover}) - (\mathit{harvest\ species\ cover} \div \mathit{total\ harvest\ species\ cover}))$

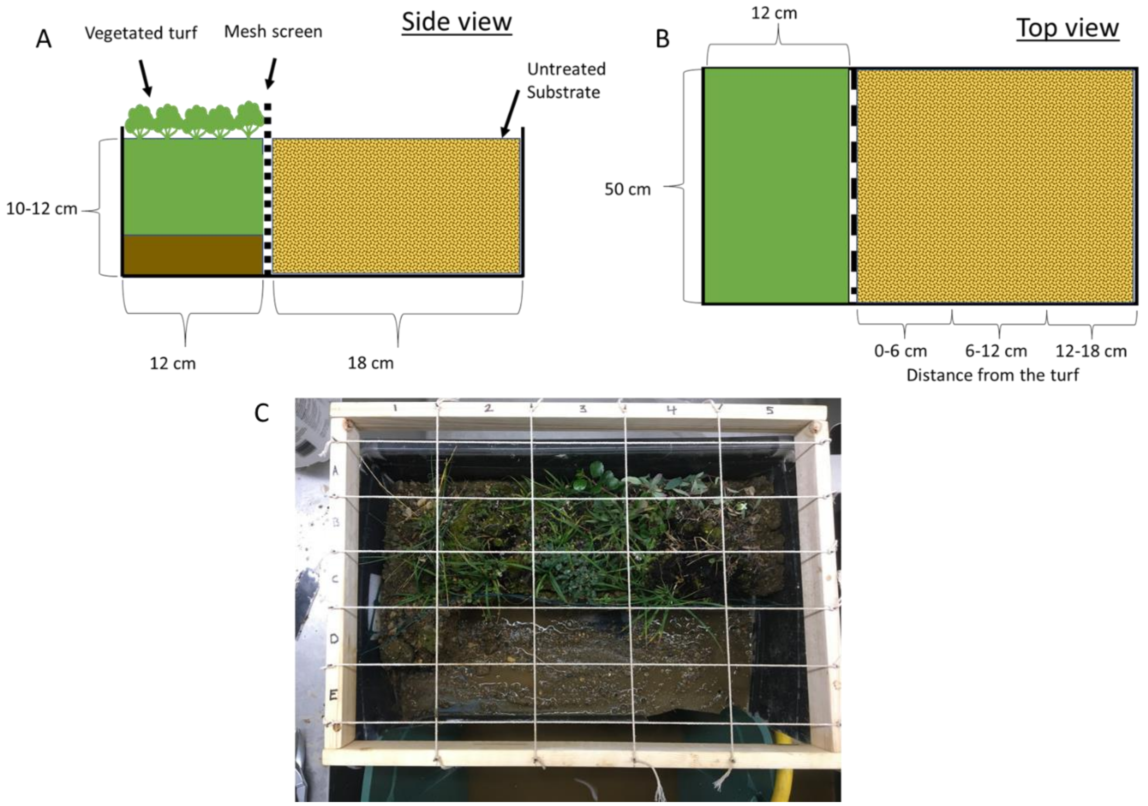

Figure B.2. Side view (Panel A), and top view (Panel B) conceptual diagram of experimental treatments. Panel C shows a turf during root harvesting, with gridded quadrat overtop.

Appendix B.1 Detailed DNA sequencing information:

All protocols from the kit were followed except for a second final rinse, with extracted DNA eluted in 60 µl of EB solution. DNA quantification was performed using a Qubit 2.0 Fluorometer (Invitrogen, Massachusetts, USA) with the Qubit HS assay kit (Invitrogen, Massachusetts, USA). All samples were standardized to 1 ng µl^{-1} for amplification. Plant DNA was amplified in the trnL operon region, using the modified trnL c-1 forward primer (5' - CGAAATYGGTAGACGCTACG - 3') and the modified trnL h-1 reverse primer (5' - CCDTYGAGTCTCTGCACCTATC - 3') found in Lamb *et al*. (2016). Reactions were performed at a final volume of 25 µl; 2 µL of template DNA, 12.5 µL of Platinum Green (2X) Master Mix (Thermo Fisher, Thermo Fisher, Massachusetts, USA), 1.5 µL forward primer (10µM), 1.5 µL reverse primer (10 µM), and 7.5 µL of Polymerase Chain Reaction (PCR) grade water and cycling conditions of 5 mins at 95°C, 35 cycles: 95°C 30 s, 55°C 45 s, 72°C 60 s, and a final extension time at 72°C for 10 mins. PCR products were visualized on a 1.5% agarose gel to confirm amplification. Products were purified using Nucleo-Mag NGS Clean-up and Size Select Beads, and followed the manufacturer's instructions, except for a reduced drying time after the second ethanol wash (2 minutes).

