Trail degradation in Cape Breton Highlands National Park:

An ecological approach to vegetation restoration

By
Madeline Clarke

A Thesis Submitted to
Saint Mary's University, Halifax, Nova Scotia
in Partial Fulfillment of the Requirements for
the Degree of Master of Science in Applied Science

April 2020, Halifax, Nova Scotia, Canada

Approved:
Dr. Jeremy Lundholm
Supervisor

Approved:
Dr. Colleen Barber
Committee member

Approved:
Mr. Tony Bowron
Committee member

Approved:
Mr. Dan Kehler
External Examiner

Date: April 14, 2020

Trail degradation in Cape Breton Highlands National Park:

An ecological approach to vegetation restoration

by Madeline Clarke

Abstract

Ecotourism and hiking are increasingly popular, but trails can lead to vegetation loss, substrate compaction and erosion. Degradation and failure of vegetation recovery was observed at two closed sections of trails in Cape Breton Highlands National Park (Skyline and Mica Hill). In 2018, when compared to undamaged vegetation, trail conditions had reduced vascular plant cover and substrate nutrients, as well as higher temperature, compaction, moisture, and pH. Additionally, Skyline had no seed bank and Mica Hill's seed bank was a different community. In 2019, five treatments were implemented and monitored at Skyline: topsoil addition with erosion control mats combined with direct seeding and transplanting treatments. When compared to controls, all treatments improved vegetation cover and quality where added topsoil in combination with transplanting and seeding increased improvement. This study provides the basis for a long-term restoration study where further monitoring over many years can elucidate or modify these findings.

Date: April 29, 2020

Acknowledgements

First, I would like to thank my supervisor, Dr. Jeremy Lundholm, for your guidance and support over the past two years and for giving me the chance to work on this project. The opportunities you have given me have fed my thirst for knowledge and passion for plants. Thank you as well to the members of my supervisory committee, Dr. Colleen Barber and Tony Bowron. You have provided me with invaluable advice, knowledge, and expertise. Also, a big shout out goes to Dr. Tim Frasier, whose guidance on all things statistics was indispensable.

I am also grateful for all the members of the Ecology of Plants in Communities Lab, who were my amazing support system of these past two years. All of you were always there to help with ideas and contributed many bouts of creative problem-solving. A huge thank you to all those who quite literally carried my project on your backs: Dr. Jeremy Lundholm, Tasha Rabinowits, Maggie Scarrow, Lyndsey Burrell, Terrell Roulston, Maka Ngulube, and Élise Rogers. This restoration project would not have been possible with your help.

I would also like to thank Park Ecologist James Bridgland and all the staff at the Cape Breton Highlands National Park who helped with field work and project management. Particularly, I would like to thank Louis Cormier and Shane Cormier, for your company throughout many long drives and hikes and for your valued field assistance.

Finally, I would like to thank my family and friends, your love and support have made all this possible.

Table of Contents

List of Tables

List of Figures

[redacted] General Introduction

1.1 Background

Human activity has greatly altered and disturbed the natural world on a global scale. Climate change and the decline of global biodiversity have created an imperative to understand how ecosystems function and interact in order to help find ways to mitigate, reduce, and prevent negative environmental impacts caused by humans. Habitat loss is one of the driving factors for species loss globally (Tilman et. al., 1994; Fagúndez, 2012) and the quality of remaining habitat is also of concern. Damaged or altered habitats can eventually lead to local species loss through slower evolutionary processes such as genetic inbreeding, genetic drift and population fragmentation (Kuussaari et. al., 2009). Finding ways to reinvigorate affected areas can help reduce the human impact on the natural world.

Restoration practices are becoming increasingly commonplace as they have the potential to reduce or reverse the loss of biodiversity (Bullock et. al., 2011). Human impacts on the environment have created a new facet of ecology, known as restoration ecology, to study which can be used to further our understanding of ecosystem processes and ecosystem recovery. Restoration ecology is the scientific practice of restoring ecosystems that have become damaged, degraded, or destroyed (Martin, 2017). While it is important that there are many restoration efforts being conducted to test and understand various treatment strategies, there is an increasing call to be more proactive and help affected environments before they become degraded (Acosta et. al., 2018). Restoration efforts can be time consuming and costly, which drives the need to increase monitoring efforts of affected areas and introduce mitigation strategies as they are identified and before damage has a chance to accumulate.

Recreation ecology studies the environmental impact of human visitation to wilderness and protected areas (Marion et. al., 2016) and is thus often associated with restoration and mitigation practices. It has had a relatively short history where interest on the ecological impacts of tourism began in the 1960s and collaborative research began in the 1970s (Cole, 2004). As the world becomes more developed and most work environments are now indoors, there are increasing numbers of people who are spending time outdoors as a form of leisure (Liddle, 1997; Manning & Anderson, 2012). The increase in tourism directed toward natural environments, or ecotourism, has, in many cases, led to negative impacts on the environment which impacts the overall aesthetics of the area and plays an important component in visitor experience (Santarém et. al., 2015). When considering recreation ecology as a discipline, there are three major factors involved: the environmental impact concerning resource use, the social impact concerning visitor experience, and the managerial impact (Manning & Anderson, 2012).

1.2 Visitors and ecotourism

Most often, ecotourism has a positive economic and social influence on the region being promoted. The creation and management of easily accessible trails with scenic views can increase public awareness of nature-based conservation. Interpretive trails can inform visitors of the value of the environment that surrounds them and has been shown to increase their appreciation of nature (Timothy, 2015). As more people live in cities and are removed from nature, increasing nature education can be a tool to help conserve the natural world by forming bonds with nature

Ecotourism can also have positive economic impacts. The influx of visitors to a region can support the management of the park itself and fund conservation efforts (Santarém et. al., 2015) and support the local economy, which can often be substantial in more rural areas (Lemky, 2017). The creation of trail systems can create employment opportunities through maintenance and enforcement requirements as well as to run visitor centres, campgrounds, and equipment rental offices (Timothy, 2015). Nearby towns often benefit greatly in areas with active trails and scenic attractions where visitors populate hotels, bed and breakfasts, and restaurants.

It is also important to note that trails, especially in national parks, are not only about showcasing the surrounding environment and supporting the economy, but they will also often have cultural significance. Cultural values can include historic routes for early settlers of an area and are often accompanied with signage to inform visitors of the significance of the trail. Often trails are created for there cultural importance and this can be the primary factor that draws in visitors. This, in turn, helps to preserve historic values and gives the region in question a 'sense of place' wherein the history of the region becomes tangibly present (Timothy, 2015). Visitors thus can have a more holistic experience of the region and further entice visitors to the area.

Visitor numbers to parks or trails can fluctuate greatly based on promotion, either inadvertent or intentional and, in a short amount of time, sensitive environments can be negatively impacted if not properly managed (Buckley, 2000) or with insufficient funding allocated for conservation efforts (Buckley, 2011). Large numbers of visitors can create crowding on trails, increase conflicts, and increase the risk of damaging the area through vandalism, littering, and trampling (Manning & Anderson, 2012). To add, highly popular trails tend to be focused in areas with greater ecological sensitivity and thus have high conservation values (Buckley, 2000). Therefore, areas that present greater visitor experience and attract large volumes of visitors oftentimes tend to be easily impacted and require the most management.

While the global environmental impacts of ecotourism are not clearly known (Buckley, 2004), we do know that ecotourism has a positive economic impact for surrounding areas and can help economically to support the park. Ecotourism can directly contribute to conservation practices through entrance fees (Buckley, 2000) and while it is important to maximize revenue which can be used to fund conservation and restoration initiatives, it is important to avoid visitor crowding. Different management practices to reduce the effects of overcrowding include increasing recreation opportunities through expansion of the activities and promoting activities during-off peak hours and months (Manning & Anderson, 2012). The result is a more even distribution of visitors which can potentially reduce the overall impact on the environment. Other more specific management tactics include visitor education, the use of fines or increased entry prices, the implementation of law enforcement to limit access, rules and regulation, and zoning (Manning & Anderson, 2012).

Overall, ecotourism can provide significant economic boosts to associated communities and can, in many instances, help to preserve cultural values and add to nature appreciation. However, overuse or misuse has led to new areas of research where scientists and practitioners alike are attempting to recover and restore degraded areas. The overarching goal is to find a way promote outdoor recreation activities whilst maintaining a balance with the natural world. Often this may entail the exclusion of people from a degraded area followed by passive or active restoration procedures.

1.3 Restoration ecology and ecological theory

The main goal of restoration ecology is to create or return an ecosystem to a point where it is self-sustaining and resilient to further disturbance (Ruiz-Jaen & Aide, 2005). The two main restoration strategies are passive or active restoration. Passive restoration occurs where the identified stressor is removed, and the ecosystem can recover naturally without further intervention. A common passive restoration strategy is simply to fence off the degraded area and wait for natural processes to recover and resume normally. As a result, passive restoration is generally cost effective and should always be the first option considered in a restoration project. For example, passive restoration efforts of simply closing off degraded sections of trails and allowing the area to regenerate have been successful but only when the seedbank is still intact (Sawtshuk et. al., 2010).

Therefore, passive restoration is often not enough to create a self-sustaining ecosystem and active strategies need to be implemented. These include biotic and abiotic manipulations of the ecosystem, in order to recreate a self-sustaining ecosystem or to create conditions capable of recreating self-sustainability at a rate faster than passive restoration techniques could offer. As active restoration can quickly become a costly process, and so careful considerations need to be made with regard to identifying site-specific environmental stresses.

Likewise, prior to beginning any restoration project, it is vital to understand the underlying theory that supports our understanding of ecosystems and communities. A community is an assemblage of species and the interactions between them whereas an ecosystem encompasses all biotic and abiotic interactions in a given area. Ecological assembly theories range from completely non-random assembly, where the communities exist only in certain assemblages, to completely random assembly, where the species that make up a community are more coincidental. It is thought that local communities are likely assembled somewhere along a gradient between these two opposing theories, taking both random and non-random effects into consideration (Götzenberger et. al., 2012).

When considering community reassembly following a disturbance, a component of the recovery process will be random and thus the outcome of a restoration project will be similarly randomly affected. Fenton and Bergeron (2013) found that stochastic processes were important in the reestablishment of boreal bryophyte species following fire disturbances, suggesting that processes other than habitat quality were at play during the recovery process. Their study places the emphasis of recovering ecosystem functions ahead of creating a ‘field of dreams’ (Hilderbrand et. al., 2005) wherein habitat quality is restored in the hopes that the intended species colonize the area.

It is important to note that community assemblages are very dynamic and are constantly changing in time and space. Once a community is established, it will continue to change as species are slowly introduced and removed from a local area. If enough species from a different community type are added, the whole community can shift to another type via the process of succession. According to Howell et. al. (2012) ecological succession is defined as “a shift in the presence or relative abundance of species populations over time in a given location under a relatively stable climate”. A savannah can turn over into a shrubland, which can eventually give way to forest, for example. A community can be considered stable when a successional stage persists assuming underlying conditions do not change, and it should be noted that, at each successional stage, there can still be significant local species turnover (Walker & del Moral, 2009). Communities are continuously shifting throughout time and this must be considered when attempting to regenerate an area. Disturbance events initiate succession and can even alter previous successional states. Incorporating natural recovery processes, which vary from site to site but are inevitably strongly influenced by regional climate (Howell et. al., 2012), into restoration practices can greatly influence the outcome (Walker & del Moral, 2009).

A major driver of succession are the effects of individuals as they interact with each other and alter the environment (Howell et. al., 2012). In general, if an interaction between plants occurs, there are two possible outcomes: facilitation or competition. This is to say that when two individuals co-occur, they will either help each other, compete, or have no effect. This is often observed at the species level, where, for example, a certain species can outcompete others and become the dominant vegetation.

Facilitation occurs when one species provides a beneficial service for another species, or they provide mutually beneficial services to each other. For plants, facilitation typically occurs between neighbouring plant species who are in close enough proximity to physically and/or chemically affect each other. Benefits can include improved germination rates and growth (Rodríguez-Echeverría et. al., 2013), and are often in some form of a barrier or buffer for protection from sun, wind, or rain exposure. These positive interactions tend to more predominant in stressful environments (Callaway et. al., 2002).

While facilitation is beneficial, competition is the more commonly observed strategy amongst plants (Damgaard, 2004). Neighbouring plants must compete for sunlight, water, and nutrients. Eventually one individual may acquire enough resources to outcompete its neighbour, who suffers as a result from lack of resources. Competition is most notable at the species-level, where a species who is better adapted to the region or who is better able to acquire resources will grow more quickly and outcompete other species. Often the competitor will thrive and become the dominant vegetation. Factors such as keystone species (Ballantyne & Pickering, 2015), stressful climatic conditions (Callaway et. al., 2002), and environmental heterogeneity (Dufour et. al., 2006) can help reduce the population of a competitively dominant species and maximize biodiversity.

Other important considerations in a restoration projects include resistance and resilience of ecosystems and target species populations. In terms of restoration ecology, resistance is the ability to withstand disturbance or a damaging event, and resilience is the ability to recover following a disturbance or damaging event. Both resistance and resilience can vary between different plant communities, community types, and functional traits (Bernhardt-Römermann et. al., 2011). For example, species with cespitose and rosette growth forms have more resistance, erect growth forms have less resistance, and woody perennial plants with low-growing buds have less resilience (Cole, 1995). Many restoration practices are either resistance-based (e.g., implementing mitigation strategies to avoid future change to an ecosystem) or resilience-based (e.g., introducing a wider array of genotypes). While they are not mutually exclusive, there is a tendency for practitioners to advocate for resilience-based programs (Heller & Zavaleta, 2009). Promoting resilience can also lead to recovery of interspecific interaction processes (Aslan et. al., 2016). Restoring an ecosystem to a point where it is capable of adapting to unforeseen or stochastic processes can increase the chances of a successful restoration project.

Restoration projects represent unique opportunities to apply ecological concepts to restore degraded ecosystems while simultaneously testing these concepts through active manipulation of the environment. A holistic understanding of ecological theory can improve the chances of restoration success. This is especially important in an era where there is increasing awareness of climate change-related impacts on ecological research and nature conservation (Prober et. al., 2019).

1.4 Nova Scotia Barrens

In Nova Scotia, barrens are dominated by ericaceous vegetation and are generally subject to extreme environmental stressors which reduce vegetation height. (Oberndorfer & Lundholm, 2009; Cameron & Bondrup-Nielson, 2013). These include high winds, high altitudes, and can often include a coastal influence. Akin to heathlands and shrublands, barrens also generally tend to have nutrient poor, acidic soils when compared to arable farmland (Clarke, 1997), but they contain complex vegetation communities that often contain many uncommon alpine plant species as the local climate can resemble an alpine environment (Cameron & Bondrup-Nielson, 2013).

Nova Scotia's Museum of Natural History describes three types of barrens habitat in the province: coastal, highland, and inland (1996). Most highland barrens are in Cape Breton's Highlands National Park. This plateau-taiga region exceeds 500 m in elevation and receives a total annual precipitation of 1600 mm which is the most of anywhere in Nova Scotia (Nova Scotia Museum, 1996). The growing season begins when average temperatures reach 5 °C and end when they fall below 5 °C (Nova Scotia Museum, 1996).

Barrens are relatively rare in Nova Scotia and they also provide habitat for rare species within the habitat (Cameron & Bondrup-Nielson, 2013). Their vegetative composition can vary greatly between sites (Oberndorfer & Lundholm, 2009) which increases the need to protect and restore multiple areas. Barren communities are also very complex and dynamic, where interactions occur along abiotic gradients (Canals & Sebastià, 2002; Oberndorfer & Lundholm, 2009) and have potentially diverse and long-lived seed banks (Ghorbani et. al., 2007).

Barrens can be successional habitats when they are maintained by fires or by some other extreme event causing tree loss. Evidence of periodic fire events in inland barrens of Nova Scotia suggest that these areas were once forested, and the existing barrens are in the process of succession and thus are reverting to forested habitat

(Clarkson et. al., 2012). However, not all barrens are considered to be successional and those that can persist against tree encroachment tend to be nearer to the coast and/or at higher elevations (Burley & Lundholm, 2010).

While coastal influences can help maintain barren ecosystems, they are still at risk for habitat loss via human disturbances which includes coastal development projects, hiking trails, etc. (Oberndorfer & Lundholm, 2009). Careful consideration needs to be made when creating trails and routes through these environments. Alpine and heath vegetation, often found in barrens habitat, tends to have poor resistance to damaging events (Liddle, 1997, Hill & Pickering, 2009) and following damage, barren vegetation tends to be slow growing and it can take a long time to recover from disturbance (Cole, 2004; Jägerbrand & Alato, 2011), indicating poor resilience, from a restoration perspective.

1.5 Restoration Goals

The overarching goals of this project are to 1) understand the effects associated with trail degradation at two closed portions of hiking trails in Cape Breton Highlands National Park, 2) determine which factors may be limiting the natural recovery of the ecosystem following trail closure, and finally 3) determine which method of active restoration is most appropriate for these specific environments. The project took place over two years, where trail conditions were assessed during the first year and restoration treatments were implemented at one of the two sites during the second year.

This project represents the set-up of a potential long-term restoration study in the barrens of Nova Scotia. Outcomes for this project build on prior knowledge as to the effects of trampling and degradation, such as differences in compaction and substrate moisture, on barrens vegetation types. It also compares various restoration treatment types and provides general information on restoration for future barren restoration studies as well as site-specific information to Parks Canada as they plan to move forward with a larger scale restoration application.

1.6 Literature Cited

Acosta, A. L., d'Albertas, F., de Souza Leite, M., Saraiva, A. M., & Walter Metzger, J. P. (2018). Gaps and limitations in the use of restoration scenarios: a review. *Restoration Ecology*, *26*(6), 1108–1119. https://doi.org/10.1111/rec.12882

Aslan, C. E., Bronstein, J. L., Rogers, H. S., Gedan, K. B., Brodie, J., Palmer, T. M., & Young, T. P. (2016). Leveraging nature's backup plans to incorporate interspecific interactions and resilience into restoration. *Restoration Ecology*, *24*(4), 434–440. https://doi.org/10.1111/rec.12346

Ballantyne, M., & Pickering, C. M. (2015). The impacts of trail infrastructure on vegetation and soils: Current literature and future directions. *Journal of Environmental Management*, *164*, 53–64. https://doi.org/10.1016/j.jenvman.2015.08.032

Bernhardt-Römermann, M., Gray, A., Vanbergen, A. J., Bergès, L., Bohner, A., Brooker, R. W., … Stadler, J. (2011). Functional traits and local environment predict vegetation responses to disturbance: A pan-European multi-site experiment. Journal of Ecology, 99(3), 777–787. https://doi.org/10.1111/j.1365-2745.2011.01794.x

Buckley, R. (2000). Tourism Recreation Research Tourism in the Most Fragile Environments Tourism in the Most Fragile Environments. *TOURISM RECREATION RESEARCH*, *25*(251), 31–40. https://doi.org/10.1080/02508281.2000.11014898

Buckley, R. (2004). Impacts Positive and Negative: Links Between Ecotourism and Environment. In R. Buckley. *Environmental Impacts of Ecotourism*. (Series no. 2). Massachusetts, USA: CABI Publishing.

Buckley, R. (2011). Tourism and Environment. *Annual Review of Environment and Resources*, *36*, 397–416. https://doi.org/10.1146/annurev-environ-041210-132637

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology and Evolution*, *26*(10), 541–549. https://doi.org/10.1016/j.tree.2011.06.011

Burley, S. T., & Lundholm, J. T. (2010). Environmental predictors of forest expansion on open coastal barrens. *Biodiversity Conservation*, *19*, 3269–3285. https://doi.org/10.1007/s10531-010-9891-0

Callaway, R., Brooker, R. W., Choler, P., Kikvidze, Z., Lortie, C. J., Michalet, R., Paolini, L., Pugnaire, F. I., Newingham, B., Aschehoug, E. T., Armas, C., Kikodze, D., & Cook, B. J. (2002). Positive interactions among alpine plants increase with stress. *Nature, 417*(6891), 844–848. https://doi.org/10.1038/nature00805.1.

Cameron, Robert P., & Bondrup-Nielson, S. (2013). Plant Communities within Atlantic Coastal Heathlands in Nova Scotia. *Northeastern Naturalist*, *20*(4), 694–709.

Canals, R.-M., & Sebastià, M.-T. (2002). Barren dynamics in biotically disturbed areas: on the role of some features enhancing heath success. *Acta Oecologia*, *23*, 303–312.

Clarke, C. T. (1997). Role of soils in determining sites for lowland barren reconstruction in England. *Restoration Ecology*, *5*(3), 256–264. https://doi.org/10.1046/j.1526-100X.1997.09730.x

Clarkson, B. R., Smale, M. C., Williams, P. A., Wiser, S. K., & Buxton, R. P. (2011). Drainage, soil fertility and fire frequency determine composition and structure of gumland heaths in northern New Zealand. *New Zealand Journal of Ecology*, *35*

Cole, D. N. (1995). Experimental Trampling of Vegetation. II. Predictors of Resistance and Resilience. *Journal of Applied Ecology*, *32*(1), 215-224

Cole, D. N. (2004). Impacts of hiking and camping on soils and vegetation: a review. In R. Buckley. *Environmental Impacts of Ecotourism*. (Series no. 2). Massachusetts, USA: CABI Publishing.

Damgaard, C. (2004). Evolutionary ecology of plant-plant interactions: an empirical modelling approach. Aarhus: Aarhus University Press.

Dufour, A., Gadallah, F., Wagner, H. H., Guisan, A., & Buttler, A. (2006). Plant species richness and environmental heterogeneity in a mountain landscape: effects of variability and spatial configuration. *Ecography*, *29*, 573–584.

Fagúndez, J. (2017). The interplay of nutrients and tree invasion shapes the plant community in barren habitats. *Applied Vegetation Science*, *20*(1), 5–6. https://doi.org/10.1111/avsc.12294

Fenton, N. J., & Bergeron, Y. (2013). Stochastic processes dominate during boreal bryophyte community assembly. *Ecology*, *94*(9), 1993-2006

Ghorbani, J., Le Duc, M. G., Mcallister, H. A., Pakeman, R. J., & Marrs, R. H. (2007). Temporal responses of propagule banks during ecological restoration in the United Kingdom. *Restoration Ecology*, *15*(1), 103–117. https://doi.org/10.1111/j.1526-100X.2006.00195.x

Götzenberger, L., de Bello, F., Bråthen, K. A., Davison, J., Dubuis, A., Guisan, A., Leps, J., Lindbord, R. Moora, M., Pärtel, M., Pellissier, L., Pottier, J., Vittoz, P., Zobel, K., & Zobel, M. (2012). Ecological assembly rules in plant communities-approaches, patterns and prospects. *Biological Reviews*, *87*(1), 111–127. https://doi.org/10.1111/j.1469-185X.2011.00187.x

Heller, N. E., & Zavaleta, E. S. (2009). Biodiversity management in the face of climate change: A review of 22 years of recommendations. *Biological Conservation*, (142), 14–32. https://doi.org/10.1016/j.biocon.2008.10.006

Hilderbrand, R. H., Watts, A. C., & Randle, A. M. (2005). The myths of restoration ecology. *Ecology and Society*, *10*(1). https://doi.org/10.5751/ES-01277-100119

Hill, R., & Pickering, C. (2009). Differences in resistance of three subtropical vegetation types to experimental trampling. *Journal of Environmental Management*, *90*, 1305–1312. https://doi.org/10.1016/j.jenvman.2008.07.015

Howell E. A., Harrington, J. A., & Glass, S. B. (2012). *Introduction to restoration ecology*. Washington, DC: Island Press.

Jägerbrand, A. K., & Alatalo, J. M. (2015). Effects of human trampling on abundance and diversity of vascular plants, bryophytes and lichens in alpine heath vegetation, Northern Sweden. *Springer Plus*, *4*(95), 1–12. https://doi.org/10.1186/s40064-015-0876-z

Kuussaari, M., Bommarco, R., Heikkinen, R. K., Helm, A., Krauss, J., Lindborg, R., Öckinger, E., Pärtel, M., Pino, J., Rodà, F., Stefanescu, C., Teder, T., Zobel, M., & Steffan-Dewenter, I. (2009). Extinction debt: a challenge for biodiversity conservation. *Trends in Ecology and Evolution*, *24*(10), 564–571. https://doi.org/10.1016/j.tree.2009.04.011

Lemky, K. (2017). Journal of Heritage Tourism The revitalization of a heritage travel route: Canada's Cabot Trail. *Journal of Heritage Tourism*, *12*(5), 526–535. https://doi.org/10.1080/1743873X.2016.1242592

Liddle, M. J. (1997). *Recreation Ecology.* London; New York: Chapman & Hall.

Manning, R. E. & Anderson, L. E. (2012). *Managing outdoor recreation: cases studies in the national parks.* Wallingford, Oxfordshire; Cambridge, Mass.: Cabi.

Marion, J. L., & Leung, Y.-F. (2004). Environmentally sustainable trail management. In R. Buckley. *Environmental Impacts of Ecotourism*. (Series no. 2). Massachusetts, USA: CABI Publishing.

Marion, J. L., Leung, Y.-F., Eagleston, H., & Burroughs, K. (2016). A Review and Synthesis of Recreation Ecology Research Findings on Visitor Impacts to Wilderness and Protected Natural Areas. *Journal of Forestry*, *114*(3), 352–362. https://doi.org/10.5849/jof.15-498

Martin, D. M. (2017). Ecological restoration should be redefined for the twenty-first century. *Restoration Ecology*, *25*(5), 668–673. https://doi.org/10.1111/rec.12554

Nova Scotia Museum of Natural History (1996). *Natural history of Nova Scotia volume II: themes regions*. Nova Scotia Museum of Natural History, Nova Scotia.

Oberndorfer, E. C., & Lundholm, J. T. (2009). Species richness, abundance, rarity and environmental gradients in coastal barren vegetation. *Biodiversity and Conservation*, *18*, 1523–1553. https://doi.org/10.1007/s10531-008-9539-5

Prober, S. M., Doerr, V. A. J., Broadhurst, L. M., Williams, K. J., & Dickson, F. (2019). Shifting the conservation paradigm: a synthesis of options for renovating nature under climate change. *Ecological Monographs*, *89*(1), 1–23. https://doi.org/10.1002/ecm.1333

Rodríguez-Echeverría, S., Armas, C., Pistón, N., Hortal, S., & Pugnaire, F. I. (2013). A role for below-ground biota in plant – plant facilitation. *Journal of Ecology*, *101*, 1420–1428. https://doi.org/10.1111/1365-2745.12159

Ruiz-Jaen, M. C., & Aide, T. M. (2005). Restoration success: How is it being measured? *Restoration Ecology*, *13*(3), 569–577. https://doi.org/10.1111/j.1526-100X.2005.00072.x

Santarém, F., Silva, R., & Santos, P. (2015). Assessing ecotourism potential of hiking trails: A framework to incorporate ecological and cultural features and seasonality. *Tourism Management Perspectives*, *16*, 190–206. https://doi.org/10.1016/j.tmp.2015.07.019

Sawtschuk, J., Bioret, F., & Gallet, S. (2010). Spontaneous Succession as a Restoration Tool for Maritime Cliff-top Vegetation in Brittany, France. *Restoration Ecology*, *18*(SUPPL. 2), 273–283. https://doi.org/10.1111/j.1526-100X.2010.00653.x

Tilman, D., May, R. M., Lehman, C. L., & Nowak, M. A. (1994). Habitat destruction and the extinction debt. *Letters to Nature*, *371*, 65–66.

Timothy. D. J. (2015). *Tourism and Trails: Cultural, Ecological and Management Issues.* Bristol, UK: Channel View Publications.

Walker, L. R. & del Moral, R. (2009). Transition dynamics in succession: implications for rates, trajectories and restoration. In Hobbs, R. J. & Suding, K. N. *New models for ecosystem dynamics and restoration*. Washington, DC, USA: Island Press.

[redacted] **Trail degradation in barrens and alpine landscapes**

2.1 Introduction

While hiking trails and other forms of recreational paths are very important promoters of the natural environment and ecotourism, overuse or improper use can damage the area. A trail can be defined as a visible linear pathway that does supply, or may have supplied, some transportation service (Timothy, 2015). Hiking trails are created to support foot traffic and managed trails often have hardened surfaces and sometimes raised boardwalks which can reduce the impact of the trail users. However, most often hiking trails are narrow dirt paths through the vegetation. Intentional trails can avoid degradation with proper management. This includes, but is not limited to, proper trail placement, appropriate construction and maintenance, informed trail standards, visitor management, and adequate monitoring (Marion & Leung, 2004). A hiking trail becomes degraded when any or all these areas are affected to the point where biotic and abiotic conditions of the surrounding environment are negatively altered. Often, degradation can promote further degradation, creating a positive feedback effect which inhibits natural regeneration processes (Crisfield et. al., 2012). Degradation can include loss of vegetation cover, altered species composition, trail widening, and soil loss and compaction (Ballantyne & Pickering, 2015).

2.1.1 Impacts on barren and alpine vegetation

Barren and alpine vegetation are particularly prone to degradation as these areas are already subject to more extreme conditions and its low vegetation height can often encourage hikers to go off designated hiking trails which can lead to trail widening and trail braiding. This can lead to short-term and long-term damage on surrounding heath which is slow-growing and can take a significant time to recover. Many studies have been conducted around the world to assess trampling damage on barrens, shrubland, or alpine vegetation along with its ability to recover from damaging events (Ólafsdóttir & Runnström, 2013; Ballantyne & Pickering, 2015; Jägerbrand & Alatalo, 2015).

At low to moderate trampling levels, species richness and productivity tend to increase, but as trampling continues to increase, species richness will then decline (Liddle, 1975; Parikesit et. al., 1995). Initial damage can stimulate the growth of vegetation, but exposure can allow for the colonization of other species and as the intensity of the damage increases, the ability for the native vegetation to recover depends on species-specific characteristics and environmental factors (Bernhardt-Römermann et. al., 2011). In general, the relationship between the amount of use of a trail and the amount of impact on soil and vegetation is curvilinear (Liddle, 1997; Cole, 2004). Thus, some use will have little effect on the soil and vegetation but will exponentially increase as the amount of use increases up to a point where no more habitat can be destroyed or lost.