Library preparation for the Illumina MiSeq followed the Illumina Library Preparation Guide (#15044223 Rev. A) and used the Nextera XT Index Kit v2 Adapters (Illumina, San Diego, USA). Reactions were performed at a final volume of 25 µl; 2 µL of template DNA, 12.5 µL of Platinum Green (2X) Master Mix (Thermo Fisher, Thermo Fisher, Massachusetts, USA), 1.5 µL forward primer (10µM), 1.5 µL reverse primer (10 µM), and 7.5 µL of PCR grade water and cycling conditions of 5 mins at 95°C, 35 cycles: 95°C 30 s, 55°C 45 s, 72°C 60 s, and a final extension time at 72°C for 10 mins. Products were purified using Nucleo-Mag NGS Clean-up and Size Select Beads (Macherey-Nagel, Germany), and followed the manufacturer's instructions, except for a reduced drying time after the second ethanol wash (2 minutes). The final concentration of the trnL library was 10 pM, along with a 25% spike of PhiX (Illumina, San Diego, USA).

Table B.3. Results of a Permutational Multivariate Analysis of Variation (PerMANOVA) on turf community composition between recording on Day 0 and recording on Day 140, and on substrate community composition on Day 140. Boldface indicates significance ($p<0.05$).

	Variable	**df**	**Sum of Squares**	**Mean Squares**	**Pseudo-F**	**R^2**	**Significance**
Turf Community Composition	Treatment	1	0.16	0.16	1.3	0.03	0.25
	Time	1	0.71	0.71	5.5	0.16	**0.001**
	Treatment* Time	1	0.04	0.04	0.31	0.09×10^{-1}	0.97
	Residuals	28	29	0.13		0.80	
	Total	31	4.5			1.00	
Substrate Community Composition	Treatment	1	0.67	0.67	10	0.42	**0.001**
	Residuals	14	0.93	0.07		0.58	
	Total	15	1.6			1.00	

Table B.4. Results of two-way within-subjects analysis of variance tests on biological soil crust cover and species richness, and belowground biomass and species richness. df$_{\mathrm{Num}}$ indicates degrees of freedom numerator, df$_{\mathrm{Den}}$ indicates degrees of freedom denominator. SS$_{\mathrm{Num}}$ indicates sum of squares numerator, SS$_{\mathrm{Den}}$ indicates sum of squares denominator. ŋ$^{2}_{g}$ indicates generalized eta-squared. Boldface indicates significance ($p<0.05$).

	Variable	**df$_{\mathrm{Num}}$**	**df$_{\mathrm{Den}}$**	**SS$_{\mathrm{Num}}$**	**SS$_{\mathrm{Den}}$**	**F**	**p**	**ŋ$^{2}_{g}$**
Biological Soil Crust Cover	Treatment	1	7	5.7	1.6	25	**0.0016**	0.50
	Distance	2	14	0.09×10^{-1}	0.46	0.14	0.77	0.02×10^{-1}
	Treatment* Distance	2	14	0.1	0.74	0.93	0.41	0.17×10^{-1}
Biological Soil Crust Species Richness	Treatment	1	7	3.7	1.7	15	**0.005**	0.43
	Distance	2	14	0.19×10^{-1}	0.37	0.36	0.70	0.04×10^{-1}
	Treatment* Distance	2	14	0.16	0.50	2.2	0.15	0.31×10^{-1}
Belowground Biomass	Treatment	1	7	0.25	0.28	6.3	**0.041**	0.26
	Distance	2	14	0.20	0.83×10^{-1}	17	**<0.001**	0.22
	Treatment* Distance	2	14	0.14×10^{-1}	0.24×10^{-1}	3.9	**0.045**	0.19×10^{-1}
Belowground Species Richness	Treatment	1	7	0.28	0.64	3.1	0.12	0.11
	Distance	2	14	0.17	0.39	3.1	0.075	0.69×10^{-1}
	Treatment* Distance	2	14	0.37×10^{-1}	0.29	0.91	0.42	0.16×10^{-1}