In general, low shrubs have high resistance to trampling, but once damaged, their ability to recover, or resilience, is low (Cole, 2004). Grasses can be stimulated by light trampling but will experience a reduction in biomass as trampling intensity increases (Liddle, 1975), however, they are the most resistant to trampling effects. Cushion, cespitose, or rosette life forms tend to be more prevalent on trails (Crisfield et. al., 2012). They can tolerate more trampling as their clustered stems are better able to conserve soil moisture and act as a litter trap to gain extra nutrients. The local climatic conditions of the area can also greatly influence the resiliency and resistance of a plant community. It was found that more humid barrens are more resilient than arid barrens (Bernhardt-Römermann et. al., 2017).

One study on trails in an alpine tundra meadows community of the Northern Canadian Rockies found that, while cover was greatly reduced on trails, species diversity and evenness increased (Crisfield et. al., 2012). The authors suggested that the reduction in cover of a particularly dominant species at this site allowed more species to colonize the trails despite the stressful conditions. It is possible that this is in line with the intermediate disturbance hypothesis (Connell, 1978) whereby maximum diversity is achieved at moderate levels of disturbance intensity and frequency.

As more bare ground becomes available, more habitat becomes accessible to more trampling-resistant, non-native species that would not otherwise have access to the area. Human visitors likely play a role in non-native species establishment on trails where seeds can be transported on clothes, shoes, and equipment (Dickens et. al., 2005). Therefore, trampling can create open niche spaces that can be colonized by native and non-native species alike, which can lead to an increase in species diversity. However, the introduction of non-native species can pose a threat to the integrity of the ecosystem as these trampled areas can act as a gateway for further colonization and potentially affect ecosystem functioning.

2.1.2 Impacts on soil

Equally important are the below ground effects of trampling such as soil compaction (Beckett et. al., 2017), the loss of organic matter and altered soil moisture content (Bryan, 1977), altered plant root structure (Hudeck et. al., 2017), the reduction of the amount of oxygen available in the soil for root growth, as well as increased soil strength described as the mechanical resistance of soil (Bassett et. al., 2005). Overall, trampling can directly damage the vegetation as well as cause indirect changes to the soil structure and composition (Liddle, 1975). Changes to soil structure are a common response to intensive trampling and are most often due to soil compaction which reduces pore space and affects available oxygen and water (Liddle, 1997). Plant growth, in the form of root mass and shoot mass, was found to be the lowest when plants were grown in compacted soil under optimal water conditions (Beckett et. al., 2017) such that their ability to grow roots through compacted soil was hindered and subsequent nutrient uptake was negatively affected.

Trampling leads to highly compacted areas that can lead to changes in soil microbial communities and to an overall reduction in soil microbial biomass (Kissling et. al., 2009). Kissling et. al., (2009) proposed that this was likely due to low nutrient cycling as a result of reduced litter biomass. Soil microbial density is positively related to plant production and plant diversity (Zak et. al., 2003) and below ground microbial communities have also been found to play in important role for positive interactions between plants (Rodríguez-Echeverría et. al., 2013). Thus, changes to the soil structure can have negative effects on both soil microbial communities and plant communities.

Other soil characteristics can be affected by trampling such as soil nutrients. The affected biomass, microbial communities, and soil structure as a result of trampling have the ability to alter nutrient cycling. De Graaf et. al., (2009) found that soil ammonium was negatively correlated with plant diversity in dry European heathlands whereas in wet

European heathlands, phosphorus was the most important factor and was negatively correlated with plant diversity. Grass species, such as *Deschampsia flexuosa* tend to be more resistant to ammonium and can outcompete dwarf shrub species when ammonium is high (Van Den Berg et. al., 2005). Many ericaceous species have a preference for ammonium as the concentration of ammonium tends to be higher in more acidic soils (Bardon et. al., 2018). Soil acidity and moisture are also very important factors in barren environments (De Graaf et. al., 2009). Germination of heathland species in Norway can be reduced by decreasing the pH to less than 5 and by the addition of aluminum (Roem et. al., 2002).

Another large issue with trampling includes soil erosion (Pickering & Norman, 2017). Soil erosion is a natural process by which soil particles are slowly removed from an environment over time through wind action, water, or some other natural processes. Vegetation cover is very effective at keeping the soil from eroding too drastically by providing protection from the elements and anchoring it in place with its root network (Ellis, 2017). Thus, when vegetation cover is reduced via trampling events or otherwise, exposed topsoil can be rapidly eroded away. Further erosion can damage remaining plant life and continue to affect the area in a positive feedback loop.

Many factors can influence the rate of erosion. Trail topography should be considered where soil will erode more quickly via gravity when slopes are steeper than 12° (Marion & Leung, 2004) and whether the trail passes through depressions where water is likely to accumulate. Climatic conditions are important such that increased rainfall events intensify soil erosion where water can carry soil particulate offsite (Ellis, 2017). Also, soil characteristics are an important consideration where organic soils are more prone to erosion than mineral soils and are particularly susceptible when moisture content is high (Bryan, 1977).

2.1.3 Impacts on seed banks

There are two major sources of potential passive regeneration following a disturbance event: clonal vegetative regeneration from surviving branches and roots and germination of seeds residing in the soil profile in seed banks (Piessens et. al., 2005). Clonal revegetation can be a very slow process, particularly in barren or alpine habitats where growing conditions are restricting. Alternatively, seed banks are a known repository of genetic material, generally found in the upper centimeters of topsoil (Putwain & Gilham, 1990; Miller et. al., 2017) that have the potential to contribute to future generations. They have even been known to form on extensive green roofs (Vanstockem et. al., 2018). They can also contribute to a population's resilience and can improve its ability to recover (Måren & Vandvik, 2009).

Seed banks are representations of past generations and contain seeds from earlier successive states which do not always match the present vegetation (Warr et. al., 1993, Shang et. al., 2013). Thus, it is possible to identify past successive states by analyzing soil seed banks. To add, seed bank density and richness have been found to decrease with successional maturity (Warr et. al., 1993), and in general, over time (Bossuyt & Hermy, 2003). Therefore, a seed bank cannot always be inferred by observing aboveground vegetation as it depicts a unique assemblage of past species from the area as well as those that are currently present.

The length of time the seeds can stay in the soil also plays an important factor in determining the assemblage and role of the seed bank in the recovery process. In general, there are two seed bank strategies: transient and persistent (Thompson & Grime, 1979). Transient seed banks are seeds that stay in the soil for less than one year and tend to germinate following predictable seasonal disturbances, whereas persistent seed banks survive longer than a year and are a source of regeneration when disturbances, in space and time, are unpredictable. To add, seeds that get buried deeper in the soil also tend to be long-lived, likely due to the time it takes for seeds to penetrate the soil profile (Bekker et. al., 1998). Seed longevity can affect the community's resilience such that transient seed banks that can only survive one or two growing seasons are likely to quickly die out of the seed bank if the seed source is removed (Måren & Vandvik, 2009). Understanding which species contribute to transient seed banks and which contribute to persistent seed banks can determine the extent to which a given area is able to recover following a disturbance.

It was found that the seed banks of northern heathlands in Norway are a potential source of regrowth following fire disturbance and not merely a byproduct of seed rain (Måren & Vandvik, 2009). This was due to large stores of seed from the most abundant heathland species of the area, *Calluna vulgaris*. However, in other areas seed banks may not contribute to regeneration. In Tibetan alpine grasslands, regeneration following the installation of a fenced enclosure was mostly by colonization capacity (seed rain) rather than from the seed bank (Shang et. al., 2013). Thus, the potential benefits of seed banks tend to vary greatly between environments.

Seedling recruitment in tundra habitat is considered low and most growth is clonal from the pre-existing vegetation (Huebner & Bret-Harte, 2018). Alpine species also tend to contribute poorly to seed banks (Shang et. al., 2013), and likely the case is similar for barrens habitat which can often have similar environmental conditions (Cameron & Bondrup-Nielson, 2013). Determining the presence of a seed bank in barrens is important to understand whether biotic or abiotic factors are most affected by trail degradation. The presence of a seed bank does not necessarily mean that environmental conditions are appropriate for seed germination and seedling survival.

There are two common techniques implemented to quantify seed bank density and richness: extraction and emergence. Extraction methods include sieving and flotation techniques to sort through soil material and identify all seeds. Emergence methods are defined by collecting soil samples and leaving them in appropriate conditions for germination to occur. Both techniques can characterize a portion of the seed bank, but neither method can accurately quantify the seed bank in its entirety. In general, both methods tend to find similar species richness in samples, but density is always significantly higher when using the extraction method (Gonzalez & Ghermandi, 2012; Abella et. al., 2013).

While extraction methods find more seed density than emergence methods (Gonzalez & Ghermandi, 2012; Abella et. al., 2013), they tend to be very time-consuming and are biased towards large-seeded species (Gonzalez & Ghermandi, 2012). There are also difficulties determining the viability of seeds (Abella et. al., 2013) and so it is uncertain to which degrees the extracted seeds contribute to the living seed bank. For the emergence method, all counted individuals must have come from viable seeds, however, the set of germinated species may not be wholly accurate such that different species have a different requirement of conditions necessary for germination to occur. Germination in a greenhouse is not always a suitable condition for all species present in the soil (Heerdt et. al., 1996; Abella et. al., 2013). The lack of knowledge on germination requirements for each species can inhibit the accuracy of emergence studies (Bossuyt & Hermy, 2003). To add, the emergence method requires a significant amount of time and space and so is not always a viable option (Heerdt et. al., 1996).

2.1.4 General impacts on flora and fauna

Overall, trail degradation is marked by a reduction in vegetation cover with a subsequent increase in bare ground, which leads to a reduced performance of ecosystem functions. The impacts on trails vary between different use types (hiking, horseback riding, mountain biking, ATVs, etc.) and the degree of their impacts vary with site-specific characteristics and intensity (Havlick et. al., 2016). Vegetation and soil structure become altered. In areas where degradation has occurred, fewer facilitative shrubs are found (Ballantyne & Pickering, 2015), there are more occurrences of non-native species (Pywell et. al., 1997; Wolf & Croft, 2014), species richness is affected by distance from the trail and degree of trampling (Parikesit et.al., 1995), and canopy height decreases (Liddle, 1975; Korkanç et. al., 2014; Mason et. al., 2015).

Another important consideration is the effects of fauna on barren ecosystems. Barrens can contain diverse populations of wildlife and often numerous bird species (Bried et. al., 2014) thus highlighting the importance of conserving both faunal and floral diversity of these regions. To add, invertebrates have been found to be linked to soil nutrient conditions (Culver & Beattie, 1983; de la Peña et. al., 2012). Culver and Beattie (1983) showed that burrow ants (*Formica* canadensis) in Colorado barrens significantly increased the amount of important nutrients, mainly potassium and phosphorus, in nearby soils. de la Peña et. al. (2012) found that soil conditions in barrens have been linked to altered interactions of pollinators with barren plants. Therefore, a loss of barren habitat can lead to a potential loss of nesting habitat and a loss of a food source for pollinators, birds, and others.

2.1.5 Experimental trampling studies

Many experimental trampling studies have been conducted to quantify the impacts that foot traffic has on vegetation. Most trampling studies have been conducted following the protocol described by Cole and Bayfield (1993) or using a similar framework of simulated trampling events increasing of increasing intensity of 0 passes up to 500 passes. While experimental trampling studies do impact the natural environment, they can infer some cause and effect relationships through controlled manipulations of the environment that descriptive, non-invasive surveys generally cannot account for (Cole, 2004).

Trampling studies of barrens, heathland and alpine vegetation have been conducted in many regions around the world including the United States (Bell & Bliss, 1973), France (Gallet & Rose, 1993), and Australia (Whinam & Chilcott, 2003; Scherrer & Pickering, 2006; Mason et. al., 2015). It was found as few as 30-100 passes will cause lasting damage for up to four years (Bell & Bliss, 1973; Whinam & Chilcott, 2003;

Mason et. al., 2015). This further establishes the importance of pre-emptively protecting these sensitive areas as well as implementing either passive or active restoration strategies when they do become degraded.

2.1.6 Project Objectives

To understand the effects associated with trail degradation in Cape Breton Highlands National Park, the project aimed to assess vegetation damage in terms of vegetation cover, trail characteristics, and soil characteristics. Vegetation, or biotic, factors were examined such as seed bank size, as well as species-specific vascular plant cover, and general moss and lichen cover which were visually estimated both on and off the hiking trails. Soil factors including substrate depth, compaction, surface temperature, moisture, and substrate nutrient content were measured and analyzed. The objective was to determine which factor(s) show the most difference between on- and off-trail conditions. The gathered information was then used to make informative decisions about potential restoration treatments in Chapter 3.

2.2 Methods

2.2.1 Study sites

Cape Breton Highlands National Park, located in Northern Nova Scotia, was established in 1936 following the construction of the Cabot Trail and protects 950 km^2 of land containing a mix of Acadian, Boreal, and Taiga forests and is the largest national park in the Maritimes (Parks Canada, 2010). The region experiences long fall and winter seasons due to its high altitude, coastal proximity, and strong winds, known locally as Les Suêtes, wherein gusts of wind often exceed 90 km/h during the winter (McIldoon & Pilon, 2008; Parks Canada, 2010). Some barrens within the Cape Breton Highlands

National Park have evidence of past fire (Bridgland et. al., 2011). The combination high altitude, high winds, and coastal influences have allowed for the formation of barrens habitat within the Park, which often are very scenic and attract many visitors.

Visitor numbers to the Cabot Trail slowly declined in the 1990s and were stagnant throughout the 2000s. The park and surrounding area began exploring different ways to revitalize the tourism and cultural economy of the region (Lemky, 2017). In 2008, there were roughly 175 000 visitors to the Park (Parks Canada, 2010). Since 2012, visitor numbers have increased every year and, from April 2017 to March 2018, they reached more than 330 000 visitors (Parks Canada, 2018). Increased visitor numbers have increased Park staff concern for trail degradation throughout the Park, but especially at the Skyline hiking trail and the Mica Hill hiking trail.

Figure 2.1. Map of Cape Breton Highlands National Park, depicting locations of the two hiking trails examined. Blue (left) is the Skyline hiking trail and red (right) is the Mica Hill hiking trail. The map was created using Google maps.

Skyline hiking trail:

The Skyline hiking trail is easily the most highly trafficked trail in Cape Breton Highlands National Park, where it easily receives thousands of visitors every season (Park staff, personal correspondence). Located on the west coast of Cape Breton, roughly 20 km north of Chéticamp, the trail is found in a mountain flank region with many hydraulically active faults (Baechler & Boehner, 2014) overlooking the Gulf of St. Lawrence. The exposed area near the end of the 6.5 km hiking trail is a coastal headland barren with shallow soil and is dominated by heathland vegetation and most notably bearberry, *Artostaphylos uva-ursi*.

In recent years, the trail has received many improvements to accommodate the growing number of visitors. In the early 2000s, the main portion of the trail was given a hardened surface with improved drainage, and wooden boardwalks were added in the most sensitive sections including the coastal headland section at the end of the trail. The main area of concern is a section of eroded path that extends beyond the last viewing platform of the boardwalk, where foot traffic has eroded the vegetation down to bare soil (Figure 2.2). This section was part of the original trail but was officially closed off when the boardwalk was installed but has continued to see foot traffic at unknown rates. Observations by Parks Canada staff noted that erosion and trail widening in this area has become increasingly apparent since 2016. This is the area where the extent of the vegetation damage was assessed, and restoration treatments were implemented (see Chapter 3).

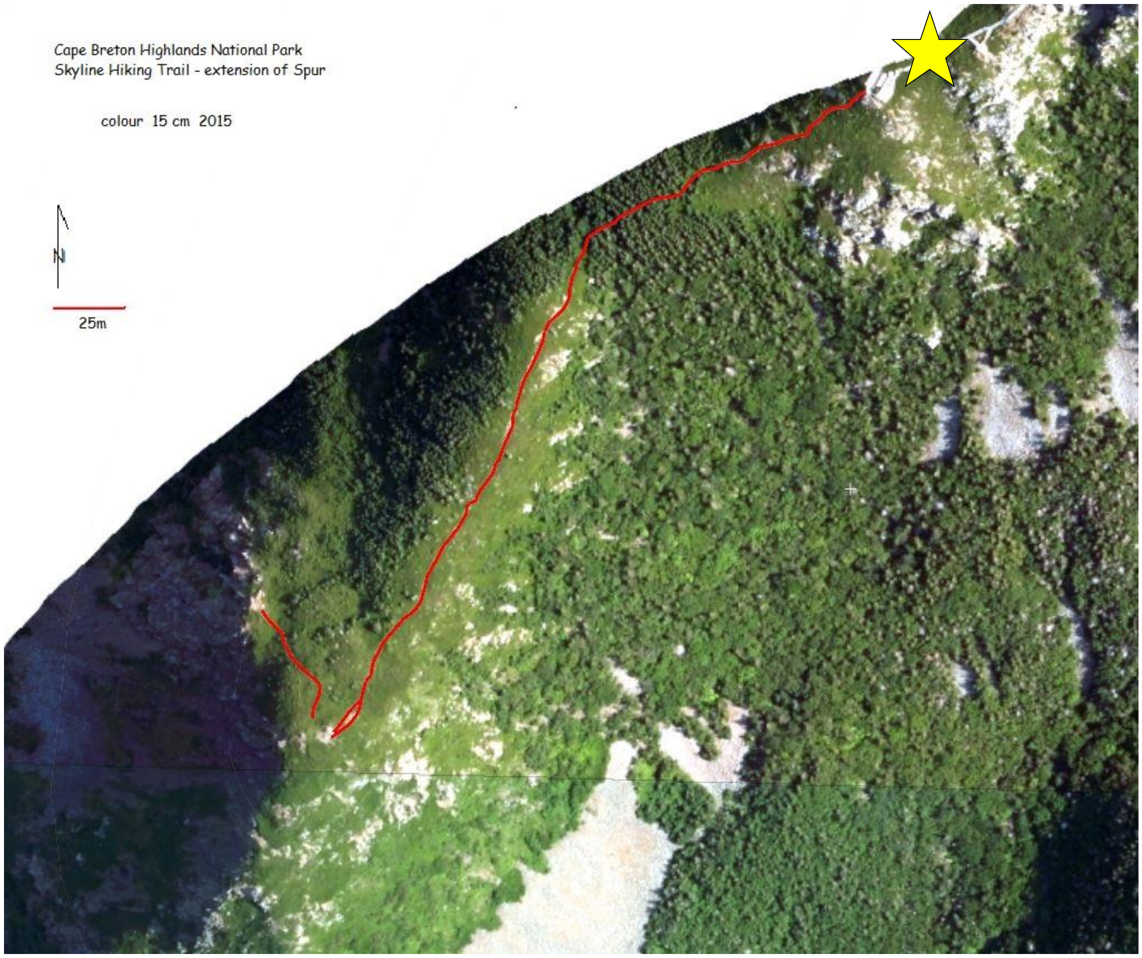

Figure 2.2. Aerial imagery, in colour at a resolution of 15 cm, of the degraded trail site at the Skyline hiking trail, 2015. The red line marks the eroded trail that goes beyond the last viewing platform of the boardwalk (yellow star). This is the portion that has been officially closed off.

Mica Hill hiking trail:

The Mica Hill hiking trail, formerly known as the Glasgow Lake trail, is in the northern region of the Park, about 10 km east of Cape North. Originally created as a fire road in the late 1960s, the trail runs through highland barrens dominated by stunted black spruce heath. Initially the trail went all the way to Glasgow Lake, but water drainage issues forced the trail to be shortened and in the past few years it has been further shortened and rerouted south to Mica Hill around the worst eroded sections. The new section of trail serpentines around the old one and creates four distinct sections of decommissioned trail (a, b, c, and d, Figure 2.3). There has been no natural revegetation in these eroded sections since the trail was rerouted, and this is the area where the extent of the damage was assessed. Assessments were conducted for all four sections of decommissioned trails, however, due to time and cost constraints, restoration treatments were not implemented at this site. Further monitoring of water-related impacts should be assessed to fully understand its implications.

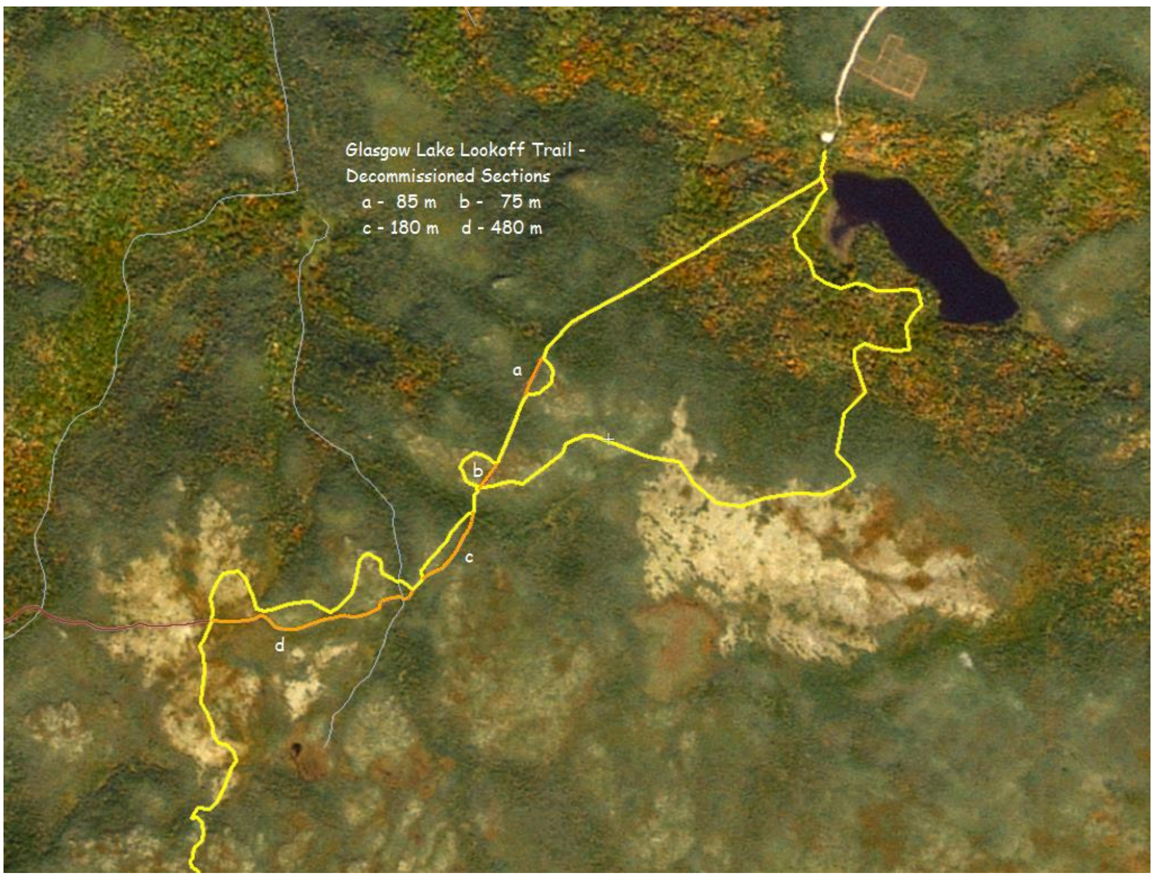

Figure 2.3. Aerial imagery of the Mica Hill hiking trail. The yellow line represents the current active sections of the official trail. The burgundy line represents a former portion of the Glasgow Lake trail (Glasgow Lake is not visible on this map). The orange lines represent the decommissioned sections which were the areas observed.

2.2.2 Data collection

Assessment of the vegetation as well as trail conditions were completed during the Summer of 2018. Conditions that were observed included trail factors (slope, width, and depth), abiotic factors related to the substrate (depth, surface temperature, moisture, nutrients, as well as exposed soil, gravel and rock), and biotic factors related to the vegetation (canopy height, vascular species, mosses, and fruticose lichens). Water-related issues were not monitored at Mica Hill due to timeframe and equipment constraints.

Trail factors were measured at intervals for the entirety of the designated study areas for both sites. For every 10 m section of closed-off trail, trail slope was recorded at the midpoint using an inclinometer on a compass and trail depth was recorded at the centre of the trail using a tape measure. One end of the tape measure was placed at the edge of the trail and the other end was drawn to the centre of the trail, in line with the trail edge, and there it was bent at 90° until it contacted the bottom of the trail. The depth was estimated, in cm, as the distance from the bottom of the trail to the 90° bend. Trail width was recorded three times for every 10 m of trail from one edge of the trail to the other. Trail edge was visually noted where an abrupt shift in vegetation height and density occurred.

To measure abiotic and biotic factors, paired 0.5 m x 0.5 m quadrats were set up every 10 m segment of trail such that one quadrat was placed directly in the center of the trail to represent the conditions of the trail and a second quadrat was placed 3 or 5 m away from the edge of the trail to serve as a reference to represent pre-disturbance conditions. Quadrat size was determined based on the size used by Parks Canada staff to collect similar data (personal correspondence) where the quadrat was able to fit completely within the trail with little to no edge overlap for the majority of locations. In areas where the trail width exceeded 5 m, two additional quadrats were placed on the trail and the subsequent data was averaged across the three.

At both sites, reference plots were placed to assess the natural vegetation that could represent a target for trail restoration efforts. At Mica Hill, reference plots were placed perpendicular to the trail section, 5 m from the trail away for all sections except for section c which went through a densely treed area such that it was not feasible to go 5 m without taking a substantial amount of time. Reference plots were located 3 m away from the trail edge instead. Reference plots were also placed uphill to avoid any potential runoff from the trail that might influence conditions. They were placed on the downhill side only when the uphill side was less than 5 m away from the active trail. At Skyline, for safety reasons, reference plots were placed 3 m away from the edge, where any further would have been too close to the edge of the cliff of the headland, and were placed, when possible, on the ‘inner side’ of the headland. For every plot (on the trail and in the reference vegetation), soil characteristics and vegetation variables were recorded. Environmental factors included soil depth, soil compaction, soil type, rock, gravel and soil exposure, fruticose lichen and moss cover, substrate moisture, and substrate surface temperature.

Substrate (abiotic) factors:

Rock, gravel, and soil exposure were estimated visually as percent cover for every quadrat. Substrate was considered gravel when the particle size of the substrate was from

2 to 63 mm, according to the International Organization for Standardization (ISO, 2017), soil was considered to be any particle < 2 mm, and rock was considered to be any particle > 63 mm. Rock, gravel, and soil can become exposed with repeated disturbances and negatively affect plant communities particularly with respect to root structure.

Substrate compaction and substrate depth were measured randomly in three different locations within each plot. Substrate compaction was recorded using a penetrometer up to a maximum of 4.5 kg/cm^2. Compaction was not taken for exposed rock, and when compaction was greater than 4.5 kg/cm^2, it was simply recorded as 4.5 kg/cm^2. Compaction is one of the most common effects of trail disturbance (Leung & Marion, 1996) and can impact seedling root development (Bassett et. al., 2005). Substrate depth was recorded by inserting a long, thin 40 cm metal rod into the substrate. Measurements that were greater than 40 cm were simply recorded as 40 cm. Barrens tend to have already shallow substrate, and with disturbance comes the potential for substantial loss of substrate. Substrate type was identified visually as either being predominantly organic substrate or mineral substrate. Organic substrates tend to be more prone to erosion, especially when moisture content is high (Bryan, 1977).

Substrate moisture and substrate surface temperature were recorded at every plot and were repeated approximately every three weeks for a total of five times over the course of the growing season. At Skyline they were recorded on May 31st, June 20th, July 7th, July 30th, and August 15th. At Mica Hill, they were recorded on June 5th, June 22nd, July 13th, July 31st, and August 17th. All measurements were taken within 3-5 days of a rainfall event in attempt to capture a high degree of on-site variability as the rates of moisture loss may vary between the exposed trail plots and vegetated reference plots. Substrate moisture was recorded in mV using the 10HS Soil Moisture Sensor and the Decagon Inc. Procheck. When the substrate was too shallow for the sensor, moisture was recorded as NA. Substrate surface temperature was recorded using a single Traceable® thermometer to the nearest 0.1 °C between 10:00 am and 2:00 pm. Soil compaction can alter substrate moisture regimes where it can cause pooling and poor drainage in some areas or reroute water away from others. Surface temperatures tend to fluctuate more dramatically on exposed trails where daytime temperatures are much higher and nighttime temperatures are much colder compared to undisturbed vegetation (Liddle, 1997). This large range of temperature can make it more difficult for seedlings to survive.

Substrate nutrients were also analysed. 500 mL samples were taken from 10 locations at both Skyline and Mica Hill in August 2018. Locations were selected prior to going out in the field on the day of collection and were made to be evenly distributed across the closed sections of trail. At each location, one sample was taken next to the trail plot and another was taken next to the reference plot. In several cases, soil was taken from several dig spots near the plot so as not to create too large a disturbance in any single area. The samples were then frozen for preservation before being sent to the Nova Scotia Analytical Laboratory in Truro, Nova Scotia. Samples were analysed for total nitrogen (%), pH, buffer pH, organic matter (%), P_2O_5 (kg/ha), K_2O (kg/ha), calcium (kg/ha), magnesium (kg/ha), sodium (kg/ha), sulfur (kg/ha), aluminium (ppm), boron (ppm), copper (ppm), iron (ppm), manganese (pp), zinc (ppm), cation-exchange capacity (meq/100 g) LR $CaCO_3$ (laboratory reagents calcium carbonate, t/ha to pH 6.5), as well as base saturated potassium, calcium, magnesium, sodium, and hydrogen (%).

Vegetation (biotic) factors:

Moss and fruticose lichen cover were estimated visually as percent cover for every plot. Fruticose lichen can grow on several substrates and is not restricted to rock outcrops. There are also an important indicator group for other environmental factors such as air pollution (Gibson et. al., 2013). Moss cover was grouped as either sphagnum moss which is generally indicative of wetter habitats, or ‘other’. Mosses and lichens were not identified down to the species level due to the amount of time and expertise needed for accurate identification.

Other biotic factors included vegetation canopy height and vascular plant cover. Canopy height was measured as the overall vegetation height in each quadrat, estimated to the nearest cm. Vascular plant cover was estimated as percent cover for each species. As percent cover was estimated individually for each species, it was possible that total cover for a single quadrat could exceed 100% due to species overlap.