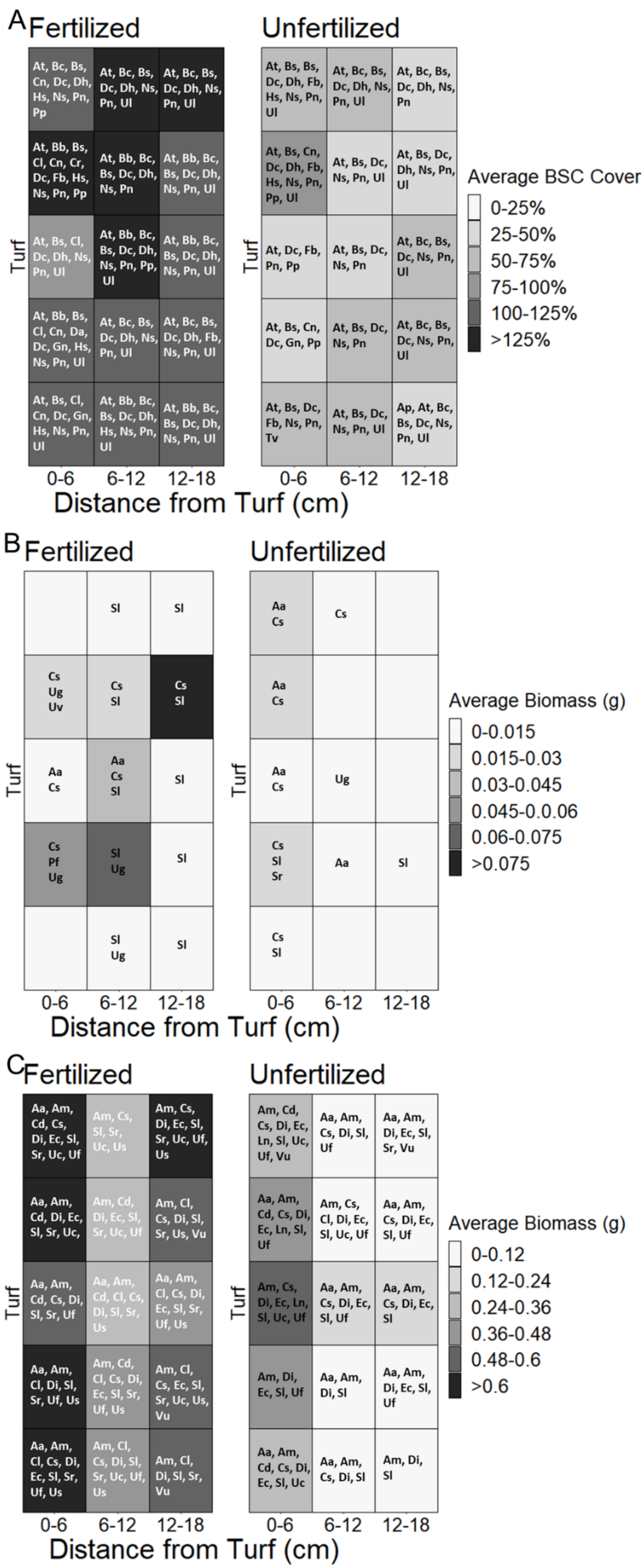
A
Fertilized
At, Bc, Bs, Cn, Dc, Dh, Hs, Ns, Pn, Pp
At, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bb, Bs, Cl, Cn, Cr, Dc, Fb, Hs, Ns, Pn, Pp
At, Bb, Bc, Bs, Dc, Dh, Ns, Pn
At, Bb, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bs, Cl, Dc, Dh, Ns, Pn, Ul
At, Bb, Bc, Bs, Dc, Dh, Ns, Pn, Pp, Ul
At, Bb, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bb, Bs, Cl, Cn, Da, Dc, Gn, Hs, Ns, Pn, Ul
At, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bc, Bs, Dc, Dh, Fb, Ns, Pn, Ul
At, Bs, Cl, Cn, Dc, Gn, Hs, Ns, Pn, Ul
At, Bb, Bc, Bs, Dc, Dh, Hs, Ns, Pn, Ul
At, Bb, Bc, Bs, Dc, Dh, Ns, Pn, Ul
Turf
0-6
6-12
12-18
Unfertilized
At, Bs, Bs, Dc, Dh, Fb, Hs, Ns, Pn, Ul
At, Bc, Bs, Dc, Dh, Ns, Pn, Ul
At, Bc, Bs, Dc, Dh, Ns, Pn
At, Bs, Cn, Dc, Dh, Fb, Hs, Ns, Pn, Pp, Ul
At, Bs, Dc, Ns, Pn, Ul
At, Bs, Dc, Dh, Ns, Pn, Ul
At, Dc, Fb, Pn, Pp
At, Bs, Dc, Ns, Pn
At, Bc, Bs, Dc, Ns, Pn, Ul
At, Bs, Cn, Dc, Gn, Pp
At, Bs, Dc, Ns, Pn
At, Bc, Bs, Dc, Ns, Pn, Ul
At, Bs, Dc, Fb, Ns, Pn, Tv
At, Bs, Dc, Ns, Pn, Ul
Ap, At, Bc, Bs, Dc, Ns, Pn, Ul
Turf
0-6
6-12
12-18
Distance from Turf (cm)
Average BSC Cover
0-25%
25-50%
50-75%
75-100%
100-125%
>125%
B
Fertilized
Sl
Sl
Cs Ug Uv
Cs Sl
Cs Sl
Aa Cs
Aa Cs Sl
Sl
Cs Pf Ug
Sl Ug
Sl
Sl Ug
Sl
Turf
0-6
6-12
12-18
Unfertilized
Aa Cs
Cs
Aa Cs
Aa Cs
Ug
Cs Sl Sr
Aa
Sl
Cs Sl
Turf
0-6
6-12
12-18
Distance from Turf (cm)
Average Biomass (g)
0-0.015
0.015-0.03
0.03-0.045
0.045-0.0.06
0.06-0.075
>0.075
C
Fertilized
Aa, Am, Cd, Cs, Di, Ec, Sl, Sr, Uc, Uf
Am, Cs, Sl, Sr, Uc, Us
Am, Cs, Di, Ec, Sl, Sr, Uc, Uf, Us
Aa, Am, Cd, Di, Ec, Sl, Sr, Uc,
Am, Cd, Di, Ec, Sl, Sr, Uc, Uf
Am, Cl, Cs, Di, Sl, Sr, Us, Vu
Aa, Am, Cd, Cs, Di, Sl, Sr, Uf
Aa, Am, Cd, Cl, Cs, Di, Sl, Sr, Us
Aa, Am, Cl, Cs, Di, Ec, Sl, Sr, Uf, Us
Aa, Am, Cl, Di, Sl, Sr, Uf, Us
Am, Cd, Cl, Cs, Di, Ec, Sl, Sr, Uf, Us
Am, Cl, Cs, Ec, Sl, Sr, Uc, Us, Vu
Aa, Am, Cl, Cs, Di, Ec, Sl, Sr, Uf, Us
Am, Cl, Cs, Di, Sl, Sr, Uc, Uf, Us
Am, Cl, Di, Sl, Sr, Vu
Turf
0-6
6-12
12-18
Unfertilized
Am, Cd, Cs, Di, Ec, Ln, Sl, Uc, Uf, Vu
Aa, Am, Cs, Di, Sl, Uf
Aa, Am, Di, Ec, Sl, Sr, Vu
Aa, Am, Cd, Cs, Di, Ec, Ln, Sl, Uf
Am, Cs, Cl, Di, Ec, Sl, Uc, Uf
Aa, Am, Cs, Di, Ec, Sl, Uf
Am, Cs, Di, Ec, Ln, Sl, Uc, Uf
Aa, Am, Cs, Di, Ec, Sl, Uf
Aa, Am, Cs, Di, Ec, Sl
Am, Di, Ec, Sl, Uf
Aa, Am, Di, Sl
Aa, Am, Di, Ec, Sl, Uf
Aa, Am, Cd, Cs, Di, Ec, Sl, Uc
Aa, Am, Cs, Di, Sl
Am, Di, Sl
Turf
0-6
6-12
12-18
Distance from Turf (cm)
Average Biomass (g)
0-0.12
0.12-0.24
0.24-0.36
0.36-0.48
0.48-0.6
>0.6