Seed bank emergence:

Soil samples were collected on May 24th, 2019 at Mica Hill hiking trail, and on May 29th at the Skyline hiking trail. At each site, five locations were sampled as evenly as possible across the closed sections of trail. Again, these locations were selected prior to going out in the field on the day of collection. At each location, a 500 mL soil sample was taken from the top 5 cm layer both on disturbed trail and adjacent to the trail in the undamaged vegetation considered reference vegetation (minimum 5 m away from the trail edge at Mica Hill, and minimum 3 m away at Skyline). Samples were left to air dry for one week before being sieved through a 4 mm sieve to remove any larger particulate and concentrate the sample. Samples were then spread out in germination trays which were comprised of a bottom layer of 2-4 cm of commercial bagged topsoil (Compliments Black Earth Topsoil) and a surface layer 0.5 cm layer of sand to act as a barrier between the commercial topsoil and the sample.

Five replicates each of positive and negative controls were added to the study. The negative control was simply trays lined with the commercial bagged topsoil with the surface layer of sand. A negative control was included to allow for differentiation between seedlings emerging from the collected seed banks and the seedlings deriving from local seed rain at the growing site. The positive control involved mixing a known quantity of seed with 300 mL of the commercial topsoil and spreading it in the same manner as the soil samples on top of the layers of commercial topsoil and sand. This was included to determine whether the experimental conditions were conducive to seed germination for common species found in the extant vegetation at both trail sites. The added seed included: 300 seeds of both *Danthonia spicata* and *Deschampsia flexuosa*, and 150 seeds of both *Vaccinium angustifolium* and *Sibbaldiopsis tridentata*, for a total of 900 seeds per tray.

All trays were covered by plastic domes and placed in a randomized block design on the northwest facing side of the Parks Canada Chéticamp staff house for 14 weeks while germination occurred. Trays were observed every 3-5 days where identified individuals were recorded and removed. When identification was difficult, individuals were removed and potted until potential identification was possible. It is important to note that the plastic domes had to be removed for the majority of the second half of the study as daily air temperatures were high and seedlings risked over-heating. After 14 weeks, germination had not stopped, but due to time constraints, the experiment was ended.

2.2.3 Statistical analysis

Data were summarized and analyzed using R 3.6.1. Data analysis and model selection were based on the analysis of general linear models using a Bayesian approach based on the rstan and loo packages. The Bayesian approach was used because classic statistical tools used are often very specialized and so not able to handle many common research questions (McElreath, 2015), and there are concerns with the use of null hypothesis significance testing and the misinterpretation of *p*-values (Halsey et.al., 2015; Kruschke, 2015). Bayesian computation involves a process whereby initial prior beliefs about a model system are updated by data to produce an outcome of posterior probabilities that describe the relationship of factors specified within the model system (Kruschke, 2015; McElreath, 2015). Model diagnostics were performed and included: running multiple chains to check for convergence (Rhat = 1), ensuring an effective sample size, $N_{eff} > 5000$, plotting posterior distributions against priors to ensure the priors are appropriately overwhelmed, and generating a new dataset from the model and comparing it against the original to ensure the data are being appropriately modeled.

Substrate factors and nutrients

To compare substrate factors between sites that were on or off the trail, each factor was used in separate models as the dependent predicted variable and trail type (on- versus off-trail) was used as a categorical predictor variable. Specifically, the substrate characteristics examined were substrate depth, compaction, surface temperature, moisture, total cover as well as substrate nutrients: N, P_2O_5, K_2O, organic matter content, and pH. Model selection using the Widely Applicable Information criterion (WAIC) determined that effects field location (Skyline versus Mica Hill) were dependent on trail type, and thus were included as an interaction effect.

For each factor, a Bayesian model was designed with a continuous metric predicted variable. With a metric predicted variable, the appropriate likelihood is a Normal distribution, where the probability is based on the probability, *p*, and standard deviation, σ, which is estimated from the data. The standard deviation was allowed to vary by type (on-/off- trail). This was implemented in Stan using the normal likelihood function. Specifically, the likelihood equation used was:

$$p = \beta_0 + \sum_{j} \beta_{1[j]} x_{1[j]} + \sum_{k} \beta_{2[k]} x_{2[k]} + \sum_{j,k} \beta_{1\times 2[j,k]} x_{1\times 2[j,k]}$$

$$y \sim \mathrm{Normal}(p, \sigma)$$

Where:

β_0 represents the baseline average values across all predictor variables. $\beta_{1\times 2}$ represents the interaction effect as the deflection of being either on- or off-trail, *j*, dependent on field location of being either at Skyline or Mica Hill, *k*. β_1 and β_2 represent the deflection of being either on- or off-trail, *j*, and the deflection of being either at Skyline or Mica Hill, *k*, respectively, after accounting for the deflection of the interaction effect. For every model, *y* is the predicted substrate factor or nutrient: substrate depth, substrate compaction, surface temperature, substrate moisture, N, P_2O_5, K_2O, organic matter, or pH. For substrate depth and compaction were taken three times per plot, the data used for these variables were averaged for every plot. Since substrate moisture and temperature were repeated on five separate days and average daily conditions can vary greatly, only one day (July 30th for Skyline, and July 31st for Mica Hill) was used in the analysis. The model was tested with data collected from different days with similar results.

The prior probabilities for β_0, β_1, β_2, and $\beta_{1\times2}$were made hierarchical where each predictor variable was sampled from a higher order distribution. Thus, the prior means were sampled from a hyperprior with a mean centered at zero and a standard deviation of 1, and the prior standard deviations were sampled from a hyperprior with a mean centered at 1 with a standard deviation of 1 to maintain positive variance (Figure 2.4). Thus, they were assumed to have no effect but with little weight on this assumption and also allow for information sharing across categories. The prior probability for σ was a normal distribution with a mean and standard deviation both centered at 1 and was allowed to vary based on trail type. All predictor variables were standardized to a mean of 0 and standard deviation of 1, then de-standardized following MCMC sampling to return values to their respective original scales.

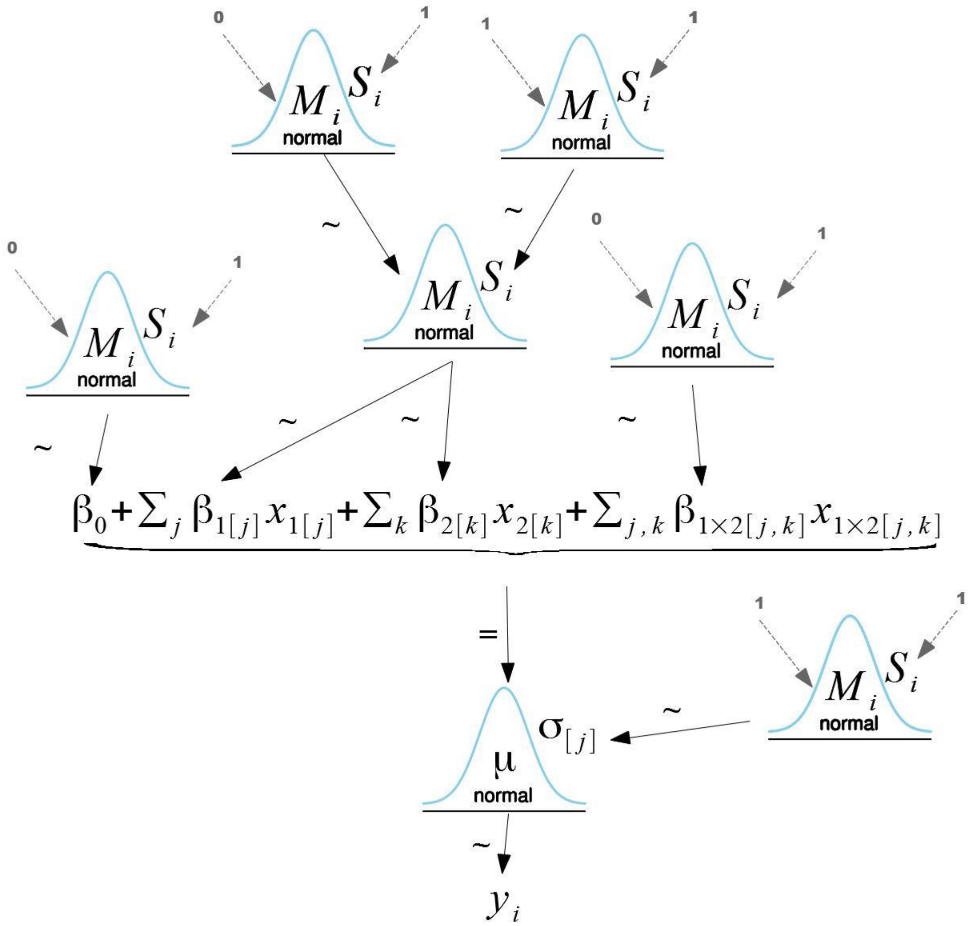

Figure 2.4. Diagram representation of Bayesian models used to compare the effect of on- and off-trail conditions and site location (Skyline versus Mica Hill) on various substrate factors and nutrients. Each factor/nutrient was assessed in a separate model using this framework.

Seed bank emergence

To compare the amount of seed bank germination for different sources, the total germination count was used as the predicted variable and seed source was used as the predictor variable. Specifically, seed source categories included on-trail and off-trail at Skyline, on-trail and off-trail at Mica Hill, as well as both positive and negative control group. Interestingly, AIC selection determined that effects of the blocked design were important while WAIC selection considered them to not. Therefore, the effect of block was included in the model.

A Bayesian model was designed with a count predicted variable where values were positive integers. With a count predicted variable, the appropriate likelihood is a Poisson distribution, where the probability of any number of germinations that will occur from each seed source is based on the probability, *p*, which is estimated from the data. As *p* can only be positive, the predictor variables must be linked to the predicted variable in a modified way. With a Poisson distribution, the appropriate link between the predictor and predicted variables is the log link, which ensures that the combined effects of the predictor variables are positive. This can be implemented in Stan using the poisson_log likelihood function. Specifically, the likelihood equation used was:

$$p = \sum_j \beta_{1[j]} x_{1[j]} + \sum_k \beta_{2[k]} x_{2[k]}$$

$$y \sim \text{Poisson\_log}(p)$$

Where:

β_1 is the effect of being from a particular seed source, *j*, (on-trail at Skyline, positive control, etc.) and β_2 is the effect of being in a particular block, *k*. All effects are thus on the log scale. Priors of both seed source and block variables were made hierarchical where each predictor variable was sampled from a higher order distribution. Thus, the prior means were sampled from a hyperprior with a mean centered at zero and a standard deviation of 1, and the prior standard deviations were sampled from a hyperprior with a mean centered at 1 with a standard deviation of 1 to maintain positive variance (Figure 2.5). This allowed for information sharing across categories. Additionally, the priors were assumed to have no effect but with little weight on this assumption.

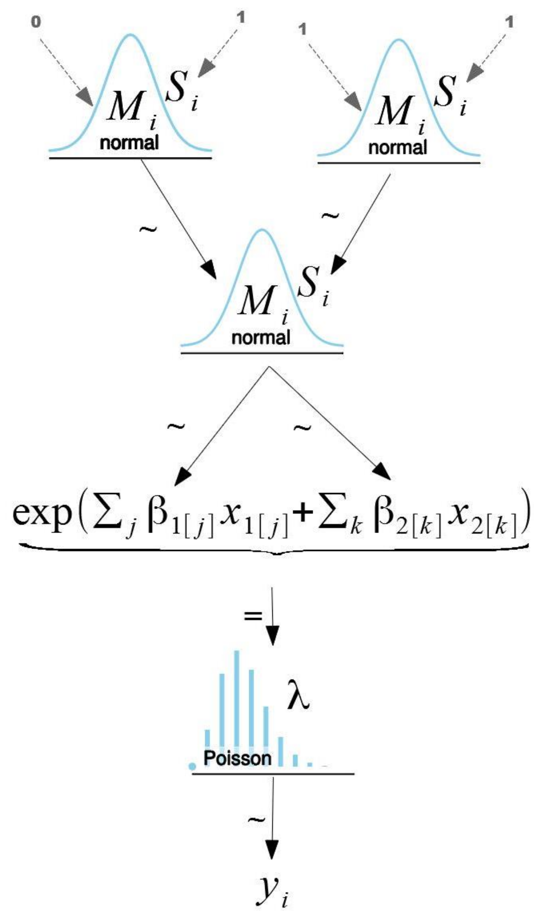

Figure 2.5. Diagram representation of Bayesian hierarchical model used to compare total germination counts across seed sources and controls.

2.3 Results

2.3.1 Trail characteristics

At the Skyline trail, the slope only exceeded 12° for the first 70 m and only for one sample observation at the Mica Hill trail. At Skyline, the trail was 430 m long with an average trail width approximately 1.25 m and spanned a total area of approximately 525 m^2. At Mica Hill, the decommissioned sections combined were roughly 979 m long, with an average trail width approximately 2.4 m, and spanned a total area of approximately 2400 m^2.

Table 2.1 shows the top-most abundant and present species found on- and off-trail at both Skyline and Mica Hill. Abundance was calculated as total percent cover across designated plots and presence was calculated as the number of plots it appeared in. At Skyline, trail and reference plots were very similar across abundance and presence, where the most abundant species were the same on- and off-trail: *Artostaphylos uva-ursi* and *Vaccinium angustifolium*¸ *Sibbaldiopsis tridentata*, *Deschampsia flexuosa,* and *Juniperus communis*. *Danthonia spicata* and *Solidago bicolor* were the only species that appeared more often on trails than in the reference vegetation.

At Mica Hill, there was a clear difference in species assembly on and off the trail. On-trail conditions included species that are more commonly associated with wet meadow communities, such as *Juncus brevicaudatus, Hypericum canadense,* and *Carex echinata*. In the reference vegetation, the most abundant and present species include *Picea mariana, Rhododendron canadense, Kalmia angustifolium,* and *Vaccinium* *angustifolium.* There is a clearer dichotomy of species assembly at Mica Hill, when compared to that of Skyline.

Table 2.1. Top 5 most abundant and most present species according to trail plots and control plots for Skyline and Mica Hill. Species are ordered from #1 being the most abundant/present, to #5 being the fifth most abundant/present.

SKYLINE			
Abundance (total percent cover)		Presence (number of plots where it was present)	
Trail	Reference	Trail	Reference
1. *Arctostaphylos uva-ursi* 2. *Vaccinium angustifolium* 3. *Sibbaldiopsis tridentata* 4. *Juniperus communis* 5. *Deschampsia flexuosa*	1. *Arctostaphylos uva-ursi* 2. *Vaccinium angustifolium* 3. *Juniperus communis* 4. *Deschampsia flexuosa* 5. *Sibbaldiopsis tridentata*	1. *Sibbaldiopsis tridentata* 2. *Vaccinium angustifolium* 3. *Danthonia spicata* 4. *Solidago bicolor* 5. *Deschampsia flexuosa*	1. *Vaccinium angustifolium* 2. *Deschampsia flexuosa* 3. *Sibbaldiopsis tridentata* 4. *Artostaphylos uva-ursi* 5. *Cornus canadensis*
MICA HILL			
Abundance (total percent cover)		Presence (number of plots where it was present)	
Trail	Reference	Trail	Reference
1. *Vaccinium angustifolium* 2. *Juncus brevicaudatus* 3. *Danthonia spicata* 4. *Carex echinata* 5. *Milium effusum*	1. *Picea mariana* 2. *Rhododendron canadensis* 3. *Vaccinium angustifolium* 4. *Kalmia angustifolia* 5. *Abies balsamea* 6. *Pteridium aquilinium* 7. *Gaylussacia baccata*	1. *Juncus brevicaudatus* 2. *Vaccinium angustifolium* 3. *Milium effusum* 4. *Danthonia spicata* 5. *Hypericum canadense*	1. *Rhododendron canadensis* 2. *Kalmia angustifolia* 3. *Vaccinium angustifolium* 4. *Picea mariana* 5. *Cornus canadensis*

Average cover types also varied on- and off-trail and between both field site locations (Figure 2.6). For both Skyline and Mica Hill, leaf litter predominated all cover types in the reference vegetation. At Mica Hill however, reference vegetation also constituted some cover of fruticose lichen, sphagnum moss and other moss while at Skyline these forms of cover were essentially non-existent. When considering on-trail conditions, again Mica Hill included more variety in types of cover including some degree of leaf litter, sphagnum moss, other moss, soil, gravel, and rock. At Skyline, the mosses were absent from the trail.

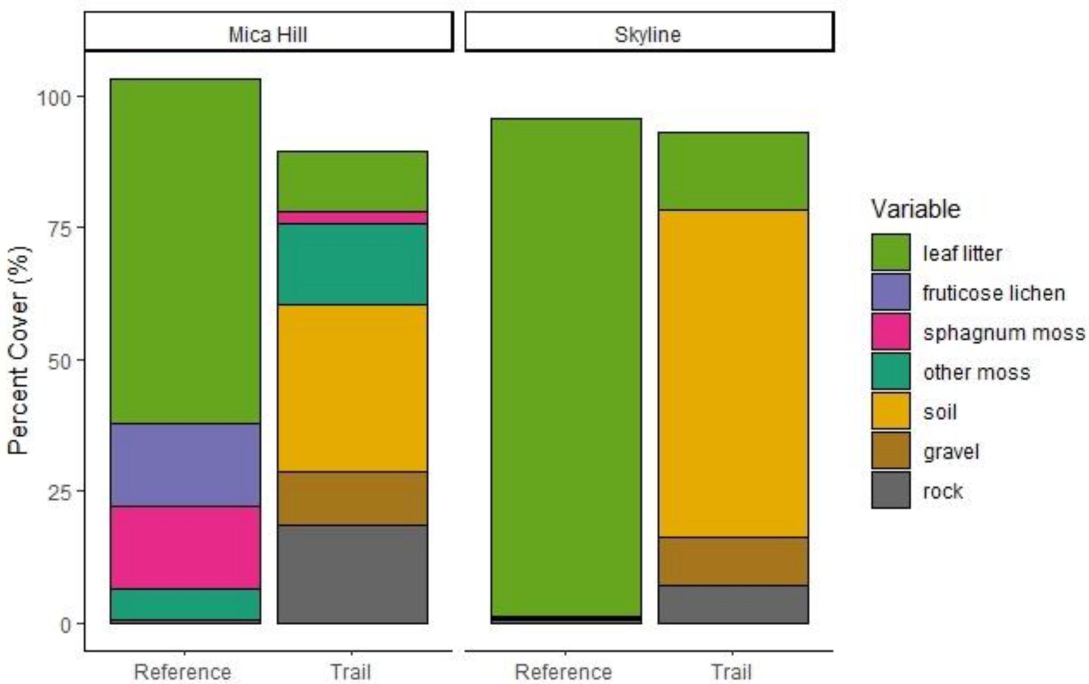

Figure 2.6. Summary of average percent cover variables for on-trail (Trail) and off-trail (Reference) plots for both Skyline and Mica Hill. Variables include leaf litter, fruticose lichen, sphagnum moss, other moss, as well as exposed soil, gravel, and rock.

2.3.2 Substrate factors

Raw data show there are some variations in depth, compaction, surface temperature and moisture between on-trail and off-trail conditions and between both Skyline (Figure 2.7) and Mica Hill (Figure 2.8). Modelling each independent response variable (e.g., substrate depth, surface temperature), against whether the value was taken on or off the trail either at Skyline or Mica Hill, allowed for site specific responses to be determined (Figure 2.9).

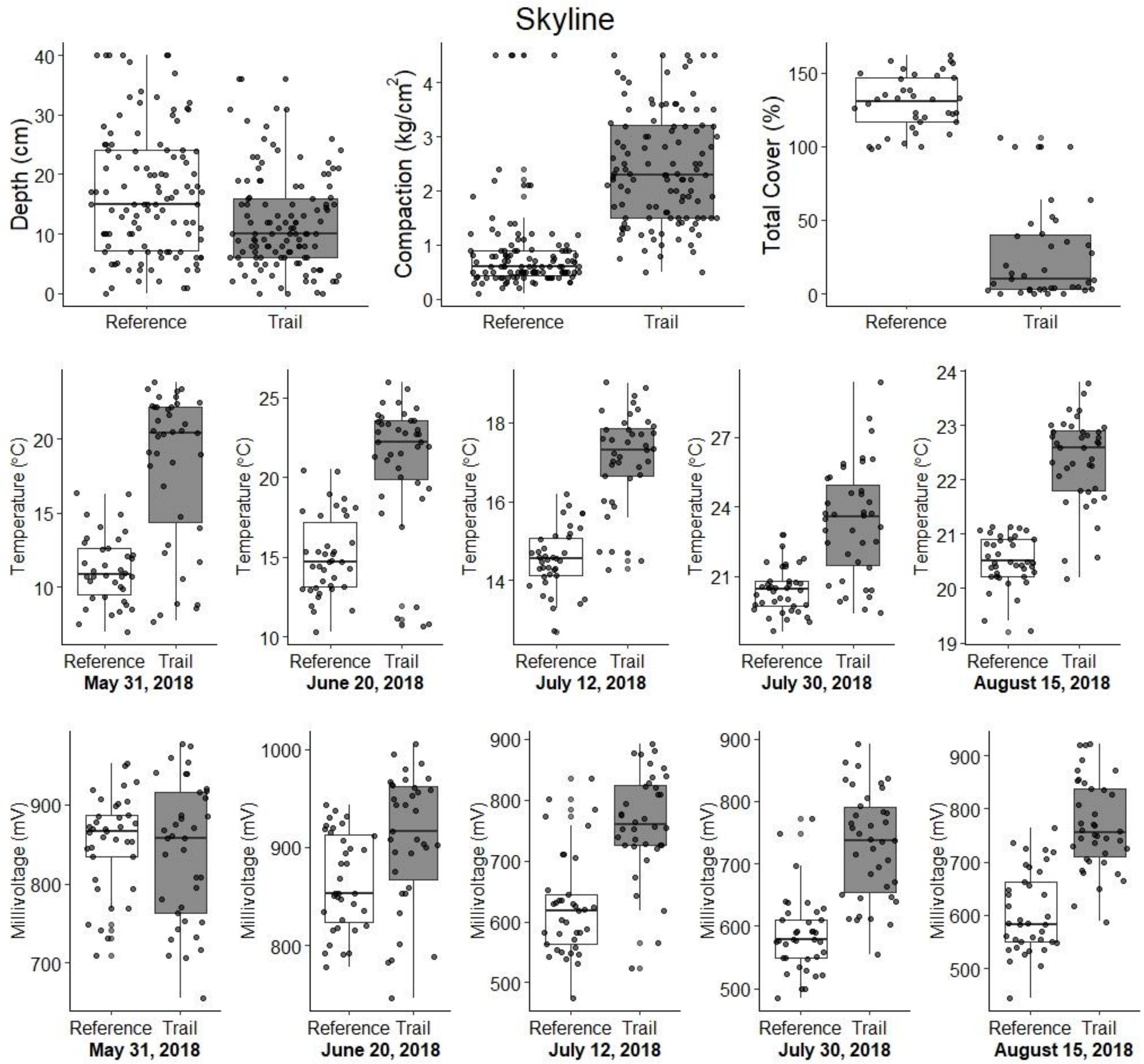

Figure 2.7. Boxplot summaries of on-trail (Trail) and off-trail (Reference) comparisons: substrate depth, compaction, vegetation cover, surface temperature, and moisture at the Skyline hiking trail. All data points are shown jittered over the boxplot. Depth and compaction measurements were repeated three times per plot. Temperature and moisture measurements were taken every 2-3 weeks on five separate occasions. These are graphed separately to account for daily temperature and moisture averages.

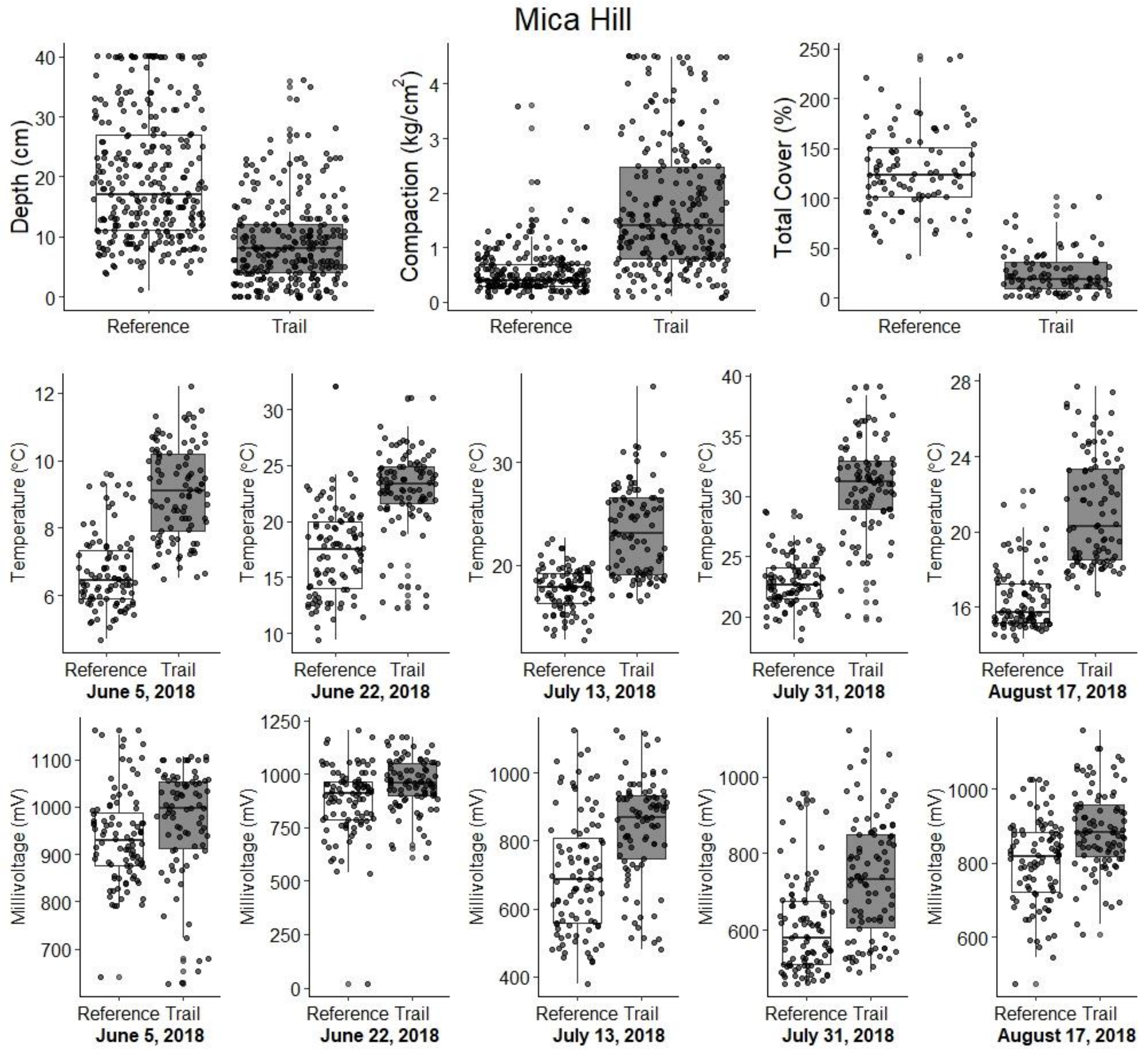

Figure 2.8. Boxplot summaries of on-trail (Trail) and off-trail (Reference) comparisons: substrate depth, compaction, vegetation cover, surface temperature, and moisture at the Mica Hill hiking trail. All data points are shown jittered over the boxplot. Depth and compaction measurements were repeated three times per plot. Temperature and moisture measurements were taken every 2-3 weeks on five separate occasions. These are graphed separately to account for daily temperature and moisture averages.

Overall, on-trail conditions had shallower depth, higher compaction, higher surface temperature, higher substrate moisture, and lower total cover, when compared to reference, off-trail conditions at both Skyline and Mica Hill (Figure 2.9). For depth, trails were, on average, 10.65 cm shallower at Mica Hill and 4.77 cm shallow at Skyline. For substrate compaction, trails were, on average 1.13 kg/cm^2 more compact at Mica Hill and 1.53 kg/cm^2 more compact at Skyline. For surface temperature, trails were, on average, 7.86 °C warmer at Mica Hill and 1.53 °C warmer at Skyline. For substrate moisture, trails were on average 122.25 mV higher at Mica Hill and 143.22 mV higher at Skyline. Finally, for total cover, trails were, on average, roughly 102 % lower for both Skyline and Mica Hill.