Figure B.3. Expanded version of Figure 3.4 showing species observed within each individual grid. Heatmaps show biological soil crust cover (A), aboveground vascular biomass (B), and belowground biomass (C). Darker colors on heatmaps represent higher biomass or cover. Species codes for BSCs are: Ap = *Aulacomnium palustre*, At = *Aulacomnium turgidum*, Bb = *Bryum blindii*, Bc = *Barbula convoluta*, Bs = Superficial biological soil crust mats, Cl = *Cinclidium latifolium*, Cn = *Cetraria nivalis*, Cr = *Cladonia rangiferina*, Da = *Dactylina arctica*, Dc = *Dicranoweisia crispula*, Dh = *Distichium hagenii*, Fb = Fungal fruiting body, Gn = *Gowardia nigricans*, Hs = *Hylocomium splendens*, Ns = *Nostoc* spp., Pn = *Pohlia nutans*, Pp = *Polytrichum pallidisetum*, Tv = *Thamnolia vermicularis*, Ul = Unidentified white crustose lichen. Species codes for above and belowground vegetation are: Aa = Astragalus alpinus, Am = *Anthoxanthum monticola*, Cd = *Cardamine digitata*, Cl = *Chamaenerion latifolium*, Cs = *Carex scirpoidea*, Di = *Dryas integrifolia*, Ec = *Eriophorum callitrix*, Ln = *Luzula nivalis*, Pf = *Poa flexuosa*, Sr = *Salix reticulata*, Sl = *Stellaria longipes*, Uc = Unidentified *Carex* spp., Uf = Unidentified *Fabaceae* spp., Ug = Unidentified graminoid, Us = Unidentified *Salix* spp., Uv = Unidentified *Viola* spp., Vu = *Vaccinium uliginosum*.