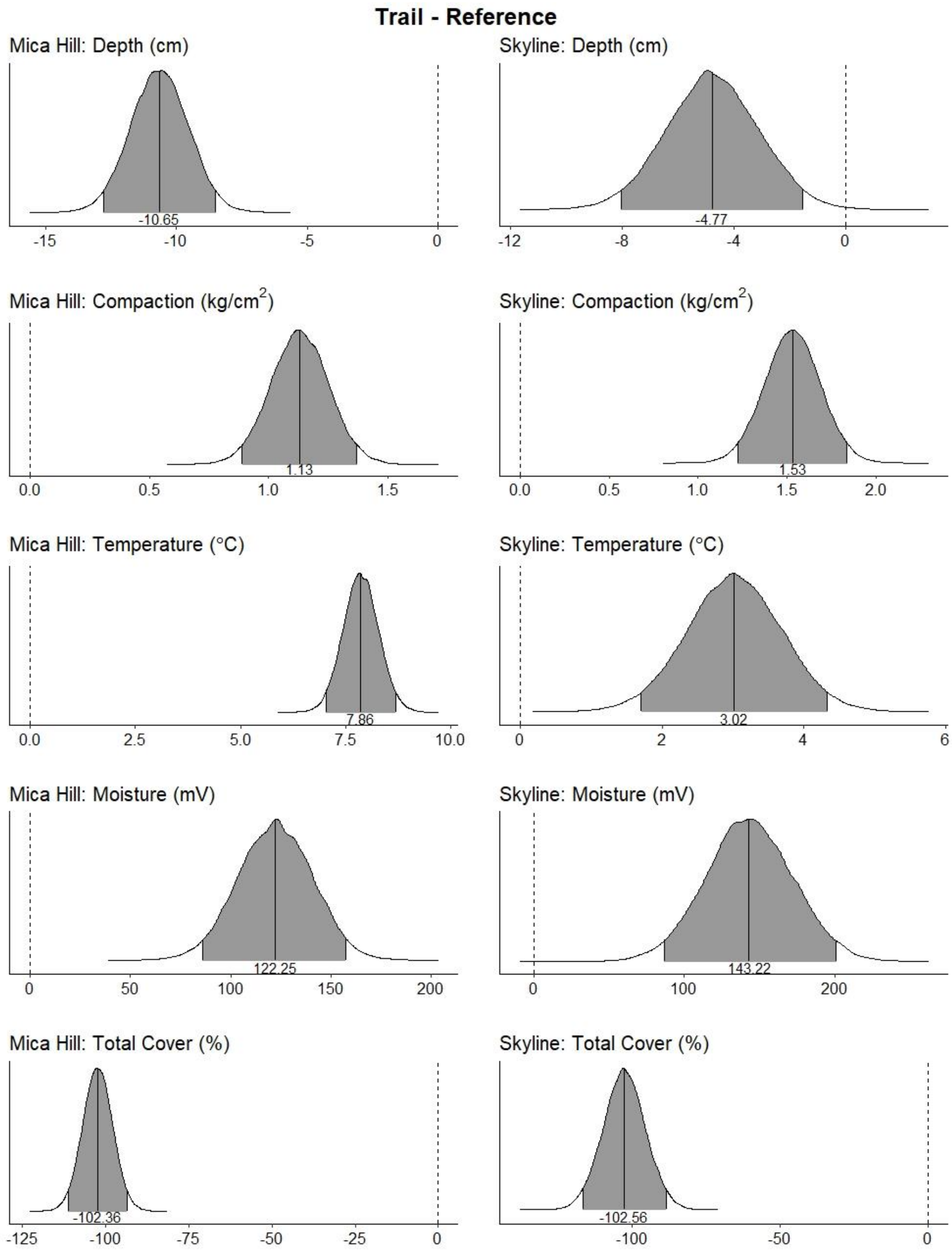

Figure 2.9. Comparing trail and reference conditions at Skyline and Mica Hill for substrate factors: substrate depth (cm), compaction (kg/cm^2), surface temperature (°C), moisture (mV), and total vegetation cover (%). Positive values indicate higher values on the trail and negative values indicate lower values on the trail. Zero values indicate no clear difference between the variable and each condition. Shaded regions represent 95% highest density intervals and displayed values represent the mean estimate.

Raw data shows there are some variations in substrate nutrients and pH between on-trail and off-trail conditions and between both Skyline and Mica Hill (Figure 2.10). Modelling each independent response variable (e.g., %N, K_2O), against whether the value was taken on or off the trail either at Skyline or Mica Hill, allowed for site specific responses to be determined (Figure 2.11).

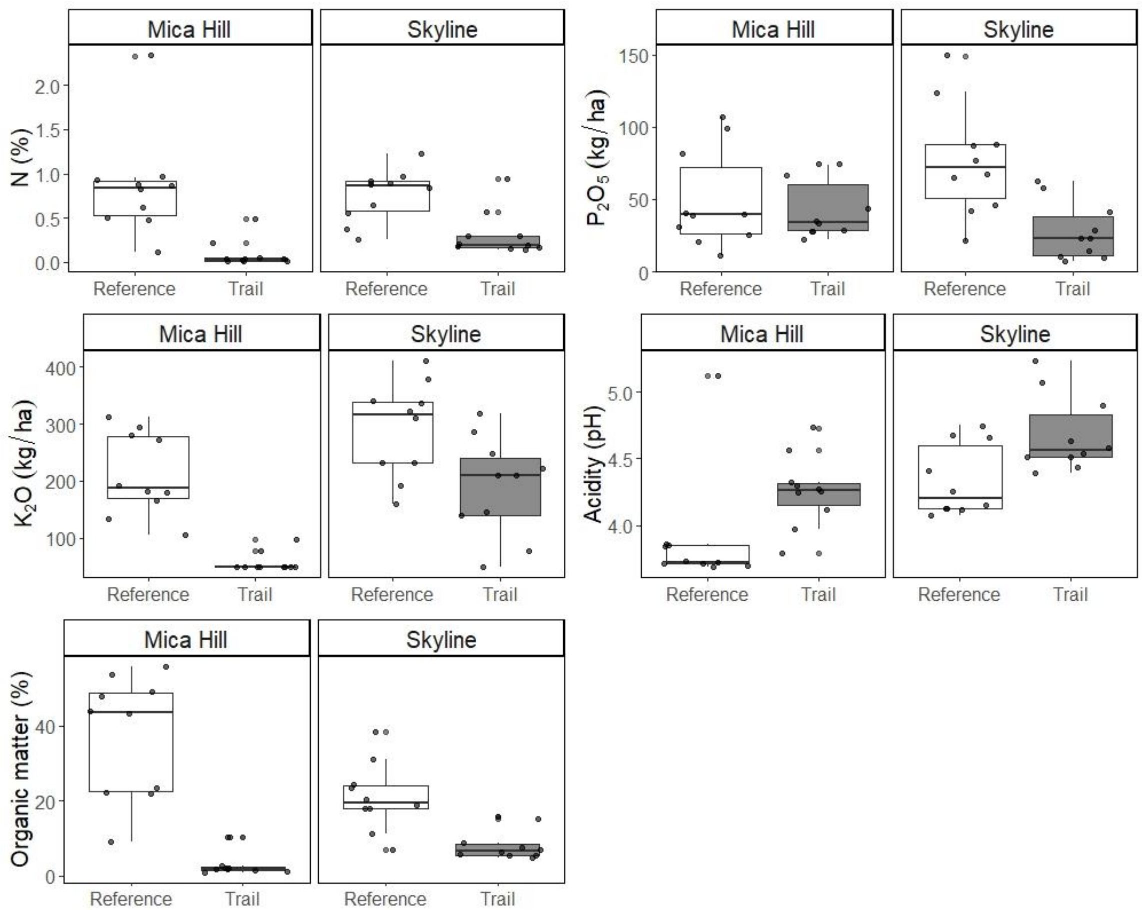

Figure 2.10. Boxplot comparison of on-trail (Trail) and off-trail (Reference) conditions of five major soil nutrients for both Skyline and Mica Hill hiking trails. Nutrients include: N, P, K, pH, and organic matter content. All data points are shown jittered over the boxplot.

Overall, on-trail conditions had lowered nutrient content and higher pH, when compared to reference, off-trail conditions at both Skyline and Mica Hill. For N, trails were, on average, 0.73 % lower at Mica Hill and 0.43 % lower at Skyline. For P_2O_5, trails were not different than the reference at Mica Hill, and, on average, 45.93 kg/ha lower at Skyline. For K_2O, trails were, on average, 151.53 kg/ha lower at Mica Hill and 98.79 kg/ha lower at Skyline. For organic matter content, trails were, on average, 32.64 % less at Mica Hill and 12.93 % less at Skyline. Finally, for pH, trails were, on average roughly 0.35 higher at both Mica Hill and Skyline.

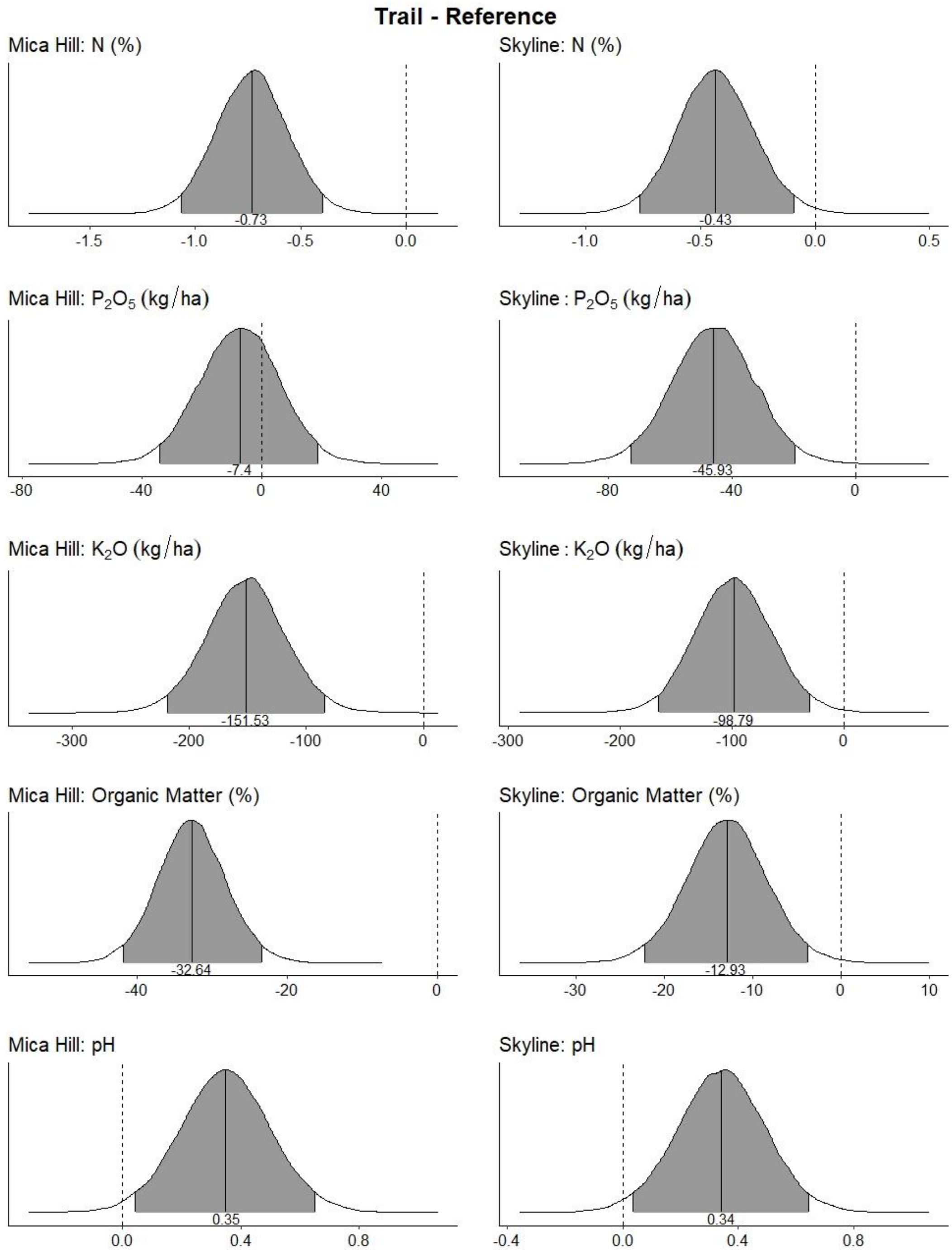

Figure 2.11. Comparing trail and reference conditions at Skyline and Mica Hill for substrate nutrients: N (%), P_2O_5 (kg/ha), K_2O (kg/ha), organic matter (%), and pH. Positive values indicate higher values on the trail and negative values indicate lower values on the trail. Zero values indicate no clear difference between the variable and each condition. Shaded regions represent 95% highest density intervals and displayed values represent the mean estimate.

2.3.3 Seed bank emergence

Overall, germination was low and the majority of species that were identified were weed species that were not present at either field sites (Figure 2.12). There were 580 germinations in the positive control trays where 4500 seeds had been sown. The summary count of seed sources, across all 5 replicates were: 154 for Mica Hill reference trays, 472 for Mica Hill trail trays, 159 for Skyline reference trays, 95 for Skyline trail trays, and finally, 93 germinations in the negative control trays. Due to the time constraints of the study, some species were only able to be identified to the genus level. The most abundant species found were: *Danthonia spicata*, *Vaccinium angustifolium*, *Deschampsia flexuosa*, *Sibbaldiopsis tridentata*, *Taraxacum officinalis*, *Hypericum canadense*, *Cerastium fontenum*, *Urtica urens*, and *Chenopodium album.*

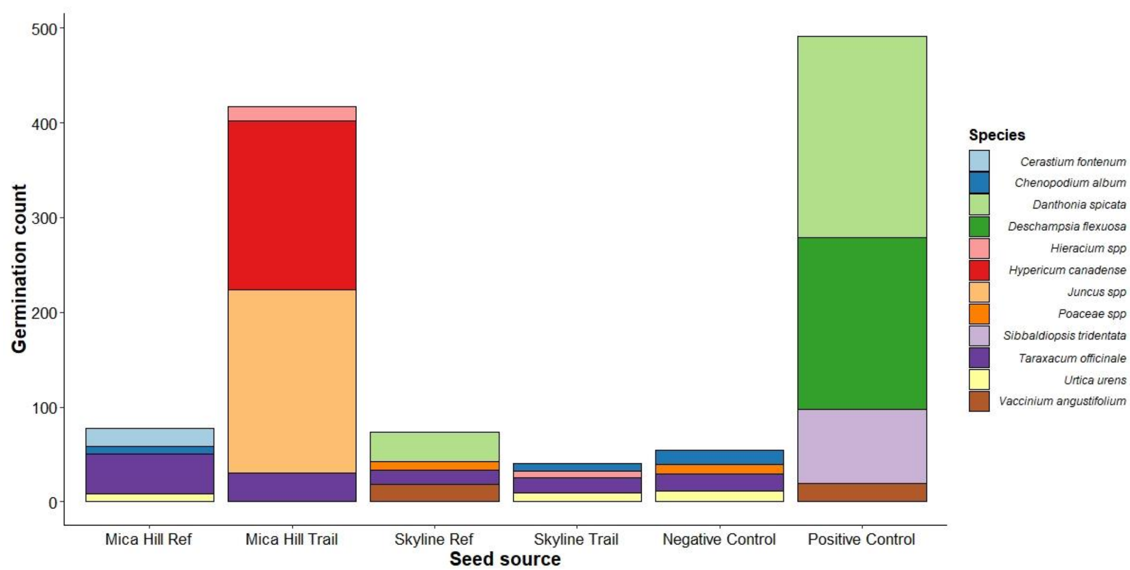

Figure 2.12. Total germination counts of the 4 most abundant species identified for each seedbank source: off the trail (Ref) and on the trail (Trail) from Mica Hill, off the trail (Ref) and on the trail (Trail) from Skyline, as well as positive and negative control groups.

To understand how posterior distributions relate to each other, the difference between them and the result is a distribution of the particular pairwise comparison. This is sampled across all values of the blocking variable. If the new distribution does not overlap zero, there is a difference between examined posterior distributions. Germination in the positive control was about 6 times higher than in the negative control (Figure 2.13). At Mica Hill, roughly 3 times as many germinations resulted from soil collected on-trail than off-trail and at Skyline, there were about 40% fewer germinations from soil collected on-trail than off-trail (Figure 2.14). Compared to the negative control, all seed sources had higher germination counts, except for that of the on-trail at Skyline, which showed no difference (Figure 2.15).

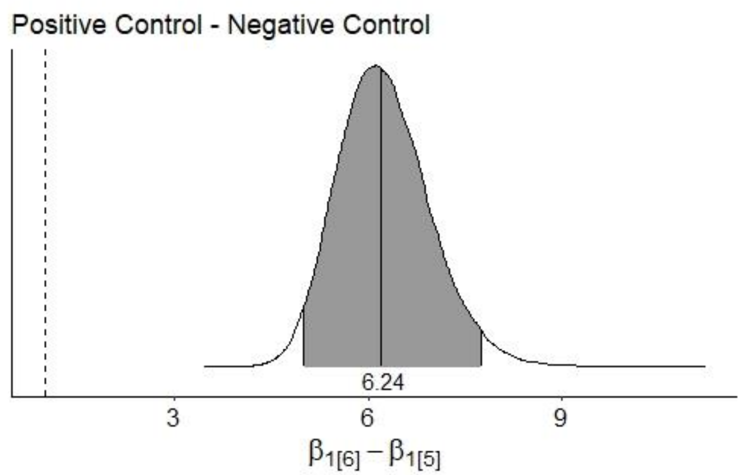

Figure 2.13. Difference in germination count between positive and negative controls. $\beta_{1[6]}$ is the marginal distribution of being in the positive control and $\beta_{1[5]}$ is the marginal posterior distribution for being in the negative control. Posterior distributions were exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a ratio of 1.

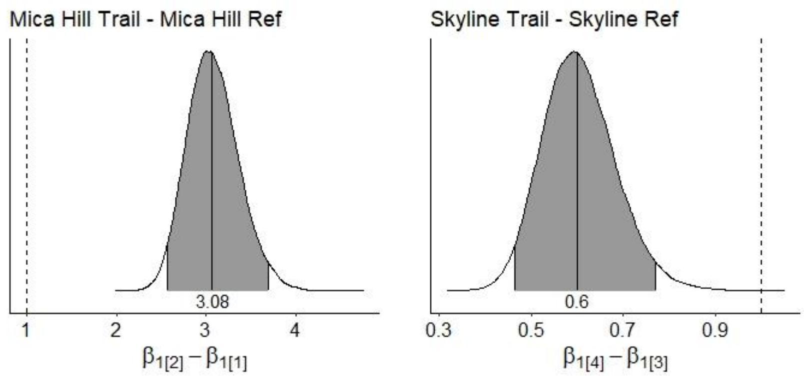

Figure 2.14. Difference in germination count between off-trail reference sites (Ref) and on-trail sites (Trail) for both sites. Posterior distributions were exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a ratio of 1. $\beta_{1[1]}$, $\beta_{1[2]}$, $\beta_{1[3]}$, and $\beta_{1[4]}$, are the marginal posterior distributions for Mica Hill reference, Mica Hill trail, Skyline reference, and Skyline trail, respectively.

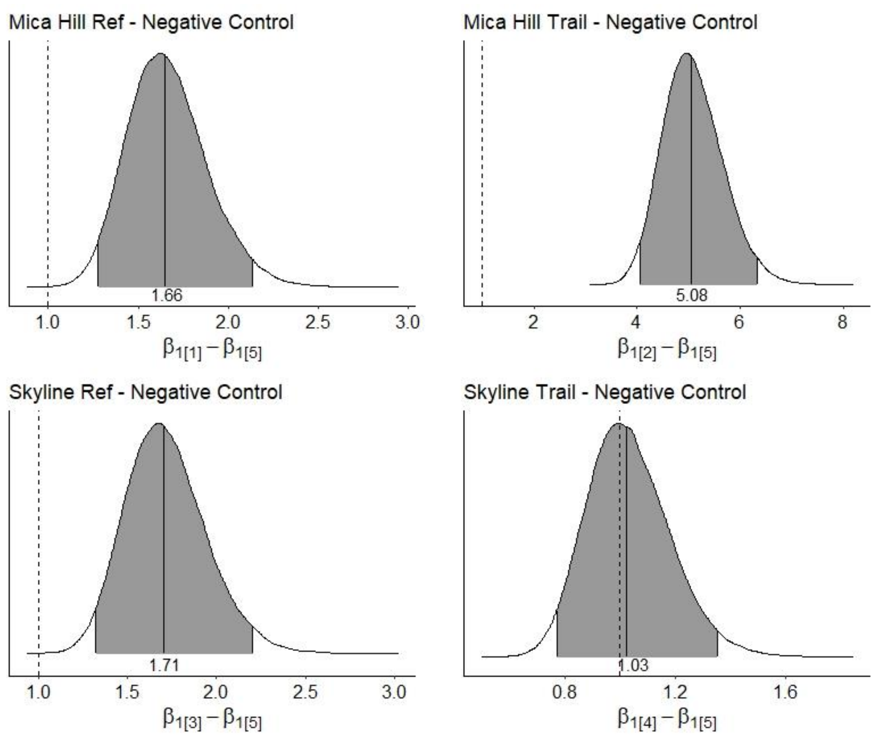

Figure 2.15. Total germination counts for each source compared against the negative control. Posterior distributions were exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a ratio of 1. $\beta_{1[1]}$, $\beta_{1[2]}$, $\beta_{1[3]}$, $\beta_{1[4]}$, and $\beta_{1[5]}$ are the marginal posterior distributions for Mica Hill reference, Mica Hill trail, Skyline reference, Skyline trail, and negative control respectively.

2.4 Discussion

Summary

Firstly, surface cover types between on-trail and off-trail conditions varied greatly (Figure 2.6) while total vegetation cover was reduced (Figure 2.9). At both sites, on-trail conditions were more varied with greater instances of exposed rock, gravel, and soil whereas off-trail conditions mainly consisted of leaf litter. This is not surprising at Mica Hill where the trail was previously maintained as a fire road and vegetation would have been cleared. Nor is this surprising at Skyline which receives thousands of visitors every tourist season and trampling of this closed area, even at low rates, has reduced vegetation cover and height and has led to the subsequent exposure of soil.

Secondly, impacts to the substrate were notable (Figure 2.9 & Figure 2.11). On-trail conditions included decreased depth, higher compaction, higher daily surface temperatures, and higher moisture content. In terms of substrate nutrients on-trail conditions were associated with lower nutrients and higher pH values.

Thirdly, while the number of germinations were low over the course of the experiment, seed banks were present in the reference conditions at both Skyline and Mica Hill (Figure 2.15). At Mica Hill, there was a noticeably larger seed bank on the trail (Figure 2.14), but the main two species that germinated (*Juncus spp* and *H. canadense*) were not found in the reference vegetation and represent a wet meadow community type. At Skyline, there was no difference in germination count between trail conditions and the negative control (no seed input) and so there was no viable seed bank on the trail at

Skyline. Therefore, on the trail at Skyline has no potential source of recovery via means of seed bank germination.

Trampling and compaction reduce cover

As trampling can contribute to lower resistance and resilience (Mason et. al., 2015), repeated trampling events can inhibit any regrowth of vegetation and soil exposure persists. There is potential for the exposed areas to expand in width every time someone steps slightly off the path. Additionally, trampling can often have a delayed impact where cover continues to be reduced up to 6 to 12 months following a trampling event (Whinam et. al., 2003). Particularly at Skyline where trampling is more frequent, the poorly resistant and poorly resilient coastal headland vegetation is quickly reduced and can potentially reduce further even after removal of trampling pressure, with little ability to recover naturally as the area in question will continue to be stressed .

Trampling is also the likely cause for increased soil compaction in on-trail conditions (Figure 2.9). Compaction can alter soil characteristics physically, biologically, and chemically (Tracy et. al., 2011). Trampling leads to compaction which reduces soil pore space and can even alter soil microbial communities (Kissling et. al., 2009). Pore space is essential in healthy soils where pore space can hold water and, more importantly, oxygen needed for root growth (Liddle 1997). To add, soil physical strength reduces root elongation (Benigno et. al., 2012). Thus, even with the removal of trampling pressure, the effects of compacted soil could persist indefinitely, and continue to negatively impact seedling establishment and halt root growth which would further the persistence of exposed, compacted soil.

Exposure alters moisture regimes, daily temperatures, and nutrient content

Compaction and trampling exposure can also lead to altered moisture regimes. Soil moisture content decreases the resistance of vegetation at both extremes, where plants are crushed into wet soils at high moisture contents and are easily broken when moisture content is low (Price, 1985). Water run-off can also be increased when compaction is increased (Korkanç, 2014). Topsoil also contains many essential nutrients for plant growth and any loss of topsoil through erosion processes such as water run-off can greatly deteriorate growth conditions (Holmes, 2001).

To add, daily temperatures were higher in on-trail conditions (Figure 2.9). Exposed, compacted soil lacks the insulating effect of vegetation and tends to experience a greater range of temperature fluctuations wherein these areas tend to be hotter at midday and colder at midnight than unaltered reference vegetation (Liddle, 1997). This study did not consider nighttime temperatures. Most species tend to germinate optimally at 20 °C (Rydgren et. al., 2017). High daily temperatures via reduced shading and increased exposure on trails can contribute to seed bank loss, poor germination rates and reduce the potential for natural recovery.

Additionally, there was very little leaf litter found on trails (Figure 2.6) where exposure to wind and water run-off likely forces leaf litter to blow elsewhere and likely get trapped off-trail where vegetation cover is higher. Litter addition enhances soil C mineralization and N mineralization at lower temperatures in alpine environments (He et. al., 2014). The loss of litter on trails can reduce these processes making it more difficult to replace nutrient loss through erosion and leaching. He et. al. (2014) found that N mineralization was reduced and rather was immobilized at higher temperatures. As daily temperatures are significantly higher on trails, there is potential for N immobilization to contribute to nutrient loss on trails. To add, leaf litter can often be a source of seed-trapping material which can promote regrowth of vegetation and any loss of soil can remove this source of recovery and potentially promote the growth of non-native species (Putwain & Gilham, 1990). This is a potential reason for the lack of any established seed bank on the trail at Skyline which had very poor leaf litter cover.

Loss of topsoil alters nutrient content and acidity

Soil depth is often negatively associated with trail degradation (Leung & Marion, 1996), substrate depths were shallower on trails at both Skyline and Mica Hill, although only by 10.65 cm and 4.77 cm, respectively (Figure 2.9). This may be because soils are generally shallow and rocky in barrens habitat. Therefore, any loss of soil on trails in terms of depth, may not be significantly different from the already shallow depth of undamaged vegetation. However, since topsoil is the most important source of nutrients for plant growth, any of its loss has the potential to negatively impact plant growth and overall cover.

The lower content of all major nutrients for plant growth on trails is likely the consequence of the loss of the organic topsoil layer via erosion. What remains is a nutrient-poor, rocky mineral soil (personal observation). Terrestrial plant communities are often limited by N or P, or both (Wetterstedt & Billberger, 2012). Bowman et. al. (1993) demonstrated that wet alpine meadow communities are limited by N while dry alpine meadows are co-limited by N and P. It is likely that Mica Hill is limited by N and

Skyline is co-limited by both N and P and the general reduction of all nutrients will to contribute to reduced vegetation cover and its ability for natural recovery.

Although topsoil has better water-holding capacity than lower layers of soil (Holmes, 2001), when trampling intensity is increased, the ability of soil to hold water is similarly reduced across all depths (Korkanç, 2014). However, water uptake by roots becomes more difficult as compaction increases (Tracy et. al., 2011). The lack of vegetation and root networks on the trails, combined with very compact soils, can reduce the ability for water to be removed from the trail (Lemauviel & Rozé, 2003) despite elevated temperatures which can increase evaporation. The result is higher moisture content on-trails when compared to off-trails where dense vegetation can quickly absorb most of the available water within 3-5 days of a rainfall event.

In addition, the loss of topsoil is akin to loss of the acidic humic layer of soil. Many ericaceous species are not well-suited to calcareous soils (Clarke, 1997), and so increased soil pH has the potential to promote the growth of non-native or non-target species. The change in pH may also affect soil microbial communities which can cause changes to the above-ground vegetation (De Vries et. al., 2012) and particularly bacterial composition which is strongly affected by pH gradients (Rousk et. al., 2010). Therefore, the increased soil pH on trails can greatly affect natural recovery of native vegetation through a variety of mechanisms that were not considered in this study. This effect is potentially more notable at the Mica Hill hiking trail, where a more distinct community has formed (Table 2.1).

Seed bank recovery potential is low

Some restoration projects have shown that vegetation is able to recover via seed bank germination, provided damage is not severe (Prach & Hobbs, 2008) or if there is an adjacent, intact seed bank (Sawtschuk et. al., 2010). Conversely, if the damage is too severe and the seed bank is compromised, there is reduced potential of spontaneous natural recovery. While seed bank emergence studies typically take place in controlled greenhouse environments, it was possible to conclude from the outdoor emergence study that there is some form of seed bank present at both Skyline and Mica Hill hiking trails in unaffected, off-trail vegetation (Figure 2.15).

When compared to the negative control, all seed sources, except for on-trail at Skyline, produced higher rates of germination (Figure 2.15). A possible cause for lack of a seed bank includes substrate compaction, which can limit seed penetration and persistence in the soil. Additionally, reduced vegetation cover can remove any potential for seeds to become trapped and remain in branches and dense leafy vegetation, and high wind exposure can cause seeds to be blown off-site. While ericaceous species often found on barrens tend to have long-lived persistent seed banks (Canals & Sebastià, 2002), low germination rates from this study do not support this statement. Albeit, many factors may have contributed to poor germination rates in this study.

While germinations were higher on the trail at Mica Hill (Figure 2.14), they mainly consisted of two species which were not present in undisturbed areas: *H. canadense* and an unidentifiable *Juncus spp.* (Figure 2.12) that is likely *Juncus brevicaudatus*, which was identified as predominant on trails at Mica Hill (Table 2.1). It is possible that outdoor growing conditions mimicked that of overheated and exposed trails, which promoted the growth of these species. To add, species of the genus *Juncus* and *Hypericum* generally contribute large quantities of seed every season that can contribute to persistent seed banks (Quintana-Ascencio & Morales-Hernández, 1997; Kirschner et, al., 2002; Måren & Vandvik, 2009). Therefore, growing conditions and seed volume likely contributed to the germination success of these species.

Setbacks regarding seed bank emergence study

Overall, germination rates were low for every treatment. This is likely due to the study design wherein environmental factors such as temperature were not controlled. Most germination studies perform germination trials within a controlled greenhouse setting (Rygren et. al., 2017; Vanstockem et. al., 2018). Most species germinate optimally at 20 °C (Rydgren et. al., 2017) and it is likely that temperatures often exceeded this range in this outdoor study which may have contributed to low overall germination rates. However, Galinato & Van Der Valk (1986) found that germination percentages were highest under alternating temperature regimes. Insufficient knowledge of germination requirements, particularly in heathland and grassland communities can misrepresent seed bank profile and important species in a population can potentially be omitted (Bossyut & Hermy, 2003). Therefore, germination studies controlled at a single temperature may produce a seed bank profile that is not representative of the actual profile.

Most species germinated were not those found in undamaged vegetation at either Mica Hill or Skyline hiking trails. As these species were present in all seed sources as well as both positive and negative controls, it is likely that outside factors were at play.

The two main causes of contamination were likely the commercial topsoil used to increase soil quantities and seed rain from weedy lawn vegetation near the study location. A layer of sand was meant to act as a barrier between soil types but was not thick enough to inhibit all germination. Some species, often considered as weeds, germinated in all tray types where some were seen present as mature individuals bearing seeds in nearby lawn vegetation, and some were not. For the latter half of the study, plastic domes, which were meant to prevent seed rain, had to be removed as rising daily temperatures and sun exposure would have been detrimental to seedling survival.

Another contributing factor to low germination rates may have been the length of the study. In general, emergence studies continue until a period of time passes, varying from 1 week to several weeks, where no new germination is observed (Ter Heerd et. al., 1996; Ma et. al., 2013; Fennell et. al., 2014). This study ran run for 14 weeks, before being halted due to time constraints. Germination continued until the final week of monitoring and possibly would have persisted into the late Fall. However, there is potential that the prolonged germination period is due to the seed rain of nearby weed species which have the potential to germinate without any stratification processes.

Consideration for restoration practices at these locations

Overall, high trail usage and disturbance at both closed sections of trail at Skyline and Mica Hill have led to degraded ecosystems that are not likely to recover naturally within a reasonable timeframe, if at all. A high degree of compaction and soil loss on both trails is likely the main cause for reduced vegetation cover, increased substrate moisture and nutrient loss. Trail exposure has also led to higher surface temperatures which can reduce germination rates, as well as increased soil erosion. All factors affect each other in positive feedback loops and thus, active intervention is required to break the cycle. Soil quality is poor and itself is lacking and so restoration measures that bring new material into the system are likely to reintroduce essential plant nutrients and decrease compaction which should allow for increased plant growth. Other measures that should be explored include directly reintroducing plant material into the system through mature plants which can increase cover, reduce temperature fluctuations, and whose root networks can stabilize and protect soil from erosion. Additionally, desirable seed can be added to restore a version of a seed bank of the species targeted for restoration. In general, it will be important to consider restoration strategies that target both soil/substrate factors and those that target the vegetation itself.

2.5 Literature cited

Abella, S. R., Chiquoine, L. P., & Vanier, C. H. (2013). Characterizing soil seed banks and relationships to plant communities. Ecology, 214(5), 703–715. https://doi.org/10.1007/s

Baechler, F., & Boehner, R. (2014). Karst geology and hydrogeology of Cape Breton Island, Nova Scotia: an overview. Canadian Journal of Earth Sciences, 51, 701–714. https://doi.org/10.1139/cjes-2013-0157

Ballantyne, M., & Pickering, C. M. (2015). The impacts of trail infrastructure on vegetation and soils: Current literature and future directions. Journal of Environmental Management, 164, 53–64. https://doi.org/10.1016/j.jenvman.2015.08.032

Bardon, C., Misery, B., Piola, F., Poly, F., & Le Roux, X. (2018). Control of soil N cycle processes by Pteridium aquilinum and Erica cinerea in heathlands along a pH gradient. Ecosphere, 9(9). https://doi.org/10.1002/ecs2.2426

Bassett, I. E., Simcock, R. C., & Mitchell, N. D. (2005). Consequences of soil compaction for seedling establishment: Implications for natural regeneration and restoration. Austral Ecology, 30(8), 827–833. https://doi.org/10.1111/j.1442-9993.2005.01525.x

Beckett, C. T. S., Glenn, D., Bradley, K., Guzzomi, A. L., Merritt, D., & Fourie, A. B. (2017). Compaction conditions greatly affect growth during early plant establishment. Ecological Engineering, 106, 471–481. https://doi.org/10.1016/j.ecoleng.2017.04.053

Bekker, R. M., Bakker, J. P., Grandin, U., Kalamees, R., Milberg, P., Poschlod, P., … Willems, J. H. (1998). Seed Size, Shape and Vertical Distribution in the Soil: Indicators of Seed Longevity. Source: Functional Ecology (Vol. 12). Retrieved from https://about.jstor.org/terms

Bell, K. L., Forest, W., & Bliss, L. C. (1973). Alpine Disturbance Studies: Olympic National Park, USA. Biological Conservation, 5(1), 25–32.

Benigno, S. M., Cawthray, G. R., Dixon, K. W., & Stevens, J. C. (2012). Soil physical strength rather than excess ethylene reduces root elongation of Eucalyptus seedlings in mechanically impeded sandy soils. Plant Growth Regulation, 68, 261–270. https://doi.org/10.1007/s10725-012-9714-2

Bernhardt-Römermann, M., Gray, A., Vanbergen, A. J., Bergès, L., Bohner, A., Brooker, R. W., … Stadler, J. (2011). Functional traits and local environment predict vegetation responses to disturbance: A pan-European multi-site experiment. Journal of Ecology, 99(3), 777–787. https://doi.org/10.1111/j.1365-2745.2011.01794.x

Bossuyt, B., & Hermy, M. (2003). The potential of soil seedbanks in the ecological restoration of grassland and heathland communities. Belgian Journal of Botany, 136(1), 23–34.

Bowman, W. D., Theodose, T. A., & Schardt, J. C. (1993). Constraints of Nutrient Availability on Primary Production in Two Alpine Tundra. Ecology (Vol. 74).

Bridgland, J., Aucoin, S., & Harvey, G. (2011). Area Burned by Past Wildfires. Cape Breton Highlands National Park, 1–30.

Bried, J. T., Patterson, W. A., & Gifford, N. A. (2014). Over Pine. Restoration Ecology, 22(4), 442–446. https://doi.org/10.1111/rec.12097

Bryan, R. B. (1977). The Influence of Soil Properties on Degradation of Mountain Hiking Trails at Grövelsjön. Geografiska Annaler. Series A, Physical Geography, 592(1), 49–65.

Cameron, Robert P., & Bondrup-Nielson, S. (2013). Plant Communities within Atlantic Coastal Heathlands in Nova Scotia. *Northeastern Naturalist*, *20*(4), 694–709.

Canals, R.-M., & Sebastià, M.-T. (2002). Barren dynamics in biotically disturbed areas: on the role of some features enhancing heath success. Acta Oecologia, 23, 303–312.

Clarke, C. T. (1997). Role of soils in determining sites for lowland heathland reconstruction in England. Restoration Ecology, 5(3), 256–264. https://doi.org/10.1046/j.1526-100X.1997.09730.x

Cole, D. N., & Bayfield, N. G. (1993). Recreational trampling of vegetation: standard experimental procedures. Biological Conservation, 63, 209–215.

Cole, D. N. (2004). Impacts of hiking and camping on soils and vegetation: a review. In R. Buckley. Environmental Impacts of Ecotourism. (Series no. 2). Massachusetts, USA: CABI Publishing.

Connell, J. H. (1978). Diversity in tropical rain forests and coral reefs. Science, 199(4335), 1302–1310.

Crisfield, V. E., Macdonald, S. E., & Gould, A. J. (2012). Effects of Recreational Traffic on Alpine Plant Communities in the Northern Canadian Rockies. Arctic, Antarctic, and Alpine Research Arctic, Antarctic, and Alpine Research, 44(3), 277–287. https://doi.org/10.1657/1938-4246-44.3.277

Culver, D. C., & Beattie, A. J. (1983). Effects of Ant Mounds on Soil Chemistry and Vegetation Patterns in a Colorado Montane. Ecology, 64(3), 485–492.

De Graaf, M. C. C., Bobbink, R., Smits, N. A. C., Van Diggelen, R., & Roelofs, J. G. M. (2009). Biodiversity, vegetation gradients and key biogeochemical processes in the barren landscape. Biological Conservation, 142, 2191–2201. https://doi.org/10.1016/j.biocon.2009.04.020

De la Peña, E., Van De Velde, H., Lens, L., & Bonte, D. (2012). Soil Conditions in Natural, Declining and Restored Barrens Influence Plant-Pollinator Interactions of Calluna vulgaris. Restoration Ecology, 20(5), 603–611. https://doi.org/10.1111/j.1526-100X.2011.00844.x

De Vries, F. T., Manning, P., Tallowin, J. R. B., Mortimer, S. R., Pilgrim, E. S., Harrison, K. A., … Bardgett, R. D. (2012). Abiotic drivers and plant traits explain landscape-scale patterns in soil microbial communities. Ecology Letters, 15, 1230–1239. https://doi.org/10.1111/j.1461-0248.2012.01844.x

Dickens, S. J., Gerhardt, F. , & Collinge, S. K. (2005). Recreational portage trails as corridors facilitating non-native plant invasions of the boundary waters canoe area wilderness (U.S.A.). Conservation Biology, 19(5), 1653–1657.

Ellis, K. (2017). Evaluation of the Erosion Reduction Potential, Stability, and Resilience of Living Shorelines in Nova Scotia. (Master's thesis). Saint Mary's University, Halifax, Nova Scotia.

Fennell, M., Gallagher, T., Vintro, L. L., & Osborne, B. (2014). Using soil seed banks to assess temporal patterns of genetic variation in invasive plant populations. Ecology and Evolution, 4(9), 1648–1658. https://doi.org/10.1002/ece3.1043

Galinato, M., & Van Der Valk, A. G. (1986). Seed germination traits of annuals and emergents recruited during drawdowns in the Delta Marsh, Manitoba, Canada. Aquatic Botany, 26, 89–102. https://doi.org/10.1016/0304-3770(86)90007-0

Gallet, S., & Roze, F. (2001). Resistance of Atlantic Barrens to trampling in Brittany (France): influence of vegetation type, season and weather conditions. Biological Conservation, 97, 189–198.

Gibson, M. D., Heal, M. R., Li, Z., Kuchta, J., King, G. H., Hayes, A., & Lambert, S. (2013). The spatial and seasonal variation of nitrogen dioxide and sulfur dioxide in Cape Breton Highlands National Park, Canada, and the association with lichen abundance. Atmospheric Environment, 64, 303–311. https://doi.org/10.1016/j.atmosenv.2012.09.068

Gonzalez, S. L. & Ghermandi, L. (2012). Comparison of methods to estimate soil seed banks: the role of seed size and mass. Community Ecology, 13(2), 238–242. https://doi.org/10.1556/ComEc.13.2012.2.14

Halsey, L. G., Curran-everett, D., Vowler, S. L., & Drummond, G. B. (2015). The fickle P value generates irreproducible results. *Nature Methods*, *12*(3), 179–185.

Havlick, D. G., Billmeyer, E., Huber, T., Vogt, B., & Rodman, K. (2016). Journal of Sustainable Tourism Informal trail creation: hiking, trail running, and mountain bicycling in shortgrass prairie. Journal of Sustainable Tourism, 24(7), 1041–1058. https://doi.org/10.1080/09669582.2015.1101127

He, Y., Xu, X., Kueffer, C., Zhang, X., & Shi, P. (2014). Leaf litter of a dominant cushion plant shifts nitrogen mineralization to immobilization at high but not low temperature in an alpine meadow. Plant Soil, 383, 415–426. https://doi.org/10.1007/s11104-014-2216-4

Heerdt, G. N. J. Ter, Verweij, G. L., Bekker, R. M., & Bakker, J. P. (1996). An Improved Method for Seed-Bank Analysis: Seedling Emergence After Removing the Soil by Sieving. Functional Ecology, 10(1), 144–151.

Holmes, P. M. (2001). Shrubland restoration following woody alien invasion and mining: Effects of topsoil depth, seed source, and fertilizer addition. Restoration Ecology, 9(1), 71–84. https://doi.org/10.1046/j.1526-100X.2001.009001071.x

Hudek, C., Sturrock, C. J., Atkinson, B. S., Stanchi, S., & Freppaz, M. (2017). Root morphology and biomechanical characteristics of high altitude alpine plant species and their potential application in soil stabilization. Ecological Engineering, 109, 228–239. https://doi.org/10.1016/j.ecoleng.2017.05.048

ISO (2017). Geotechinical investigation and testing- identification and classification of soil. International Organization for Standardization.

Jägerbrand, A. K., & Alatalo, J. M. (2015). Effects of human trampling on abundance and diversity of vascular plants, bryophytes and lichens in alpine heath vegetation, Northern Sweden. Springer Plus, 4(95), 1–12. https://doi.org/10.1186/s40064-015-0876-z

Kirschner J, Balslev H, Clemants SE, Ertter B, Alvarez MCFC, Ha¨met-Ahti L, Miyamoto F, Noltie HJ, Novara LJ, Novikov VS, Simonov SS, Snogerup S, Wilson KL (2002) Juncaceae 2: Juncus subg. Juncus, Species Plantarum: flora of the World Part 7. Australian Biological Resource Study, Canberra

Kissling, M., Hegetschweiler, K. T., Rusterholz, H.-P., & Baur, B. (2009). Short-term and long-term effects of human trampling on above-ground vegetation, soil density, soil organic matter and soil microbial processes in suburban beech forests. Applied Soil Ecology, 42, 303–314. https://doi.org/10.1016/j.apsoil.2009.05.008

Korkanç, S. Y. (2014). Impacts of recreational human trampling on selected soil and vegetation properties of Aladag Natural Park, Turkey. Catena, 113, 219–225. https://doi.org/10.1016/j.catena.2013.08.001

Krusche, J. K. (2015). *Doing Bayesian data analysis*. Oxford, UK: Elsevier Inc.

Lemauviel, S., & Oise Rozé, F. (2003). Response of Three Plant Communities to Trampling in a Sand Dune System in Brittany (France). *Environmental Management*, *31*(2), 227–235. https://doi.org/10.1007/s00267-002-2813-5

Lemky, K. (2017). Journal of Heritage Tourism The revitalization of a heritage travel route: Canada’s Cabot Trail. Journal of Heritage Tourism, 12(5), 526–535. https://doi.org/10.1080/1743873X.2016.1242592

Leung, Y.-F. & Marion, J. L. (1996). Trail degradation as influenced by environmental factors: a state-of-the-knowledge review. Journal of Soil and Water Conservation, 51(2), 130–136.

Liddle, M. J. (1975). A theoretical relationship between the primary productivity of vegetation and its ability to tolerate trampling. Biological Conservation, 8(4), 251–255.

Liddle, M. J. (1997). Recreation Ecology. London; New York: Chapman & Hall.

Ma, M., Zhou, X., & Du, G. (2013). Effects of disturbance intensity on seasonal dynamics of alpine meadow soil seed banks on the Tibetan Plateau. Plant Soil, 369, 283–295. https://doi.org/10.1007/s11104-012-1560-5

Marion, J. L. & Leung, Y.-F. (2004). Environmentally sustainable trail management. In R. Buckley. Environmental Impacts of Ecotourism. (Series no. 2). Massachusetts, USA: CABI Publishing

Måren, M., Inger E & Vandvik, Vigdis. (2009). Fire and regeneration: the role of seed banks in the dynamics of northern heathlands. Journal of Vegetation Science, 20, 871–888.

Mason, S., Newsome, D., Moore, S., & Admiraal, R. (2015). Recreational trampling negatively impacts vegetation structure of an Australian biodiversity hotspot. Biodiversity Conservation, 24, 2685–2707. https://doi.org/10.1007/s10531-015-0957-x

McElreath, R. (2015). *Statistical rethinking: a Bayesian course with examples in R and Stan*. Boca Raton, FL: Taylor & Francis Group.

McIldoon, T., & Pilon, M. (2008). Examples of mountain induced winds in Atlantic Canada. Conference on Mountain Meteorology, 5.

Miller, B. P., Sinclair, E. A., Menz, M. H. M., Elliott, C. P., Bunn, E., Commander, L. E., … Stevens, J. C. (2017). A framework for the practical science necessary to restore sustainable, resilient, and biodiverse ecosystems. Restoration Ecology, 25(4), 605–617. https://doi.org/10.1111/rec.12475

Ólafsdóttir, R., & Runnström, M. C. (2013). Assessing hiking trails condition in two popular tourist destinations in the Icelandic highlands. Journal of Outdoor Recreation and Tourism, 3–4, 57–67. https://doi.org/10.1016/j.jort.2013.09.004

Parikesit, P., Larson, D. W., & Matthes-Sears, U. (1995). Impacts of trails on cliff-edge forest structure. Canadian Journal of Botany, 73, 943–953.

Parks Canada (2010). Cape Breton Highlands: National park of Canada management plan. Parks Canada.

Parks Canada (2018) Parks Canada Attendance 2017-18. Parks Canada

Pickering, C. M., & Norman, P. (2017). Comparing impacts between formal and informal recreational trails. Journal of Environmental Management, 193, 270–279. https://doi.org/10.1016/j.jenvman.2016.12.021

Piessens, K., Honnay, O., & Hermy, M. (2005). The role of fragment area and isolation in the conservation of heathland species. Biological Conservation, 122, 61–69. https://doi.org/10.1016/j.biocon.2004.05.023

Prach, K., & Hobbs, R. J. (2008). Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restoration Ecology. https://doi.org/10.1111/j.1526-100X.2008.00412.x

Price, M. F. (1985). Impacts of Recreational Activities on Alpine Vegetation in Western North America. Mountain Research and Development, 5(3), 263–278. Retrieved from http://www.jstor.org/stable/3673358

Putwain, P. D., & Gillham, D. A. (1990). The Significance of the Dormant Viable Seed Bank in the Restoration of Heathlands. Biological Conservation, 52, 1-16

Pywell, R. F., Putwain, P. D., & Webb, N. R. (1997). The Decline of Barren Seed Populations Following the Conversion to Agriculture. Journal of Applied Ecology (Vol. 34).

Quintana-Ascencio, P., & Morales-Hernández, M. (1997). Fire-mediated effects of shrubs, lichens and herbs on the demography of Hypericum cumulicola in patchy Florida scrub. *Oecologia*, *112*, 263–271.

Rivera, D. E., Mejías, V., Jáuregui, B. M., Costa-Tenorio, M., & Ló Pez-Archilla, A. I. (2014). Spreading Topsoil Encourages Ecological Restoration on Embankments: Soil Fertility, Microbial Activity and Vegetation Cover. *PLoS ONE*, *9*(7), 101413. https://doi.org/10.1371/journal.pone.0101413

Rodríguez-Echeverría, S., Armas, C., Pistón, N., Hortal, S., & Pugnaire, F. I. (2013). A role for below-ground biota in plant – plant facilitation. Journal of Ecology, 101, 1420–1428. https://doi.org/10.1111/1365-2745.12159

Roem, W. J., Klees, H., & Berendse, F. (2002). Effects of nutrient addition and acidification on plant species diversity and seed germination in barren. Journal of Applied Ecology, 39(6), 937–948. https://doi.org/10.1046/j.1365-2664.2002.00768.x

Rousk, J., Bååth, E., Brookes, P. C., Lauber, C. L., Lozupone, C., Caporaso, J. G., … Fierer, N. (2010). Soil bacterial and fungal communities across a pH gradient in an arable soil. The ISME Journal, 4, 1340–1351. https://doi.org/10.1038/ismej.2010.58

Rydgren, K., Auestad, I., Hamre, L. N., Hagen, D., Rosef, L., & Skjerdal, G. (2016). Long-term persistence of seeded grass species: an unwanted side effect of ecological restoration. Environmental Science and Pollution Research, 23(14), 13591–13597. https://doi.org/10.1007/s11356-015-4161-z

Sawtschuk, J., Bioret, F., & Gallet, S. (2010). Spontaneous Succession as a Restoration Tool for Maritime Cliff-top Vegetation in Brittany, France. Restoration Ecology, 18(SUPPL. 2), 273–283. https://doi.org/10.1111/j.1526-100X.2010.00653.x

Scherrer, P., & Pickering, C. M. (2006). Recovery of Alpine Herbfield on a Closed Walking Track in the Kosciuszko Alpine Zone, Australia. Arctic, Antarctic, and Alpine Research, 38(2), 239–248.

Shang, Z.-H., Deng, B., Ding, L.-M., Ren, G.-H., Xin, G.-S., Liu, Z.-Y., … Wang, Y.-L. (2013). The effects of three years of fencing enclosure on soil seed banks and the relationship with above-ground vegetation of degraded alpine grasslands of the Tibetan plateau. Plant Soil, 364, 229–244. https://doi.org/10.1007/s11104-012-1362-9

Thompson, K., & Grime, J. P. (1979). Seasonal Variation in the Seed Banks of Herbaceous Species in Ten Contrasting Habitats. Journal of Ecology, 67(3), 893–921.

Timothy, D. J. (2015). Tourism and Trails: Cultural, Ecological and Management Issues. Bristol, UK: Channel View Publications.

Tracy, S. R., Black, C. R., Roberts, J. A., & Mooney, S. J. (2011). Soil compaction: a review of past and present techniques for investigating effects on root growth. Journal of the Science of Food and Agriculture, 91, 1528–1537. https://doi.org/10.1002/jsfa.4424

Van Den Berg, L. J. L., Dorland, E., Vergeer, P., Hart, M. A. C., Bobbink, R., & Roelofs, J. G. M. (2005). Decline of Acid-Sensitive Plant Species in Barren Can Be Attributed to Ammonium Toxicity in Combination with Low pH. The New Phytologist, 166(2), 551–564. https://doi.org/10.1111/j.1469-8137.2005.01338.x

Vanstockem, J., Ceusters, C., Van Dyck, K., Somers, B., & Hermy, M. (2018). Is there more than meets the eye? Seed bank analysis of a typical novel ecosystem, the extensive green roof. Applied Vegetation Science, 21(3), 419–430. https://doi.org/10.1111/avsc.12383

Warr, S. J., Thompson, K., & Kent, M. (1993). Seed banks as a neglected area of biogeographic research: a review of literature and sampling techniques. Progress in Physical Geography, 17(3), 329–347.

Wetterstedt, M., Billberger, M. F. K., & A, I. (2012). Nutrient limitation on terrestrial plant growth – modeling the interaction between nitrogen and phosphorus. *New Phytologist*, *194*, 953–960.

Whinam, J., & Chilcott, N. M. (2003). Impacts after four years of experimental trampling on alpine/sub-alpine environments in western Tasmania. *Environmental Management, 67,* 339–351. https://doi.org/10.1016/S0301-4797(02)00218-9

Wolf, I. D., & Croft, D. B. (2014). Impacts of tourism hotspots on vegetation communities show a higher potential for self-propagation along roads than hiking trails. Journal of Environmental Management, 143, 173–185. https://doi.org/10.1016/j.jenvman.2014.04.006

Zak, D. R., Holmes, W. E., White, D. C., Peacock, A. D., & Tilman, D. (2003). Plant diversity, soil microbial communities, and ecosystem function: are there any links? *Ecology*, *84*(8), 2042–2050.

Barren and alpine restoration

3.1 Introduction

Restoration practices along trails and in parks have often been employed when vegetation damage reaches a point where visitor experience becomes impacted. Visitor experience can be strongly affected by their perceived degradation of the surroundings (Timothy, 2015). A study by Lynn and Brown (2003) found that vegetation damage and fire rings contributed to the most negative visitor experiences, followed by trail extension, widening and erosion. Improving trail conditions and surrounding vegetation especially in areas where natural, passive regeneration will take a long time, can boost the quality of experience for users of the trails.

Various treatments have been tested in many barren and heathland regions to improve biodiversity and ecosystem functioning of degraded trails and regions. Passive restoration efforts of simply closing off degraded sections of trails and allowing the area to regenerate have been successful, but only when the seedbank is still intact (Sawtshuk et. al., 2010). Other restoration projects have shown that after 15 years of removal of walking pressures, revegetation still might not occur (Scherrer & Pickering, 2006). Active intervention in these areas becomes desirable. Such treatments include transplants (Ebersole et. al., 2004), turf cuttings (Pywell et. al., 1995; Vergeer et. al., 2006), seeding (Scherrer & Pickering, 2006; Burke, 2008, Rydren et. al., 2017), soil amendments (Burke, 2008) and the use of erosion control mats (Ebersole et. al., 2004).

3.1.1 Transplanting and turf mats

Transplanting and the use of turf mats in damaged areas can be an effective restoration technique in certain situations. Transplanting can involve the propagation of harvested seed or the direct transplanting of mature plants from a nearby location. In this manner, live mature plants are immediately reintroduced to degraded areas and immediately provide a form of vegetation cover. However, transplants need to be hardy enough to endure more stressful conditions in these areas which tend to have larger surface temperature fluctuations and more instances of drought (Liddle, 1997).

A common method of transplanting involves cutting turfs or sods of vegetation from a donor site which typically hosts the same community type as the recipient site. With this method, soil is also added to the restoration site which is likely host to beneficial mycorrhizal and microbial communities which are responsible for a biogeochemical transformations and soil structure (Eviner & Hawkes, 2008) and also can contain a portion of the native seed bank. Turf transplants can be more successful than transplanting single plants (Pywell et. al., 1995) where plants are already conditioned to the area and are often more mature than plants grown *ex situ*. Turf size and plant type play an important factor in success where grassland vegetation can tolerate smaller turf divisions better than shrubland vegetation (Aradottir, 2012).

While turf cutting is very effective at restoring damaged sites, it does have significant implications for the donor sites. It was found that arbuscular mycorrhizal spores, which promote growth and germination of some barren species, decreased in concentration in areas where turfs were taken (Vergeer et. al., 2006). To add, cutting turves from a wet barren in areas where hydrology has been altered can negatively impact soil seed banks (Berg et. al., 2003; Jansen et. al., 2004) and the depth of the turf cut plays a role as well (Berg et. al., 2003). Thus, while turf cuttings may be effective, it can create further issues in the regions where they were taken. They are also often time intensive and can be costly (Pywell et. al., 1995).

3.1.2 Seeding

Seeding an area is a common restoration strategy for barren and alpine communities. Seeds will often go through a period of dormancy until the right abiotic conditions arise that trigger germination. Using seed can be more cost and time effective than transplanting, but successful germination and survival in the damaged area may be more challenging as the seedling stage is often the most stressful part of the plant's life cycle and can be particularly affected by soil compaction (Bassett et. al., 2005).

Seed size can also affect germination and growth. Smaller seeds tend to spend more time in the soil before germination and have higher mortality rates than larger seeds due to higher rates of fungal and bacterial infection (Moles & Westoby, 2014). However, they take less time to reach reproductive maturity (Moles & Westoby, 2014). To add, germination rates can be affected by light. For example, *Deschampsia flexuosa* had high germination rates in its natural habitat and germinate well with the presence of light or in the darkness (Pons, 1989).

Plant type is another factor where seeding is generally more effective for graminoids and forbs, whereas shrubs and trees, which grow more slowly, may require transplanting (Cole, 2007). Also, it is important to consider which species should be a part of a seed mixture. A study in grasslands, found that high-diversity seed mixtures resulted in significantly more biomass and ground cover than a low-diversity seed mixture (Kirmer et. al., 2012). However, a study in Norway found that a species of grass established significantly more cover after 3 years when grown in a monoculture than when it was grown in a mixture with two other grass species (Rydgren et. al., 2017). Therefore, the species grown from seed may experience competition rather than co-existence or facilitation and results are likely dependent on the specific combinations of mixtures that are used.

3.1.3 Soil amendments

In barrens, the soil depth is often shallow and so is quickly eroded when disturbed. Adding fresh topsoil to a disturbed site has been shown to be effective where seedling survival and growth is significantly improved compared to sites without topsoil (Holmes, 2001). Adding an organic component to degraded areas, such as compost, has also been shown to improve biomass production in serpentine communities (Meyer-Grünfeldt et. al., 2015). To add, seeding and transplanting treatments are significantly more effective at increasing plant density when grown on organic soil than when grown on mineral soils (Cole, 2007; Rydgren et. al., 2017). Soil amendments can greatly increase the chances of a successful restoration project, and to be able do so within a more reasonable timeframe (Ohsowski et. al., 2012).

As barrens soils are generally low in nutrients, fertilization of the degraded areas can improve soil quality. Nutrient addition can also affect plants differently based on their life-history stage such that the seedling phase is often the most susceptible to damaging events and fertilization has the most potential to improve seedling success (Meyer-Grünfeldt et. al., 2015). Species are also limited by different nutrients (Roem et. al., 2002) and so different fertilizers can promote the growth of different species.

However, improving soil quality can encourage the growth of other species which may not be dominant in the given area (Helsper et. al., 1983; Holmes, 2001). For example, altering the ratio of phosphate, nitrogen, and calcium in the soil can promote the growth of species other than *Calluna vulgaris* in a *Calluna*-type barren (Helsper et. al., 1983). However, the effects of fertilization can wear off within the first few years as nutrients are taken up by vegetation and leach into the soil (Helsper et. al., 1983). To add, the effects of fertilizer tend to increase the density of non-native plant species the shallower the soil (Holmes, 2001). While there is the benefit of adding fertilizer to damaged areas, it is important to consider the associated risks of optimizing the area for non-target or non-native species that can inhibit the growth of native species (Hagen et. al., 2014).

Natural atmospheric deposition of nitrogen compounds can decrease plant species diversity in barrens environments (Roem et. al., 2002) as well as cause community shifts from ericoid shrubs characteristic of barrens to perennial grasses (Fagúndez, 2012). Nonetheless, one study in the UK has shown that nitrogen addition, simulating atmospheric deposition, can lead to an increase in plant density and size of the soil microbial community in heathlands up to eight years after fertilization (Power et. al., 2006). The effects of nutrient addition in barrens can be varied and are likely dependent on specific barren community types and influencing environmental factors.

Once the topsoil is added, the newly added soil may continue to erode if the initial erosion pressures are not removed. In some cases, the removal of walking pressures may be enough to slow erosion, but degraded trails have the potential to divert water and increase sediment loss through runoff or displacement during heavy rain or wind events, respectively (Liddle, 1997). Soil erosion controls such as mats have been proven to be effective when used in conjunction with soil seeding and transplanting treatments. Ebersole et. al. (2004) found that seedling germination and survival were significantly higher when grown under an erosion control mat. The mats generally consist of biodegradable fibers, such as aspen shavings or coconut husk, which are interwoven into a blanket that can be rolled over the affected area and staked into the ground. Many mats have different slope efficiency ratings and vary in lifespan from 6 months to 24 months. Other erosion controls via brush mats and hay mulch have also been shown to reduce sediment loss (Ellis, 2017).

3.1.4 Further restoration considerations

When choosing which species to transplant as well as designing seed mixtures to use in a restoration project, using native species is becoming increasingly important (Matesanz & Valladares, 2007; Rydgren et. al., 2017). Hagen et. al. (2014) found that, while the use of non-native species can increase the total vegetation cover, after more than 20 years, there was no natural succession from non-native species to native species. And when native species were used in mixtures with non-native commercial species, overall revegetation was not increased as the native species were outcompeted rather than facilitated by the quick-establishing non-native species (Matesanz & Valladares, 2007).

While the use of native stock seed populations through artificial selection processes can have increased germination rates and overall vigor, they can be poor competitors (Herget et. al., 2015). In addition, commercially native and non-native grown species are found to have more homozygous genes and thus suffer from inbreeding depression as a result of stock populations being propagated repeatedly over many generations without introducing new genetic material (Aavik et. al., 2012). Therefore, wild-collected native species should be used when possible to re-establish vegetation in degraded barren sites to avoid the introduction of new species that could outcompete and potentially disrupt the ecosystem.

While there has been some debate about the need to use locally sourced plants and seeds in restoration projects (Wilkinson, 2001; Jones, 2013), overall, the consensus is that local is better (Hamilton, 2001; Vander Mijnsbrugge et. al., 2010; Bucharova et. al., 2017, Miller et. al., 2017). Locally adapted and non-local plants of the same species have some genetic differences such that plants will have improved fitness when planted in the same area from which they were sourced when compared to the same species sourced from a different region (Bucharova et. al., 2017). Their phenology is reflective of their source area and thus are more likely to be successfully pollinated. Furthermore, the use of non-local origins for plants and seeds run the risk of introduction new genetic material into the area which could potentially disturb the genetic structure of the ecosystem (Hamilton, 2001; Vander Mijnsbrugge et. al., 2010; Rydgren, 2017). Other genetic issues involved with non-local plant sources include maladaptation to the local environment, subsequent outbreeding depression as local and maladapted non-local hybridization occurs, or non-local plants outperform local plants and have the potential to become 'invasive' (Vander Mijnsbrugge et. al., 2010).

In general, the addition of native topsoil is very effective at promoting the regeneration of native species (Pywell et. al., 1995; Burke, 2008), and the use of non-native species in seeding and transplanting treatments does not lead to the succession of native species, even in the long term (Scherrer & Pickering, 2006; Hagen & Hansen, 2014). These results drive home the importance of sourcing native species, from nearby locations, for any restoration project as well as re-establishing an organic layer of soil from which these barrens species can thrive.

3.1.5 Current challenges

Restoration ecology is a relatively new field of study and attempts to address critical issues surrounding the degradation of an ecosystem but also offers the opportunity to explore ecology theory through direct manipulations of the environment. Unfortunately, there appears to be a disconnect between ecological theory and practice (Cole, 2004; Wainwright et. al., 2018) and restoration practices do not appear to be advancing as they have barely changed in the past 30 years (Le Roy, 2018). Conceptual frameworks for the relationships between recreation, soil, and vegetation need to be further defined and refined (Cole, 2004).

To add, restoration goals are often too idealistic and fail to define realistically feasible and achievable goals (Ehrenfeld, 2000). Restoration goals can also vary from different perspectives. Some goals involve the preservation of rare or endangered species, while others focus on recreating pre-disturbance conditions, and yet others are concerned with landscape-scale restoration of ecosystem services such as the restoration of watersheds (Ehrenfeld, 2000). Restoration projects also require a significant passage of time to determine the long-term effects of the treatments. Most restoration data are described within a time span of less than five years (Le Roy et. al. 2018) which may not be enough to determine the full impacts of the restoration treatments on the area. This likely contributes to the apparent stagnation of the field.

When restoration practices focus solely on improving biodiversity, ecosystem services are not always restored (Bullock et. al., 2011). However, the definition of ecosystem service is not always clear and can vary from site to site leading to criticism of the concept (Erhenfeld, 2000). In conjunction, the very definition of success in a restoration project is not always clearly defined or varies from project to project (Prach et. al., 2019). Prach et. al. (2019) suggest that all restoration goals should be met with specific and measurable restoration targets and subsequently followed up with clearly defined, measurable indicators of success.

Another issue in restoration projects is that many projects, especially those carried out by the private sector, do not on report costs which can strongly affect the feasibility of the treatments with respect to the restored ecosystem services (Bullock et. al., 2011). Including cost data would provide meaningful information to future restoration projects on the time, cost, and feasibility of the treatments should be considered alongside the effectiveness of the treatments in producing vegetation cover, soil stabilization, etc.

Overall, a significant amount of research has been done in the field of restoration ecology since it began in the 1960s. Many strategies have been implemented in barrens and alpine communities around the world, but have not been tested, to our knowledge, in Nova Scotia. Using the knowledge gained from these past studies in related environments, as well as incorporating key concepts of ecological theory will be extremely relevant to the subject of this thesis: a restoration project that took place on a closed portion of the Skyline hiking trail in Cape Breton Highlands National Park, Nova Scotia. While this portion of trail has been closed for more than 10 years, there has been no natural recovery of the headland barrens vegetation, prompting the need for active restoration techniques to identify potential methods that could be implemented at a large scale in the future to speed up the recovery process.

3.1.6 Project objectives

Results from Chapter 2 determined that both soil and vegetation were affected on closed portions of trail at both Skyline and Mica Hill. Due to time constraints, restoration strategies could only be tested at Skyline. At Skyline, the substrate was highly compacted, reduced, and nutrient poor, and so topsoil addition was included as part of the treatments. Additionally, trails were exposed with little vegetation cover and subject to erosion, and so transplants were included as well. Lastly, the closed sections of trail at Skyline had no viable seed bank, and so seed addition was also included.

To determine potential successful restoration strategies in Cape Breton Highlands National Park, experimental restoration treatments were implemented on the closed sections of the Skyline hiking trail. Replicates of five treatments were set up along 5 m stretches of the closed trail. The project aimed to determine which of the 5 treatments contributes to the largest increase in plant cover and quality over the course of the first growing season. To add, as the project post-restoration monitoring will be conducted by Parks Canada for at least the next 5-10 years, two commonly used methods to collect vegetation cover data were compared for their similarity and determine which is most appropriate for the long-term monitoring. This project included the design, baseline data collection and implementation of restoration treatments and monitoring during the first year.

3.2 Methods

3.2.1 Seed collection and germination

Native and locally sourced species were emphasized to minimize the introduction of new genetic material and potentially invasive species (Bucharova et. al., 2017; Rydgren et. al., 2017). Seeds were collected from Skyline and Mica Hill, and various nearby locations within the Park. Chosen species were those who were most abundant within quadrats at Skyline, who were present across many quadrats, and who had readily available seed. These included lowbush blueberry, *Vaccinium angustifolium*, bunch berry, *Cornus canadensis,* three-toothed cinquefoil, *Sibbaldiopsis tridentata*, wavy hair grass, *Deschampsia flexuosa*, and poverty oat grass, *Danthonia spicata. V. angustifolium, C. canadensis, S. tridentata,* and *D. flexuosa* were all found in high abundance and presence in the reference undamaged conditions at Skyline (Table 2.1). While *D. spicata* was not among the most abundant in the reference vegetation at Skyline, it was found there and was common on the trail. It is also known to be a highly disturbance-tolerant species that is present in many successional habitats and thus is capable of thriving in stressed systems. All species were also chosen because they had easily accessible seed sources and were known to be relatively easy to germinate and grow in large quantities under greenhouse conditions.

Collected *D. flexuosa*, *D. spicata*, and *S. tridentata* were air-dried before being cleaned using a 2 mm sieve and then stored dry in a fridge at 4 °C. *C. canadensis* and *V. angustifolium* seeds were extracted from berries using a blender and water. Once blended, viable seeds settled to the bottom and water was slowly drained off and seeds were spread out on paper towel to dry. *C. canadensis* was then warm moist stratified for 1 week before being cold moist stratified for 3 weeks. *V. angustifolium* was cold moist stratified for 2 months. Water used for stratifications was collected rainwater, which has a lower pH than tap water and is more akin to the acidic environment of Nova Scotian barrens. At the end of the stratification period, *C. canadensis* and *V. angustifolium* were then air dried before being stored dry in a fridge.

All species were germinated using in trays of Pro-Mix LP15 multi-purpose growth medium. Trays were free-draining. For the entirety of the germination process, collected rainwater was used to water the trays to promote germination of the ericaceous species (Cullina, 2002). Due to variations in germination times and growth requirements, seed germination began at different times for each species (Table 3.1). In January 2019, seedlings were transplanted into small pots and put in a greenhouse located on the Saint Mary's campus, Halifax. Grass species, *D. flexuosa* and *D. spicata,* were transplanted using Pro-Mix BX Mycorrhyzae general purpose growth medium and Ericaceous species *V. angustifolium*, *C. canadensis,* and *S. tridentata* were transplanted using the original growth medium. Ericaceous species are known to have a distinct community of ericoid mycorrhizae and thus to avoid any potential interference with other mycorrhizae, they were not transplanted using the Pro-Mix BX medium as this was designed for crop plants and includes arbuscular mycorrhizal fungus inoculum (not ericoid mycorrhizae). Due to the increased volume of water needed to saturate the plants in the greenhouse, standard water from a hose was used. *C. canadensis* germination rates were low and most did not survive the transplanting process and so was not a part of the transplanting treatments, but was still be present to a lesser degree in the seeding treatments.

Table 3.1. Dates seeds were sown

Species	Date sown
Vaccinium angustifolium	October 8th and 22nd, 2018
Cornus canadensis	November 5th, 2018
Sibbaldiopsis tridentata	November 5th, 2018
Danthonia spicata	November 19th, 2018
Deschampsia flexuosa	November 19th, 2018

3.2.2 Restoration treatments

In Summer 2019, restoration treatments were implemented on the portion of the Skyline trail as outlined in Chapter 2 (Figure 3.1). This involved four different strategies: soil addition, seeding, transplanting, and the use of biodegradable erosion control mats. Treatments were not tested on Mica Hill due to time constraints.

Figure 3.1 Drone-captured image of the Skyline headland and restoration project. Photo taken by Samantha Howard, in July 2019.

Experimental design:

The four strategies were combined to create five different potential restoration treatments: 1) control (no treatment), 2) transplant only, 3) topsoil addition with erosion control matting, 4) topsoil addition with erosion control matting and seed, and 5) topsoil addition with erosion control matting and transplant. Each treatment was repeated 5 times and spanned 5 m segments, or sites, of trail (length). Due to variations in trail width, each site had a variable area and so treatments were applied based on consistent planting and seeding densities. On average, each site was approximately 6 m^2, and approximately 150 m^2 of trail was used to form part of the experiment. All 25 sites were set up on a relatively uniform section of trail, and semi-randomly ordered. To allow for spatial variability, every group of 5 sites had one of every type of treatment, but were randomly ordered within each grouping (random block design with each group of 5 sites representing a block). In this manner, each type of treatment was present over most of the experimental area to account for variance with respect to physical location on the trail.

A seeding-only treatment was not chosen for this study as a previous study by Ebersole et. al. (2004) showed that seeding with no matting is only marginally better than having no treatment. A topsoil-only treatment (with no erosion control matting) was not implemented in the experimental design, however, two approximately 10 m sections were covered in topsoil and simply observed throughout the duration of study with very little success. By the end of the growing season, the exposed added topsoil had been nearly completely eroded.

Treatment specifications:

Soil addition was required for three sets of treatments and thus covered 15 sites. Enough topsoil was added to completely fill in the depth of the trail to match the soil level of adjacent vegetation. Therefore, topsoil volume was variable between sites. When able, some of the pre-existing soil was raked and partially mixed in with the new topsoil in an attempt to create a soil gradient rather than an abrupt dichotomy between soil types. The topsoil chosen for this study was a bagged Scotts® Pro Blend Top Soil. This is a pre-blended topsoil comprised of topsoil, compost, and manure, and has a fertilizer rating of 0.05- 0.01- 0.05 (N-P-K), and pH of 6.0-6.5. A commercial, bagged topsoil was chosen for ease of transport as the soil needed to be manually carried part-way to reach the closed trail section where the restoration project took place.

The erosion control material used was TerraFix Coir Mat 400. This 100% biodegradable coir mat has a typical lifespan of 36-72 months before biodegrading. Erosion control mats anchor the soil in place and provide buffering from the wind and other elements for seedlings. The mats were installed following topsoil application and seeding where is the case, and prior to transplanting. Mats were anchored using metal pin staples, hammered in at roughly 60 cm intervals.

Seeds are only required for one set of treatments and thus covered 5 sites in total. Seed mixtures were created by separating what seed was collected in August of 2018 and remained after germination. All sites have different areas and thus to maintain consistency, seed were distributed to achieve consistent density. Seed from each species were thus divided into groups that equate to an even density across sites. The density was 11.15 g of seed/m^2 or roughly 10 200 seeds/m^2. Density by species was approximately: 100 seeds/m^2 *C. canadensis*, 280 seeds/m^2 *V. angustifolium*, 1400 seeds/m^2 *S. tridentata*, 3400 seeds/m^2 *D. spicata*, and 5000 seeds/m^2 *D. flexuosa.* Seeds were distributed as evenly as possible over the first 5 cm layer of topsoil for every site before the erosion mat was immediately installed.

Transplanting was required for two sets of treatments and thus covered 10 sites. Similar to the seeding treatment, plants were distributed evenly across sites based on density. Table 3.2 shows the amount of plants, per species, that were used in the study. Overall, plant density was roughly 19.8 plants/m^2. This was calculated based on the total area the treatments needed to cover, space availability in the greenhouse, and the number of individuals per species that germinated and were able to survive until the treatments could be set up.

Table 3.2. Number of individual plants, per species, that were transplanted on Skyline.

Species	Number of individuals
Vaccinium angustifolium	333
Danthonia spicata	379
Deschampsia flexuosa	231
Sibbaldiopsis tridentata	237

3.2.3 Monitoring protocol

Once all treatments were established, sites were monitored every three weeks until mid-September 2019. For every treatment site there were two fixed 0.5 m x 0.5m quadrats, placed at 1 m and 3 m from the beginning of each site (Figure 3.2). Plant abundance data were collected using both the frequency method and percent cover method. For the frequency method, the quadrat was divided into 25 0.1 m x 0.1 m sub-quadrats. Species and seedlings were identified by presence or absence in each sub-quadrat. Frequency is a common method to monitor vegetation and is repeatable across different observers, making it a good choice for a long-term project. The frequency method was implemented every 3 weeks following the set-up of all treatments, for a total of 5 monitoring events.

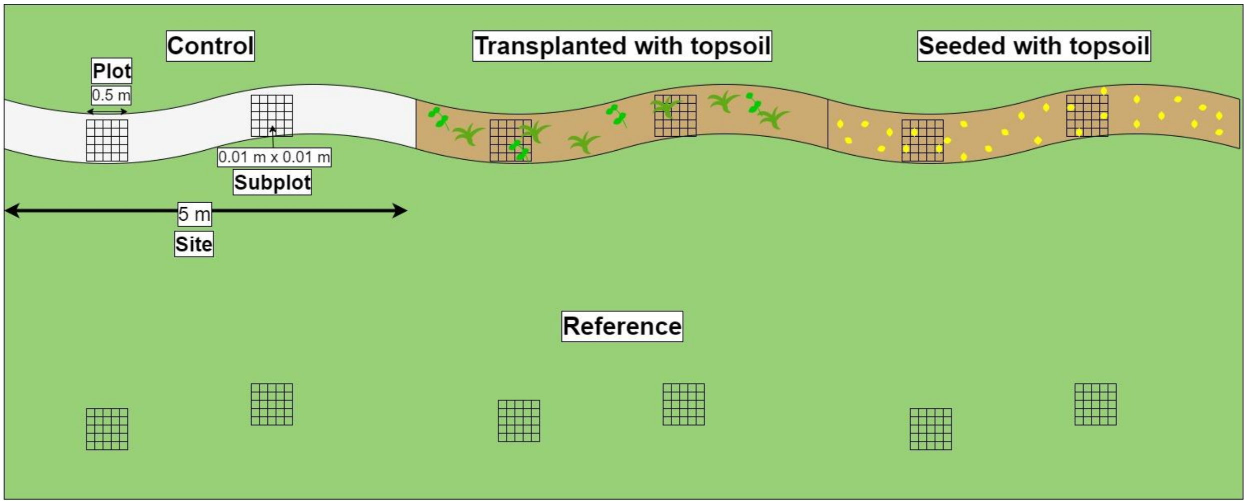

Figure 3.2. Trail diagram depicting site, plot, and subplot. Only three of the five treatments types are represented. Frequency cover was calculated as presence or absence in each subplot.

To add, percent cover evaluation was also implemented as outlined in section 2.2.2 during the first and last monitoring events (Week 1 & Week 13) in order to be comparable to data collected from the previous year. For every site, each of the two 0.5 m x 0.5 m plots on the treated area was paired with reference plots 3 m away from the edge of the trail, for a total of 50 plots. Plant abundance estimates in all plots were recorded using both the frequency method and the percent cover method to allow for comparison between both types of data sets. Monitoring using both methods can be used to determine their relatedness and help guide method choice for an extended long-term monitoring plan.

For every treatment where transplanting occurred, a species health was calculated by measuring a health index for every individual, for each of the four transplanted species (*V. angustifolium*, *S. tridentata*, *D. spicata*, and *D. flexuosa*). The health index was measured on an ordered categorical scale from 0 to 4, every three weeks. For this method, 0 indicates 0% of plant matter is alive, 1 indicates that 1%-25% of plant matter is alive, 2 indicates that 26%-50% of plant matter is alive, 3 indicates that 51%-75% of plant matter is alive, and 4 indicates that 76%-100% of plant matter is alive (Anastasiou & Brooks, 2006). In this case, an index of 0 described dead or missing individuals as it related to the initial amount planted. By repeating these measures over the course of the growing season, it was possible to identify trends in growth rates as well as death rates of the vegetation and determine the performance of each species.

3.2.4 Statistical analysis

Equivalent to Chapter 2, data were summarized and analyzed using R 3.6.1. Data analysis and model selection were based on the analysis of general linear models using a Bayesian approach based on the rstan and loo packages. Bayesian computation involves a process whereby initial prior beliefs about a model system are updated by data to produce an outcome of posterior probabilities that describe the relationship of factors specified within the model system (Kruschke, 2015; McElreath, 2015). Model diagnostics were performed and included: running multiple chains to check for convergence (Rhat = 1), ensuring an effective sample size, $N_{eff} > 5000$, plotting posterior distributions against priors to ensure the priors are appropriately overwhelmed, and generating a new dataset from the model and comparing it against the original to ensure the data are being appropriately modeled.

Plot Frequency

Frequency counts at the plot level were observed at every 0.1 m x 0.1 m subplot, for a total of 25 observations. At several locations along the trail, quadrats overlapped off the trail onto adjacent, undamaged vegetation. For the sake of this analysis, the two outermost columns of observations were removed from the dataset, resulting in a total count of 15 observations (Figure 3.3). This was done to get a more accurate representation of treatment success.

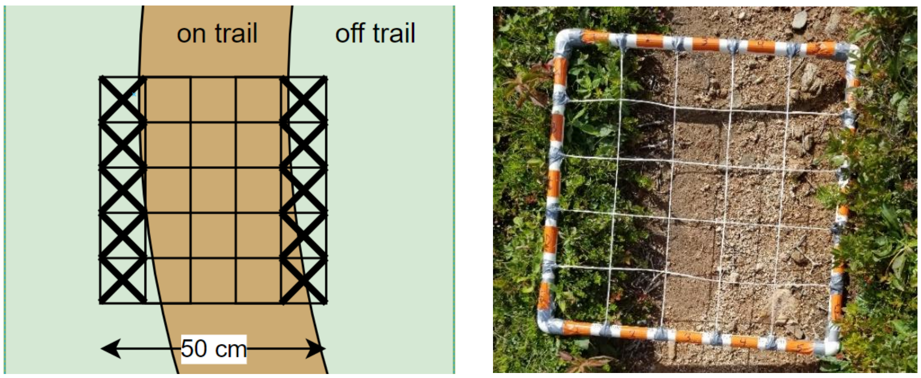

Figure 3.3. Left: Diagram of a single plot divided into 25 10 cm x 10 cm subplots. Subplots with a black "X" are those that were removed from analysis as many quadrats were not always fully on the trail. Right: Example photo of plot 14-A showing edge overlap, taken on Sept 13th, 2019.

To compare frequency cover across treatments over the course of the growing season, a count value (up to 15) was used as the predicted variable and treatment as well as time effects were used as predictor variables. Specifically, the five treatment categories were examined: passive control, topsoil addition with erosion control, direct transplanting, transplanting with topsoil and erosion control, and seeding with topsoil and erosion control. Data were collected on five separate occasions across a spatially varying environment.

A Bayesian model was designed with an count predicted variable where values were positive integers. With count predicted variable, the appropriated likelihood is an

Poisson distribution, where the probability of plant cover in each treatment, over time, is based on the probability, *p*, which is estimated from the data As *p* can only be positive, the predictor variables must be linked to the predicted variable in a modified way. With a Poisson distribution, the appropriate link between the predictor and predicted variables is the log link, which ensures that the combined effects of the predictor variables are positive. This can be implemented in Stan using the poisson_log likelihood function. Specifically, the likelihood equation used was:

$$p = \beta_0 + \sum_j \beta_{1[j]} x_{1[j]} + \sum_k \beta_{2[k]} x_{2[k]} + \sum_{j,k} \beta_{1\times 2[j,k]} x_{1\times 2[j,k]}$$

$$y \sim \text{Poisson\_log}(p)$$

Where:

β_0 is the base effect, or intercept. β_1 is the ordered effect of being at a particular round of data collection, *j*, (eg. Week 1, Week 7, etc.) and β_2 is the effect of being in a particular treatment, *k*, (topsoil addition, topsoil and seed addition, etc.). A two-way interaction term, $\beta_{1\times 2}$, was included in this model to understand the combined effects between the round of data collection (ie. passage of time) and the particular treatment used.

All predictor variables had hierarchical structure. Thus, the prior means were sampled from a hyperprior with a mean centered at zero and a standard deviation of 1, and the prior standard deviations were sampled from a hyperprior with a mean centered at 1 with a standard deviation of 1 (to maintain positive variance) (Figure 3.4). Thus, they were assumed to have no effect, but with little weight on this assumption and have information sharing across categories within each variable.

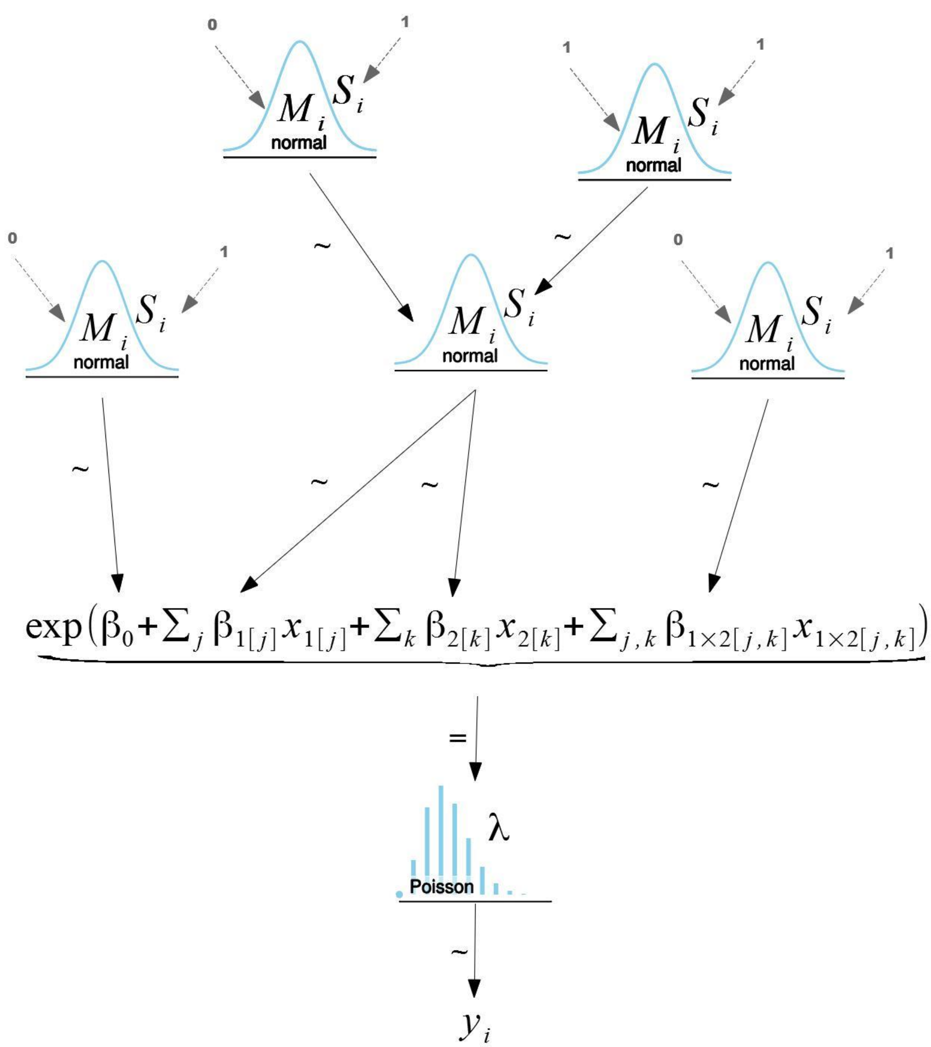

Figure 3.4. Diagram representation of Bayesian hierarchical model used to compare plot frequencies with treatment type, and round of data collection.

Transplant health index

To compare health indices across transplanted species in two treatments over the course of the growing season, a 0 to 4 health index was used as the predicted variable and species, treatment and time effects were used as predictor variables. Specifically, four species were compared, *D. spicata, D. flexuosa, S. tridentata,* and *V. angustifolium*, across two treatments, direct transplanting and transplanting into added topsoil with erosion control. Data were collected on 5 separate occasions across a spatially varying environment. Maximum likelihood model selection processes using AIC and also WAIC determined that the location effect of site, but not block (as per the random block design of the study), was an important factor and thus was included in the model.

A Bayesian model was designed with an ordinal predicted variable as a plot health index scale from 0 to 4, where $0 < 1 < 2 < 3 < 4$. With an ordinal predicted variable, the appropriated likelihood is an Ordered distribution, where the probability of each index value below the maximum value is based on the probability, *p*, which is estimated from the data. With an Ordered distribution, the appropriate link between the predictor and predicted variables is the logistic link, which ensures that the effects of the predictors variables are on a cumulative odds scale. This can be implemented in Stan using the ordered_logistic likelihood function. Specifically, the likelihood equation used was:

$$p = \beta_0 + \sum_j \beta_{1[j]} x_{1[j]} + \sum_k \beta_{2[k]} x_{2[k]} + \sum_m \beta_{3[m]} x_{3[m]} + \sum_n \beta_{4[n]} x_{4[n]}$$

$$+ \sum_{j,k} \beta_{1\times2[j,k]} x_{1x2[j,k]} + \sum_{j,m} \beta_{1\times3[j,m]} x_{1x3[j,m]} + \sum_{k,m} \beta_{2\times3[k,m]} x_{2\times3[k,m]}$$

$$+ \sum_{j,k,m} \beta_{1\times2\times3[j,k,m]} x_{1\times2\times3[j,k,m]}$$

$$y \sim \text{ordered\_logistic}(p, \text{c})$$

Where:

β_1 is the ordered effect of being at a particular round of data collection, *j*, (eg. Week 1, Week 7, etc.), β_2 is the effect of being in a particular treatment, *k*, (transplanting in topsoil vs direct transplanting, etc.), β_3 is the effect of being a particular transplanted species, *m*, (*D. spicata*, *V. angustifolium*, etc.), and β_4 is the effect of being at a particular site, *n*.

A three-way interaction term, $\beta_{1\times2\times3}$ was included in this model to understand the combined effects between the round of data collection (ie. passage of time), the particular treatment used, and each species transplanted. As a result, all lower-order two-way interaction terms were included. All four of the predictor variables had hierarchical structure to allow for information sharing across the categories within each variable.

The prior probabilities for β_1 through β_4 were all normal distributions with means of 0 with a standard deviation of 1. Thus, they were assumed to have no effect, but with little weight on this assumption. All predictor variables were standardized to a mean of 0 and standard deviation of 1.

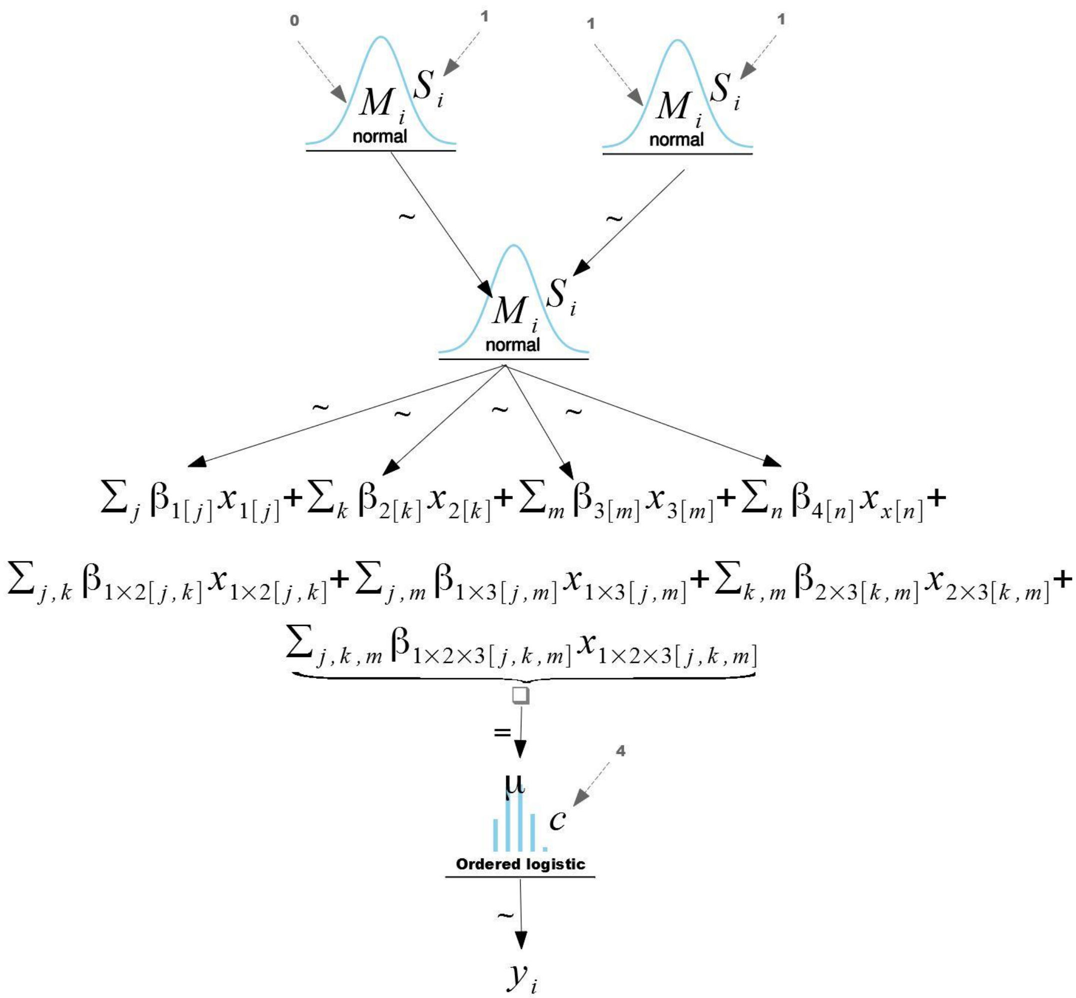

Figure 3.5. Diagram representation of Bayesian hierarchical model used to compare transplant health indices with treatment type, round of data collection, species, and site location. For the sake of simplicity, priors for interaction terms are not shown but all follow a normal distribution with a mean of 0 and standard deviation of 1.

3.3 Results

3.3.1 Plot frequency

Treatments were monitored over the Summer 2019, every three weeks, from mid-June to mid-September. The main species identified were *D. spicata*, *D. flexuosa*, *V. angustifolium*, *S. tridentata*, *A. uva-ursi*, and *S. bicolore* (Figure 3.6). These species were found across all treatments, some of which were either already present on the site, were intentional transplanted treatments, or germinated from the seeded treatments.

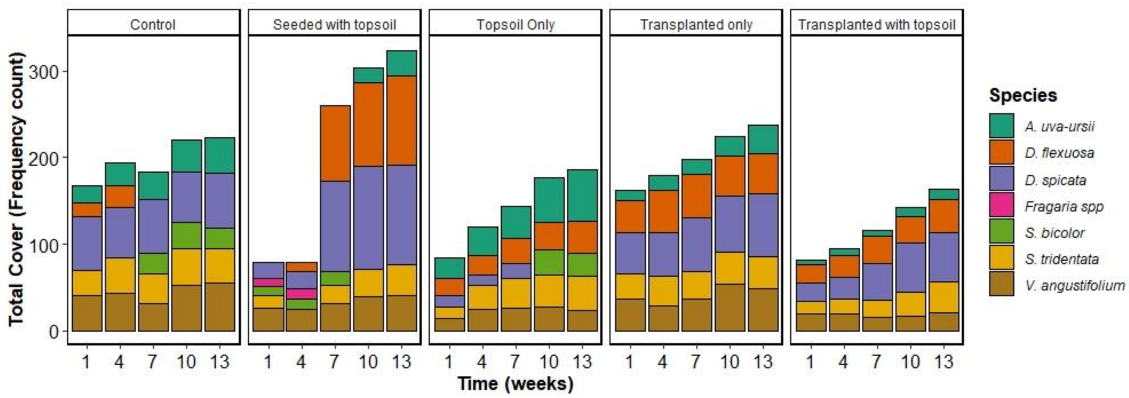

Figure 3.6. Total frequency count of the top 5 most abundant species across the five replicates of each treatment, for every week of data collection. Data used here included all 25 subplots for each of 2 plots at every site and there are 5 replicates per site, for a total of 250 recordings for any given species in any given treatment for any given week of data collection.

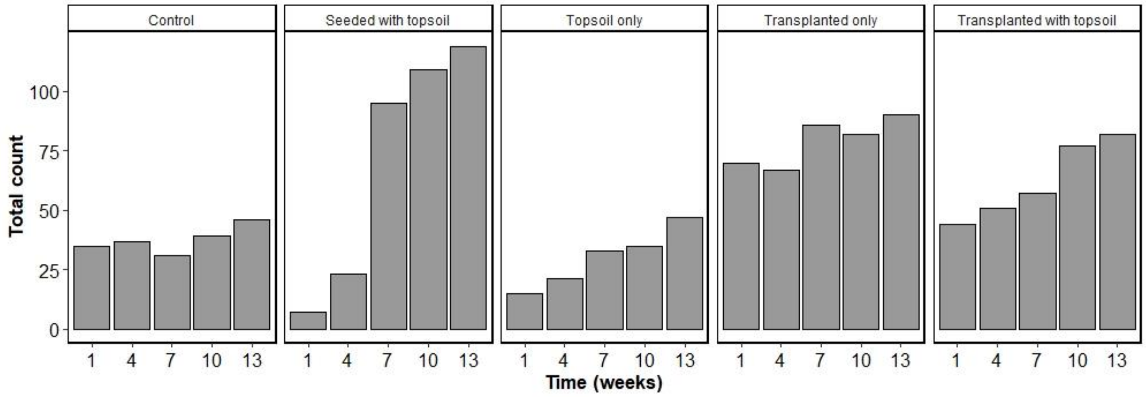

Figure 3.7. Summary of frequency cover counts, separated by the five treatments: control, seeded-with-topsoil, topsoil-only, transplanted-only, and transplanted-with-topsoil, for each week of data collection. There are 15 subplots for each of 2 plots at every site and there are 5 replicates per site, for a total of 150 recordings for any given treatment at any given week of data collection. These were the data included in the statistical model.

Overall, cover improved over the course of the study in all five treatments (Figure 3.8 & Figure 3.9). The control improved by 32% from 3.56 to 4.59. The topsoil-only treatment had an approximately threefold (3.04) increase for 1.53 to 4.66. The transplanted-only treatment improved by 29% from 6.93 to 8.95. The transplanted-with-topsoil treatment improved 86% from 4.36 to 8.14. Lastly, the seeded-with-topsoil treatment had an approximately 14-fold (14.38) increase from 0.82 to 11.79

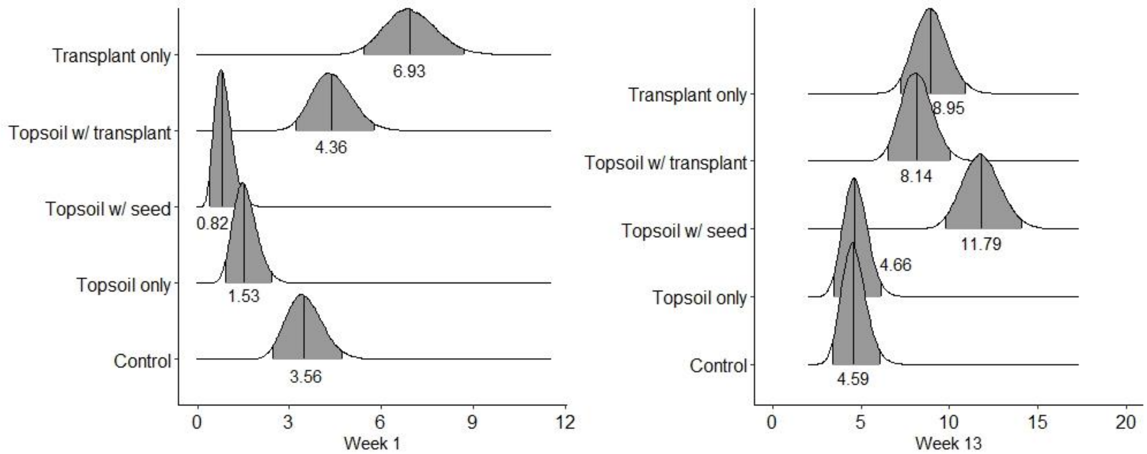

Figure 3.8. Initial (Week 1) and final (Week 13) frequency estimates for each treatment. Posterior distributions have been exponentiated to return to the normal scale. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals.

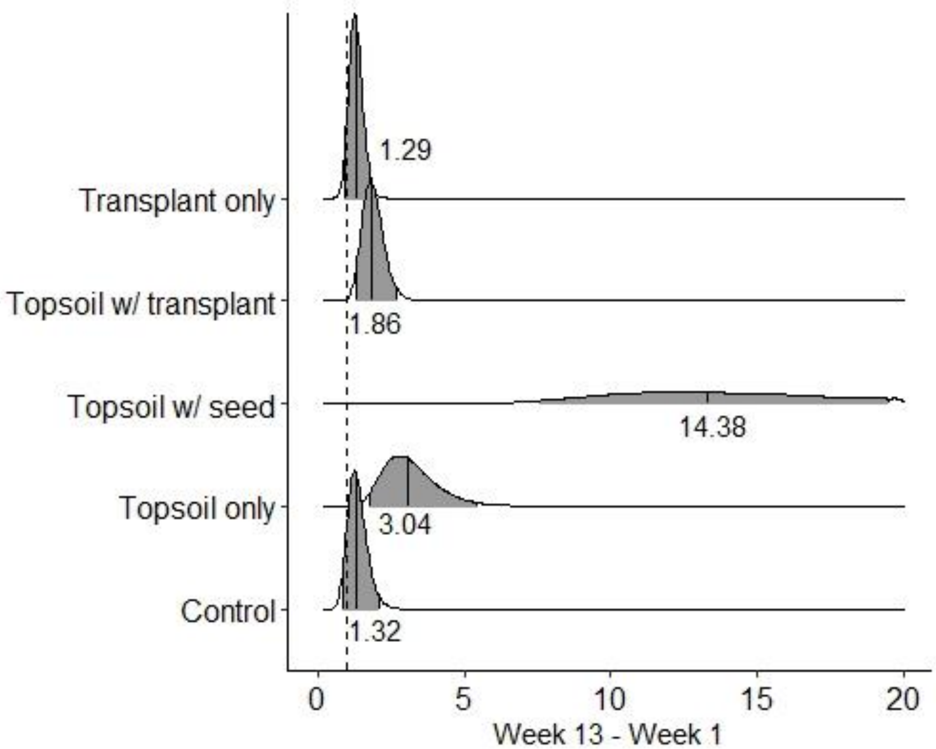

Figure 3.9. Difference from first to last week of data collection, for each treatment. Posterior distributions were then exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a proportional ratio of 1:1.

Treatments were compared to the control treatment and examined at the beginning and end of the study (Figure 3.10). The directly transplanted treatment consistently had about twice as much cover than the control treatment. The transplanted-with-topsoil treatment initially had similar cover to the control but, had about 81% more cover by the end of the study. The seeded-with-topsoil treatment and topsoil-only treatment initially had lower cover than the control, 75% less and 54% less, respectively. but by the end of the study the seeded treatment cover was more than twice that of the control treatment (2.64), whereas the topsoil-only treatment was no different than the control treatment (1.04). Thus, both seeded and topsoil-only treatments improved over the course of the study, more so than the control treatment as the transplanted-with-topsoil treatment had a tendency to have 29% more cover than the control treatment, in the first week (Figure 3.10).

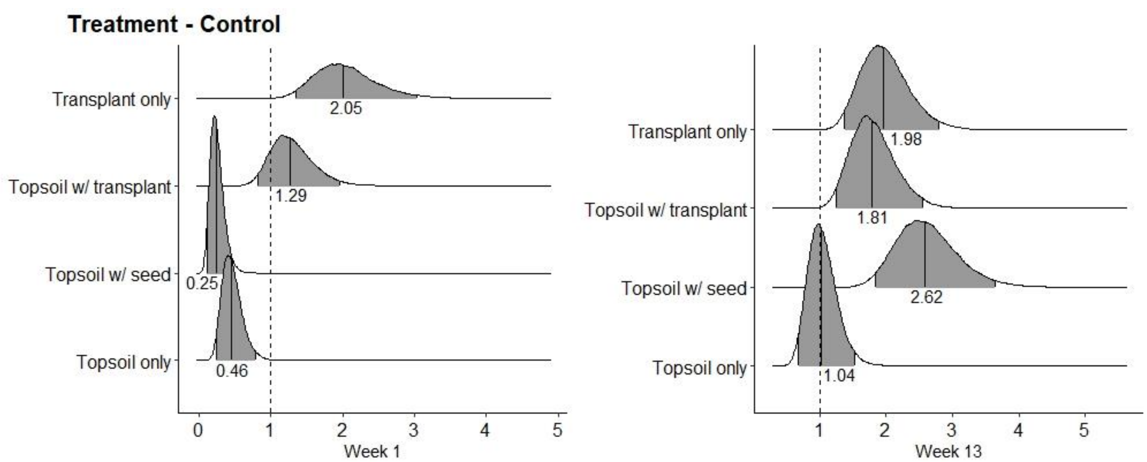

Figure 3.10 Change in cover as compared to the control treatments (passive baseline) for each active treatment. Left: initial difference during the first week of data collection. Right: final difference during the last week of data collection. Posterior distributions were then exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a ratio of 1.

In the first week of the study, the directly transplanted and transplanted-with-topsoil treatments had the highest cover, followed by the control, and then by the topsoil-only and seeded-with-topsoil treatments (Figure 3.8). In the final week of the study, the seeded-with-topsoil treatment had the highest cover, followed by both treatments where transplanting took place, and then by the topsoil-only and control treatments (Figure 3.8). The seeded-with-topsoil improved the most, where the main germinations observed from the seeded-with-topsoil treatment were *D. spicata* and *D. flexuosa*. The directly transplanted treatment did not improve any more than the control, whereas both transplanted-with-topsoil and topsoil-only treatment did improve at a rate higher than the control.

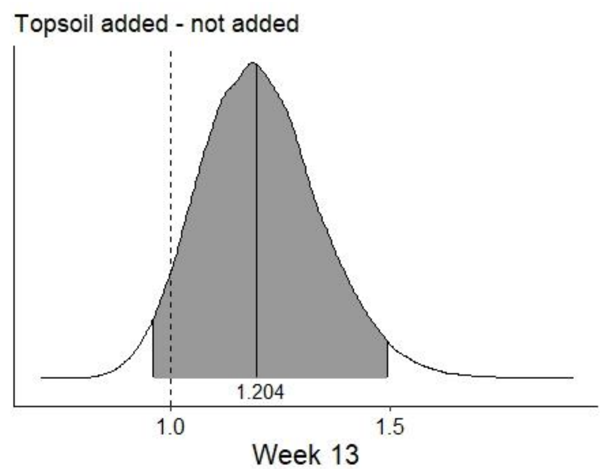

Figure 3.11. Difference in cover comparing all treatments were topsoil was added against all those who did not. for each active treatment. Posterior distributions were then exponentiated to represent ratios. Displayed values represent the median frequency estimate. Shaded regions represent 95% highest density intervals. The dashed vertical line at 1 represents a ratio of 1.

Treatments where transplanting occurred had an immediate tendency to have higher cover when compared to the control (Figure 3.10). Treatments requiring the use of topsoil initially experienced a reduction in cover but improved to be equivalent or higher than that of the control, indicating that treatments where topsoil was added improved more so than that of the control treatment. There was a tendency for treatments where topsoil was added to increase cover by 20% at the end of the study, compared to treatments that did not require the addition of topsoil (Figure 3.11). The initial reduction in cover due to added topsoil was less than that increase in cover where transplants were added.

3.3.2 Transplant health index

Firstly, by the end of the study across all transplanted treatments, 156 out of 237 *S. tridentata* individuals survived (65.8%), 194 out of 231 *D. flexuosa* individuals survived (83.9%), 162 out of 333 *V. angustifolium* individuals survived (48.6%), and 235 out of 379 *D. spicata* individuals survived (62.0%). Note that, in some instances, there was a decreasing proportion of indices of 0 or dead individuals. This is the result of observer bias, where not all individuals were counted every round of data collection and 0s were added post-data collection. Therefore, there were fluctuations in the proportions of 0s.

When averaged across all species and rounds of data collection, there was a strong tendency for index values to be higher when species were transplanted into added topsoil compared to when they were just directly transplanted into pre-existing soil (Figure 3.13). When comparing the difference between treatments for each species, averaged across all rounds of data collection there is a tendency for health index values to be higher when plants were transplanted into added topsoil, with *S. tridentata* being the only species whose 95% highest density interval fell completely above the zero value mark (Figure 3.14).

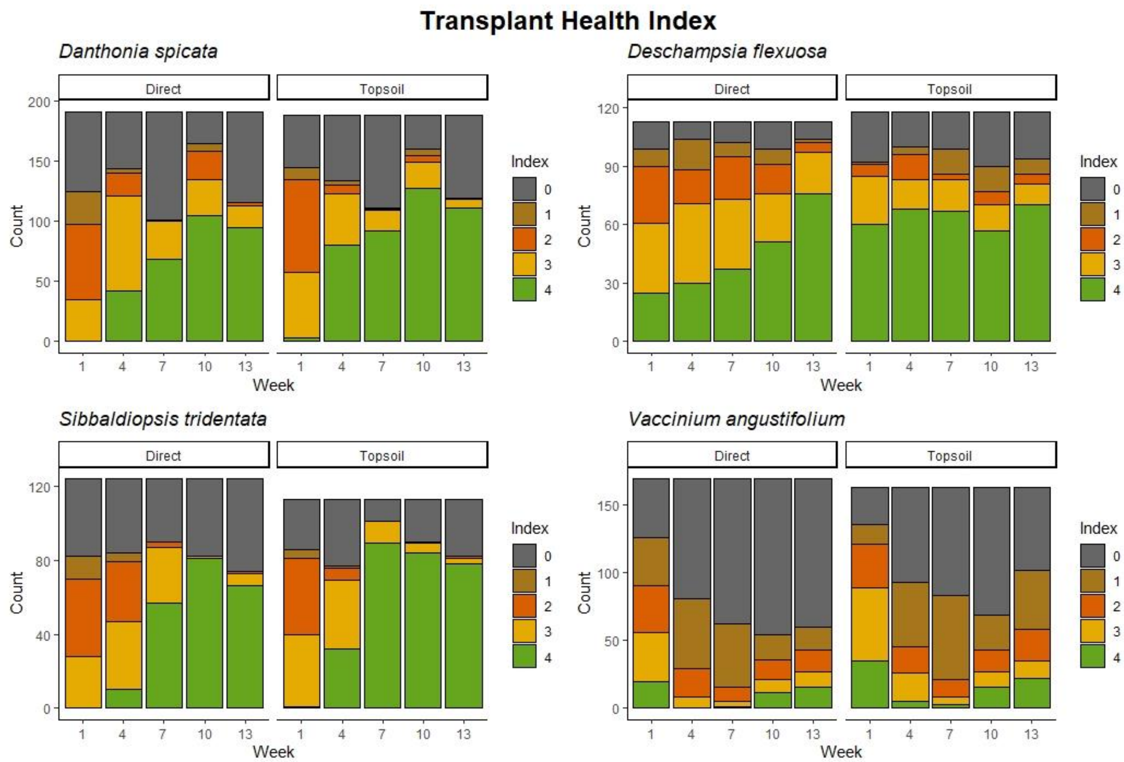

Figure 3.12. Summary count of transplant health index values, separated by the four species used: *D. spicata*, *D. flexuosa*, *S. tridentata*, and *V. angustifolium*, for both treatment types: directly transplanted versus transplanted-with-topsoil, separated across each week of data collection.

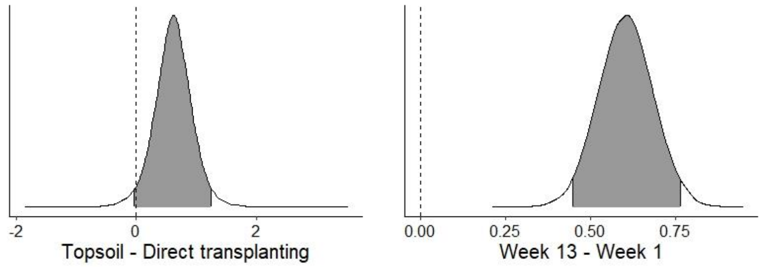

Figure 3.13. On the right: the difference in index values between topsoil and direct transplants, averaged across all species and all rounds. Positive values indicate higher index values in the topsoil treatment, negative values indicate higher index values in the direct treatment, and zero values indicate no difference between treatment types. On the left: the difference in index values between the first and last weeks of data collection. Positive values indicate higher index values in the final week, negative values indicate higher index values in the initial week, and zero values indicate no difference between first and final weeks. Shaded regions represent 95% highest density intervals.

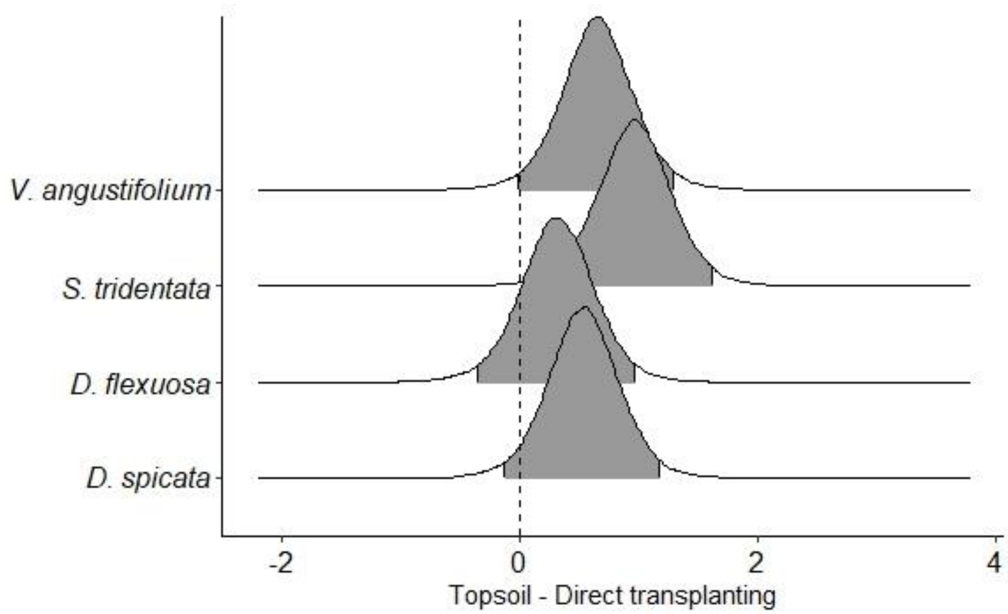

Figure 3.14. The difference between transplants in topsoil and direct transplants for each species, averaged across all rounds of data collection. Positive values indicate higher index values in topsoil transplants, negative values indicate higher index values in direct transplants, and a zero value indicates no difference between the two treatment types. Shaded regions represent 95% highest density intervals.

Secondly, when averaged across all species and treatments, health index values improved over the course of the study (Figure 3.13). When comparing the performance of each species in both direct and topsoil treatments from Week 1 to Week 13, both *D. spicata* and *S. tridentata* improved in both treatments, *V. angustifolium* worsened in both treatments, and *D. flexuosa* improved in the direct treatment, but saw no change when transplanted-with-topsoil (Figure 3.15).

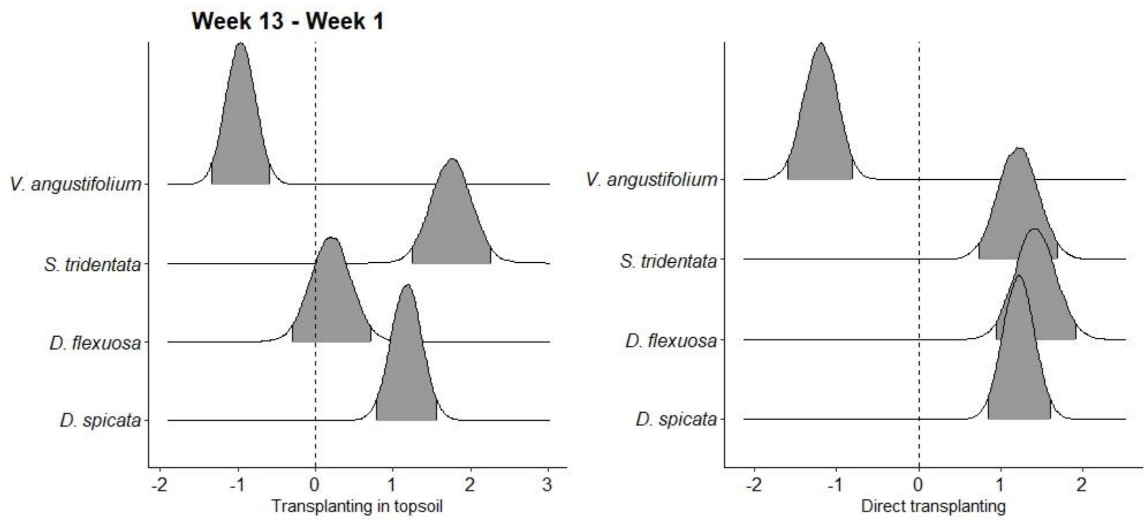

Figure 3.15. Change in transplant health index from the first week of data collection to the last week of data collection (after 14 weeks). Transplants in topsoil are on the left and direct transplants are on the right. Species are divided by rows. Negative values represent a decrease in index, positive values represent an increase in index, and a zero value represents no change between first and last rounds of data collection. Shaded regions represent 95% highest density intervals.

Initially, all species had higher quality and survival in the topsoil treatments (Figure 3.16). By the end of the study, *D. flexuosa* was the only species who performed worse in the topsoil treatment. *D. flexuosa* improved only when directly transplanted and did not change when transplanted in added topsoil. Therefore, *D. flexuosa* individuals were initially healthier in topsoil treatments when compared to directly transplanted individuals and by the end of the study, directly transplanted individuals were able to improve to a point where they were similar in health index to those transplanted-with-topsoil, and potentially more so. Both *D. spicata* and *V. angustifolium* had consistently higher health index values in the topsoil treatment throughout the study, but did not improve any more than the directly transplanted treament. Finally, not only was *S. tridentata* higher in index values at the beginning and end of study in the topsoil treatment, but they also improved more so than when they were transplanted directly.

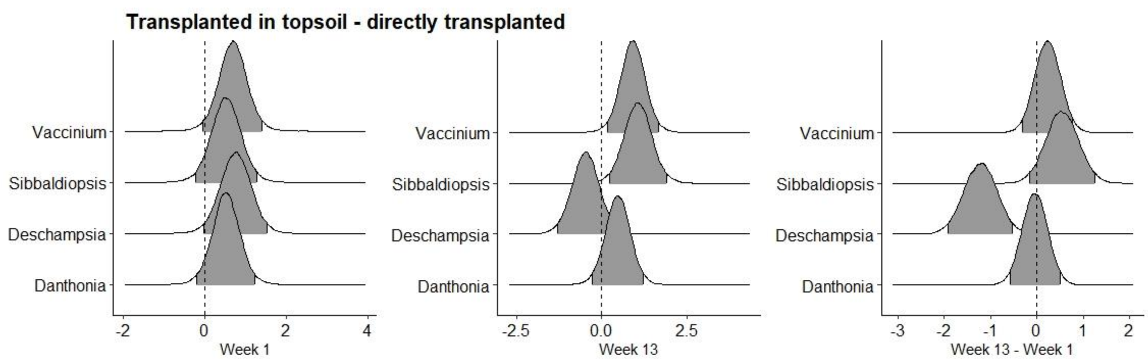

Figure 3.16. Change in transplant health index for each species, comparing the transplanted-with-topsoil treatment against the directly transplanted treatment. Left: initial difference during the first week of data collection. Centre: final difference during the last week of data collection. Right: change in index from first to last week. Positive values indicate higher index values in the topsoil treatment, negative values indicate higher index values in the direct treatment, and zero values indicate no difference between treatment types. Shaded regions represent 95% highest density intervals.

In general, *V. angustifolium* had the lowest scores of all species used. To futher investigate, the health index values across each round data collection, for both treatments of *V. angustifolium* were compared (Figure 3.17). The quality of individuals decreased from Week 1 to Week 7, remained the same between Week 7 and Week 10 for both treatmeants, but improved between Week 10 and Week 13 for the topsoil and only had a slight tendency to improve when directly transplanted.

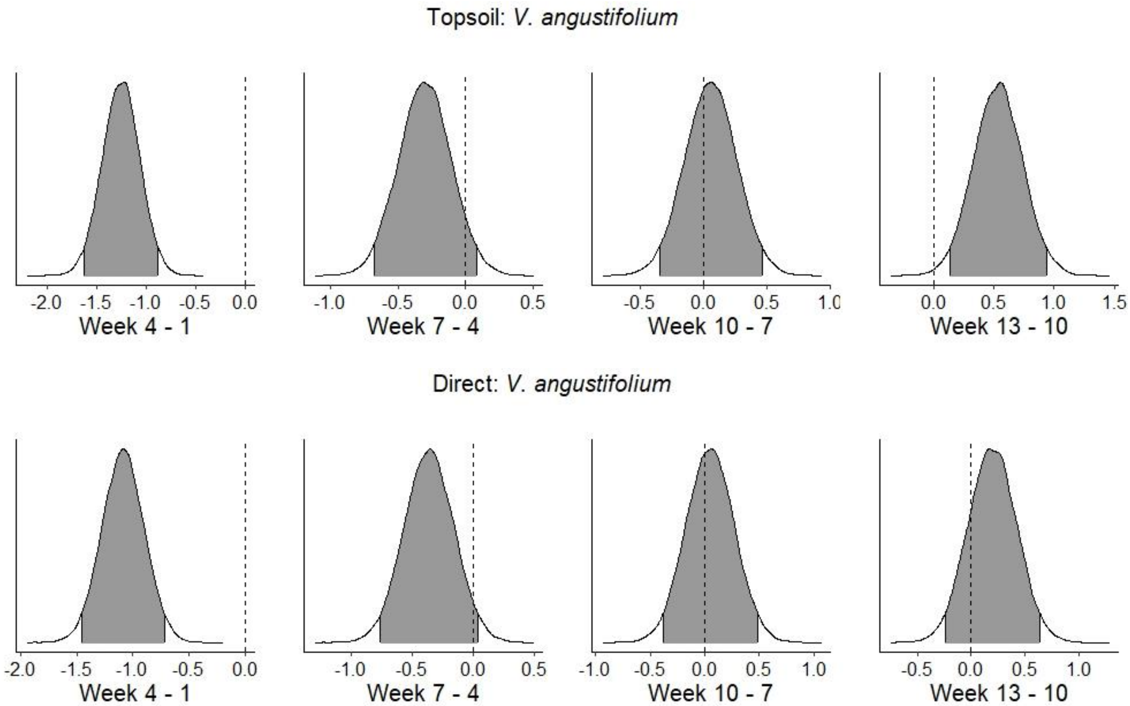

Figure 3.17. Change in index from *V. angustifolium* between each round of data collection in topsoil (upper panels) and direct (lower panels) treatments. Positive values indicate higher index values in the later round, negative values indicate higher index values in the earlier round, and zero values indicate no difference between compared rounds. Shaded regions represent 95% highest density intervals.

3.3.3 Method Comparison

Comparison of both methods of quantifying vegetation cover was recorded as a graphical analysis of both frequency cover (x axis) against percent cover (y axis) (Figure 3.18). Data points used were only for reference vegetation and did not include any data collected on the trail. Species chosen were those that were common across the site such that a high number of data points had been collected and those that represented different growth forms. The species compared were: *Vacciniuum angustifolium*, *Artostaphylos* *uva-ursi, Juniperus communis, Deschampsia flexuosa, Danthonia spicata, Sibbaldiopsis tridentata, Vaccinium vitis-idaea, Conrus canadensis, Solidago bicolor, Picae glauca, Viburnum nudum,* and *Diervilla lonicera*. In general, both frequency and percent cover methods were related to a high degree. Smaller growth forms such as *V. vitis-idaea*, *D. flexuosa*, *D. spicata,* and *S. tridentata* were strongly linearly related. Shrubbier species such as *V. angustifolium, A. uva-ursi,* and *J. communis* were related non-linearly in such a way that, at high frequency values, percent cover values tended to vary.

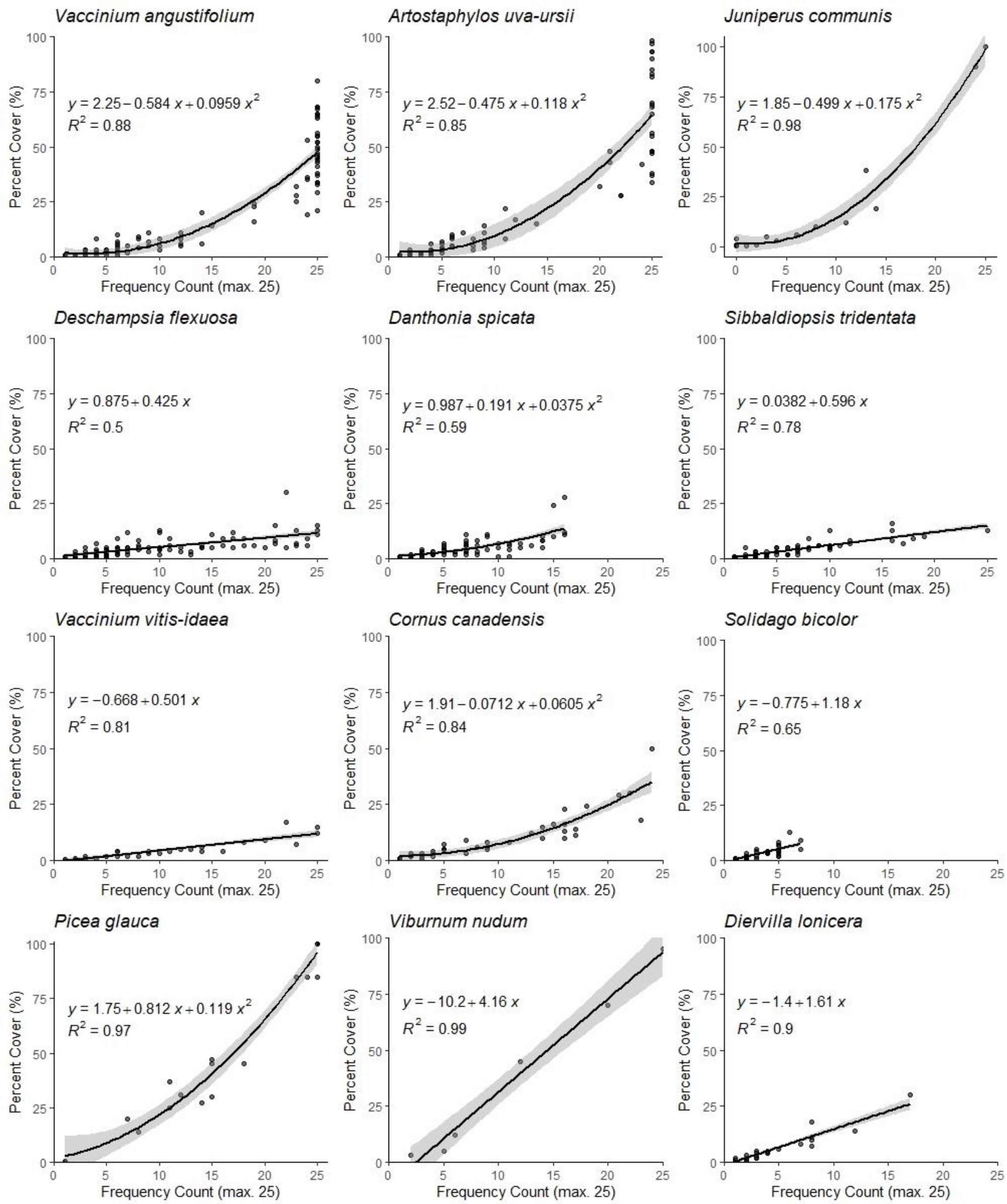

Figure 3.18. Linear correlations of the 12 most abundant species at Skyline comparing both data collection methods used: the percent cover technique and frequency count technique. Second order polynomials were used when the fit (R^2) was increased by more than 0.05. Shaded bands represent standard error.

3.4 Discussion

Summary

Firstly, the frequency cover of all treatments did improve over the course of the growing season (Figure 3.9). Therefore, transplanted species were able to survive and establish themselves, seeded species were able to germinate and grow, and topsoil-only treatments experienced encroachment. As the study began in May, before the leaf-out of many species (personal observation), there is the potential that this may have contributed to the improved cover of the control treatment, where the growth of pre-existing plants would have increased the overall cover in the quadrat.

As plants were found to be growing in the control (i.e. frequency was not 0), the key metric to test success is the relative difference in index values between the active treatments and the control (Figure 3.10). Every treatment improved compared to the improvement in the control treatment, except for the directly transplanted treatment and so all active treatments involving the use of topsoil did improve the growth of vegetation after only one growing season. The addition of topsoil with the erosion control mat, while initially reducing cover, did improve the final quality and cover of vegetation. It is important to note that every time topsoil is mentioned, it also includes the use of an erosion control mat. Both transplant treatments initially immediately increased health index values, likely through increased cover from the added plants. At the end of the study, all treatments except for the topsoil-only treatment were better than the passive control treatment.

Secondly, *D. flexuosa*, *D. spicata*, and *S. tridentata* were able to survive at high rates in both topsoil treatments and direct treatments while *V. angustifolium* performed poorly in both with significantly lower survival rates. All species were initially better with topsoil addition. *V. angustifolium* performed better in topsoil conditions despite its low success. *S. tridentata* was performed better in topsoil conditions and improved in topsoil more so than when directly transplanted. *D. spicata* performed better in topsoil conditions and improved consistently in both treatments. *D. flexuosa* performed well in topsoil treatments but was able to improve and did somewhat better when directly transplanted. General small fluctuations in the number of observed species between weeks of data collection is likely due to human observation error where some individuals may have been missed. Nonetheless, the addition of topsoil did improve the quality and survival of all four transplanted species.

Topsoil addition initially reduces cover

Adding topsoil reintroduces vital nutrients, including N, P, and K, that are essential for plant growth. There is potential that after many years, the surrounding vegetation will be able to recolonize with the help of the added nutrients found in the added topsoil. In the breakdown (Figure 3.10), it can be seen that, while initially the cover of vegetation was worse in topsoil treatments, by the end of the study treatments using topsoil outperformed those that did not use it (Figure 3.11).

It is likely that the addition of topsoil buried individuals and resulted in a reduction in overall cover. A study by Rivera et. al. (2014) found similar results where the addition of topsoil to a bank reduced cover in the first year when compared to a control, was on par the second year, and had increased cover in the third year. The trend of initial reduced cover was not seen when the transplanted-with-topsoil treatment is compared to the control since the addition of transplants more than offset the loss of plants via burial. The trend is observed, however, where the transplanted-with-topsoil treatment had a cover of approximately 4.36 is compared to the directly transplanted treatment with a cover of approximately 6.93 (Figure 3.8), where both transplanted treatments were planted with the same density of plants.

By the end of the study, the topsoil-only treatment was not any different than that of the control treatment, but since it was initially worse than the control, there was an improvement overall and more so than that the improvement of the control. This is evidence that nearby vegetation was better able to expand into the topsoil than the pre-existing degraded soil. There was no evidence of colonization of the topsoil-only treatment by seed rain (personal observation) and thus the increase in cover is likely due to encroachment of nearby pre-existing vegetation. In this sense, the topsoil-only treatment uses a form of technical, active restoration to provide an opportunity for spontaneous succession to occur (Prach & Hobbs, 2008).

It is also possible that the improvement and similarity between topsoil-only and control conditions at the end of the study may be simply due to the exposure of buried individuals, either by erosion of some of the added topsoil, or growth of those buried individuals in order to resurface. While this suggests that encroachment from nearby vegetation may not have occurred after one growing season, the addition of topsoil did not cause any negative effects to the surrounding vegetation. Likely, the addition of topsoil will provide a source of nutrients, and further years of monitoring the experiment will reveal whether revegetation by means of encroaching vegetation becomes a successful restoration strategy in barrens habitat.

Both seeded-with-topsoil and topsoil-only treatments initially had lower cover than the control treatment. Initially, there would have been no germination in the seeded treatment and so both treatments would have resembled each other. Therefore, before the application of any treatment, there were some pre-existing vegetation on the trail, albeit less than that of the reference vegetation (See Chapter 2, model 1).

Topsoil addition improves treatment cover and quality

The addition of topsoil can be a costly and labour intensive process and it is important to know whether its use provides benefit to the restoration site. By the end of the study, it is clear that the treatments that included the addition of topsoil had improved cover by roughly 20% and improved survival of vegetation when compared to treatments that did not use it ((Figure 3.11 & Figure 3.16). Topsoil can directly improve abiotic soil conditions through the addition of nutrients, which are depleted on the Skyline trail (see Chapter 2, model 2) and can increase microbial activity (Rivera et. al., 2014) as well as create a loosely packed layer for easier root penetration of nearby vegetation and/or germinated seedlings (Bassett et. al., 2005; Tracy et. al. 2011).

When comparing transplanted treatments, there was a reduction in cover during the first week of the study when topsoil was used, but, interestingly, there was little difference between in the treatments in the final week of the study with a final median cover of 8.14 when transplanted-with-topsoil and 8.95 when directly transplanted (Figure 3.8). However initially, the cover was lower for transplanted-with-topsoil treatments (4.36 compared to 6.93 for directly transplanted), demonstrating that the greater improvement in cover for the transplanted-with-topsoil treatment (86% increase compared to 29% increase for directly transplanted, Figure 3.9) was driven by the topsoil addition, rather than the transplant addition. This is further supported by the evidence that the directly transplanted treatment improved at the same rate as the control, at 29% and 32%, respectively. Therefore, both transplanted treatments increased in cover with the use of transplants, and the transplanted-with-topsoil treatment likely experienced reduced initial cover through burial of pre-existing plants in the area. Then at the end of the study, the transplanted-with-topsoil treatment did improve more significantly than the improvement of the directly transplanted treatment due to the added topsoil. Given an extended study spanning multiple years, it is possible that the transplanted-with-topsoil treatment would continue to improve the site at a rate higher than that of a directly transplanted treatment and perhaps outperform the rate of improvement of the topsoil-only treatment as well.

In terms of the risk of promoting non-target vegetation, only one non-identifiable clover was found and presumed to have germinated from within the added topsoil. However, this species was scarce and only observed on a handful of occasions. While the introduction of foreign topsoil has the potential to recruit non-target or invasive species (Bulot et. al., 2016), this study at the Skyline trail is a very exposed headland where there is less chance of seed rain dispersing to the area. Here the main source of non-target or invasive plant material is likely to be found on the trail users themselves where seeds and twigs can get trapped in the soles of boots or on clothing (Dickens et. al., 2005). Thus, so long as trail users remain off the closed section, the risk for introducing non-target species is low.

In general, topsoil addition improves species survival and quality

D. spicata had consistently higher quality and survival in the topsoil treatment compared to the direct treatment over the course of the study, and while both treatments improved (Figure 3.15), neither treatment improved more so than the other (Figure 3.16). Therefore, *D. spicata* was able to persist and improve in both treatments but performed better in the topsoil treatment. It should be noted that the health index values did not account for growth in terms of size, which did appear to be larger in topsoil treatments than in direct treatments (personal observation). It is also important to note that the increase in survival count of *D. spicata* increased significantly at week 10 (Figure 3.12). During the study, all indices were recorded by the same observer, except during week 10 for *D. spicata* and *S. tridentata*. Thus, there is possible observer bias at week 10, likely due to difficulties differentiating between *D. spicata* and *D. flexuosa*.

While *V. angustifolium* performed better in the topsoil treatment when compared to direct transplanting (Figure 3.14), its quality and survival worsened over time in both treatments (Figure 3.15), and was the only species to do so. The improved quality and survival of *V. angustifolium* in the topsoil treatment, appears to have occurred between the final two rounds of data collection (Figure 3.17). This is evidence that *V. angustifolium* may have experienced 10 weeks of transplant shock, before showing any new growth. There is potential evidence that the individuals may have also improved in the directly transplanted treatment, which may have become clearer had the length of the study extended later into September and October.

Transplant shock is common and is defined as a reduced growth rate following transplanting greater than that if the plants were left undisturbed (Mullin, 1963). Transplant shock has been linked to moisture stress and low initial root volume (Haase & Rose, 1993). To add, even gentle perturbations of roots have been shown to slow root nitrate uptake in *Hordeum vulgare* for up to several hours (Bloom & Sukrapanna, 1990). Transplanting in the field involves significantly more agitation to plant root structure and is a main contributor to plant stress and reduces growth potential.

S. tridentata had consistently higher quality and survival in the topsoil treatment compared to the direct treatment and additionally, those in the topsoil treatment improved more so than in the direct treatment Figure 3.16. Thus *S. tridentata* was able use the improved quality of topsoil to grow at a faster rate than if topsoil had not been added. To add, *S. tridentata* produces rhizomes wherein the species can expand clonally from explorative root networks. This will likely contribute to better cover in future years when compared to the clumping, cespitose growth forms of *D. spicata* and *D. flexuosa*.

D. flexuosa did improve in the direct treatment but did not improve in the topsoil treatment. However, *D. flexuosa* was initially higher in quality and survival in the topsoil treatment (Figure 3.16). Therefore, while *D. flexuosa* was able to survive at similar rates in both treatments (Figure 3.12), its quality was consistently high in the topsoil treatment whereas its quality in the direct treatment started lower and improved to closely match that of the topsoil treatment by the end of the study. *D. flexuosa* also had the highest survival rates of all four species. This species can be associated with extreme environmental gradients (Oberndorfer & Lundholm, 2009) and so may have more able to tolerate the exposed trail conditions better than the others.

Directly transplanted species survived

Despite the benefits of using topsoil to aid in the recovery process, it is important to note that the directly transplanted treatment did improve over the course of the growing season. Thus, the transplanted species were able to establish themselves the in pre-existing soil and survive the first growing season. It is possible they may continue to establish and grow in future years, or they may die out. Further monitoring over the next several years is necessary to further understand the utility of directly transplanting as a form of remediation.

While the improvement of abiotic conditions via topsoil addition improves plant cover and quality, the improvement of biotic conditions by direct transplanting is also beneficial. This confirms that the seed germination and establishment phase of the plant life cycle is most affected by degraded trail conditions such as compaction (Bassett et. al., 2005). Thus, both biotic and abiotic amendments can improve the recovery process at the Skyline trail.

Interestingly, *D. flexuosa*, by the end of the study, had slightly better quality and survival in the directly transplanted treatment than when transplanted-with-topsoil addition (Figure 3.16). *D. flexuosa* is a calcifuge (Clarke, 1997) such that it is not suited to calcareous soils and thus is an acidophile. While the pH of trail conditions was higher than that of reference vegetation at Skyline (Chapter 2, model 2), the average pH of the trail was still low at 4.7. The commercial topsoil used had a pH of 6.0-6.5 and so *D. flexuosa* may have preferred the more acidic pre-existing soil, despite its lower nutrient content.

Potential recovery option in seeding with topsoil addition

The seeded treatment had the highest cover by the end of the study. Seeds of the seeded plots were able to germinate, and seedlings were able to establish over the course of the study. Most germinations were from *D. spicata* and *D. flexuosa* grasses. *S. tridentata* germinations were observed on a handful of occasions and no *V. angustifolium* or *C. canadensis* germinations were observed. The seed density of both grass species was vastly greater than of the other three species and may have contributed to their germination success. *C. canadensis* germinated poorly in greenhouse conditions and so the outcome in the field reflects germination difficulties, especially at low densities. It is possible that the seeds will remain on site and further stratification requirements, particularly for *V. angustifolium*, may promote germination during appropriate conditions in subsequent years.

Therefore, improvement of abiotic conditions via topsoil addition allowed for the successful germination and establishment of the grass species. Again, a seeding-only treatment without topsoil and erosion control mat was not incorporated into the study design as it has been previously determined to be ineffective (Ebersole et. al., 2004). Additionally, the pre-treatment abiotic conditions were not conducive for the formation of a persistent seed bank on the trail (Chapter 2, model 3) and its amelioration and addition of seed created conditions suitable for seed germination and seedling establishment. The result is established plant cover that can produce its own seed and the potential for seed bank formation. The same can be said for both transplanted treatments.

Percent cover and frequency cover methods are related

Overall, when comparing frequency and percent cover methods (Figure 3.18), they tend to be generally linearly related for smaller species, however the relationship appears to become non-linear with more abundant, larger species. It is likely that, for all species, once the frequency caps at 25 out of 25, there is still a lot of room for variation in percent cover values (between roughly 50-100%). Thus, this relationship is only noticeable for more abundant species, like *V. angustifolium* and *A. uva-ursi*, that have enough plots to allow for variation in cover to occur. To add, the full range of cover values is not represented for most of the species considered, which may also explain the observed non-linearity.

A single blade of a grass like *D. spicata* and *D. flexuosa* garners only a fraction of space where the percent cover is observed but can count as present for 1 in a subplot where frequency cover is observed. Alternatively, a mat of *A. uva-ursi* can count as the entire 4 % of a subplot with percent cover (100 % ÷ 25 subplots = 4 % per subplot) but is still only present for 1 with frequency cover. Therefore, the relationship between methods for less abundant species becomes quite strongly linear and there is greater deviance between methods for more abundant species, but only at higher abundance values.

This relationship was considered as the experiment is intended to continue to be monitored in subsequent years, likely by multiple different observers over time. There having been issues regarding observer bias when the percent cover method is used (Bergstedt et. al., 2008). There are differences from observer to observer which could influence the interpretation of future collected data. In this sense, the frequency cover method remains a viable option for future monitoring for this experiment as repeatability across observers is more precise than using percent cover, both methods are relatively strongly correlated, and finally it is more expedient to conduct.

3.5 Literature cited

Aavik, T., Edwards, P. J., Holderegger, R., Graf, R., & Billeter, R. (2012). Genetic consequences of using seed mixtures in restoration: A case study of a wetland plant Lychnis flos-cuculi. *Biological Conservation*, *145*, 195–204. https://doi.org/10.1016/j.biocon.2011.11.004

Anastasiou, C. J., & Brooks, J. R. (2006). Effects of soil PH, redox potential, and elevation on survival of Spartina patens planted at a west Central Florida salt marsh restoration site. *Wetlands*, *23*(4), 845–859. https://doi.org/10.1672/0277-5212(2003)023[0845:eosprp]2.0.co;2

Aradottir, A. L. (2012). Turf transplants for restoration of alpine vegetation: Does size matter? *Journal of Applied Ecology*, *49*(2), 439–446. https://doi.org/10.1111/j.1365-2664.2012.02123.x

Bassett, I. E., Simcock, R. C., & Mitchell, N. D. (2005). Consequences of soil compaction for seedling establishment: Implications for natural regeneration and restoration. *Austral Ecology*, *30*(8), 827–833. https://doi.org/10.1111/j.1442-9993.2005.01525.x

Berg, L., Vergeer, P., & Roelofs, J. (2003). Heathland restoration in the Netherlands: effects of turf cutting depth on germination of Arnia montana. *Applied Vegetation Science, 6*(2), 117–124.

Bergstedt, J., Fridman, J., Odell, G., & Westerberg, L. (2008). Observer Bias and Random Variation in Vegetation Monitoring Data. *Journal of Vegetation Science, 19*(5), 633–644. https://doi.org/10.3170/2008-8-18423

Bloom, A. J., & Sukrapanna, S. S. (1990). Effects of Exposure to Ammonium and Transplant Shock upon the Induction of Nitrate. *Plant Physiology*, *94*(1), 85–90.

Bucharova, A., Michalski, S., Hermann, J.-M., Heveling, K., Durka, W., Hölzel, N., … Bossdorf, O. (2017). Genetic differentiation and regional adaptation among seed origins used for grassland restoration: lessons from a multispecies transplant experiment. *Journal of Applied Ecology*, *54*(1), 127–136. https://doi.org/10.1111/1365-2664.12645

Bullock, J. M., Aronson, J., Newton, A. C., Pywell, R. F., & Rey-Benayas, J. M. (2011). Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Trends in Ecology and Evolution*, *26*(10), 541–549. https://doi.org/10.1016/j.tree.2011.06.011

Bulot, A., Potard, K., Bureau, F., Bérard, A., & Dutoit, T. (2016). Ecological restoration by soil transfer: impacts on restored soil profiles and topsoil functions. *Restoration Ecology*, *25*(3), 354–366. https://doi.org/10.1111/rec.12424

Burke, A. (2008). The effect of topsoil treatment on the recovery of rocky plain and outcrop plant communities in Namibia. *Journal of Arid Environments, 72*(8), 1531–1536. https://doi.org/10.1016/j.jaridenv.2008.03.005

Cole, D. N. (2004). Impacts of hiking and camping on soils and vegetation: a review. In R. Buckley. *Environmental Impacts of Ecotourism*. (Series no. 2). Massachusetts, USA: CABI Publishing.

Cole, D. N. (2007). Seedling Establishment and Survival on Restored Campsites in Subalpine Forest Seedling Establishment and Survival on Restored Campsites in Subalpine Forest. *Restoration Ecology*, *15*(3), 430–439.

Cullina, W. (2002) Native trees, shrubs and vines: A guide to using, growing, and propagating North American woody plants. New York, NY: The New England Wild Flower Society

Dickens, S. J., Gerhardt, F. , & Collinge, S. K. (2005). Recreational portage trails as corridors facilitating non-native plant invasions of the boundary waters canoe area wilderness (U.S.A.). *Conservation Biology*, *19*(5), 1653–1657.

Ebersole, J. J., Bay, R. F., & Conlin, D. B. (2004). Alpine vegetation restoration of social trails on Colorado’s 14,000-foot peaks. *Proceedings of 16th International Conference, Society for Ecological Restoration*, 101–122.

Ehrenfeld, J. G. (2000). Defining the limits of restoration: The need for realistic goals. *Restoration Ecology*, *8*(1), 2–9. https://doi.org/10.1046/j.1526-100X.2000.80002.x

Ellis, K. (2017). Evaluation of the Erosion Reduction Potential, Stability, and Resilience of Living Shorelines in Nova Scotia. (Master’s thesis). Saint Mary’s University, Halifax, Nova Scotia.

Eviner, V. T., & Hawkes, C. V. (2008). Embracing variability in the application of plant-soil interactions to the restoration of communities and ecosystems. *Restoration Ecology*, *16*(4), 713–729. https://doi.org/10.1111/j.1526-100X.2008.00482.x

Fagúndez, J. (2017). The interplay of nutrients and tree invasion shapes the plant community in barren habitats. *Applied Vegetation Science*, *20*(1), 5–6. https://doi.org/10.1111/avsc.12294

Haase, D. & Rose, R. (1993). Transplant Shock in Stored and Unstored 2 + 0 Douglas-Fir Seedlings of Varying Root Volumes to. *Forest Science*, *39*, 275–294.

Hagen, D., Hansen, T.-I., Graae, B. J., & Rydgren, K. (2014). To seed or not to seed in alpine restoration: introduced grass species outcompete rather than facilitate native species. *Ecological Engineering*, *64*, 255–261. https://doi.org/10.1016/j.ecoleng.2013.12.030

Hamilton, N. R. S. (2001). Is Local Provenance Important in Habitat Creation? A Reply. *Journal of Applied Ecology*, *38*(6), 1374–1376.

Helsper, H. P. G., Glenn-Lewin, D., & Werger, M. J. A. (1983). Early regeneration of barren under various fertilization treatments. *Oecologia (Berlin)*, *58*, 208–214.

Herget, M. E., Hufford, K. M., Mummey, D. L., & Shreading, L. N. (2015). Consequences of Seed Origin and Biological Invasion for Early Establishment in Restoration of a North American Grass Species. *PLoS ONE*. https://doi.org/10.1371/journal.pone.0119889

Holmes, P. M. (2001). Shrubland restoration following woody alien invasion and mining: Effects of topsoil depth, seed source, and fertilizer addition. *Restoration Ecology*, *9*(1), 71–84. https://doi.org/10.1046/j.1526-100X.2001.009001071.x

Jansen, A. J. M., Fresco, L. F. M., Grootjans, A. P., & Jalink, M. H. (2004). Effects of restoration measures on plant communities of wet barren ecosystems. *Applied Vegetation Science*, *7*(2), 243–252. https://doi.org/10.1111/j.1654-109X.2004.tb00616.x

Jones, T. A. (2013). When local isn't best. *Evolutionary Applications*, *6*(7), 1109–1118. https://doi.org/10.1111/eva.12090

Kirmer, A., Baasch, A., & Tischew, S. (2012). Sowing of low and high diversity seed mixtures in ecological restoration of surface mined-land. *Applied Vegetation Science*, *15*(2), 198–207.

Le Roy, M., Sawtschuk, J., Bioret, F., & Gallet, S. (2018). Toward a social-ecological approach to ecological restoration: A look back at three decades of maritime clifftop restoration. *Restoration Ecology*, 1–11. https://doi.org/10.1111/rec.12849

Liddle, M. J. (1997). *Recreation Ecology*. London; New York : Chapman & Hall.

Lynn, N. A., & Brown, R. D. (2003). Effects of recreational use impacts on hiking experiences in natural areas. *Landscape and Urban Planning*, *64*, 77–87.

Matesanz, S., & Valladares, F. (2007). Improving revegetation of gypsum slopes is not a simple matter of adding native species: Insights from a multispecies experiment. *Ecological Engineering*, *30*(1), 67–77. https://doi.org/10.1016/j.ecoleng.2007.01.005

Meyer-Grünefeldt, M., Calvo, L., Marcos, E., Von Oheimb, G., & Härdtle, W. (2015). Impacts of drought and nitrogen addition on Calluna barrens differ with plant life-history stage. *Journal of Ecology*, *103*(5), 1141–1152. https://doi.org/10.1111/1365-2745.12446

Miller, B. P., Sinclair, E. A., Menz, M. H. M., Elliott, C. P., Bunn, E., Commander, L. E., … Stevens, J. C. (2017). A framework for the practical science necessary to restore sustainable, resilient, and biodiverse ecosystems. *Restoration Ecology*, *25*(4), 605–617. https://doi.org/10.1111/rec.12475

Moles, A. T., & Westoby, M. (2014). Across the whole life cycle: Seed size and plant strategy. *Oikos*, *113*, 91–105.

Mullin, R. E. (1963). Planting check in spruce. *The Forestry Chronicle*, 252–259.

Oberndorfer, E. C., & Lundholm, J. T. (2009). Species richness, abundance, rarity and environmental gradients in coastal barren vegetation. *Biodiversity and Conservation*, *18*, 1523–1553. https://doi.org/10.1007/s10531-008-9539-5

Ohsowski, B. M., Klironomos, J. N., Dunfield, K. E., & Hart, M. M. (2012). The potential of soil amendments for restoring severely disturbed grasslands. *Applied Soil Ecology*, *60*, 77–83. https://doi.org/10.1016/j.apsoil.2012.02.006

Pons, T. L. (1989). Dormancy, germination and mortality of seeds in heathland and inland sand dunes. *Acta Botanica Neerlanda*, *38*(3), 327–335.

Power, S. A., Green, E. R., Barker, C. G., Bell, J. N. B., & Ashmore, M. R. (2006). Ecosystem recovery: Barren response to a reduction in nitrogen deposition. *Global Change Biology*, *12*(7), 1241–1252. https://doi.org/10.1111/j.1365-2486.2006.01161.x

Prach, K., Durigan, G., Fennessy, S., Overbeck, G. E., Torezan, J. M., & Murphy, S. D. (2019). A primer on choosing goals and indicators to evaluate ecological restoration success. *Restoration Ecology*, *27*(5), 917–923. https://doi.org/10.1111/rec.13011

Prach, K., & Hobbs, R. J. (2008). Spontaneous Succession versus Technical Reclamation in the Restoration of Disturbed Sites. *Restoration Ecology*, *16*(3), 363–366. https://doi.org/10.1111/j.1526-100X.2008.00412.x

Pywell, R. F., Webb, N. R., & Putwain, P. D. (1995). A Comparison of Techniques for Restoring Barren on Abandoned Farmland. *Journal of Applied Ecology*, *32*(32), 400–411.

Rivera, D. E., Mejías, V., Jáuregui, B. M., Costa-Tenorio, M., & Ló Pez-Archilla, A. I. (2014). Spreading Topsoil Encourages Ecological Restoration on Embankments: Soil Fertility, Microbial Activity and Vegetation Cover. *PLoS ONE*, *9*(7), 101413. https://doi.org/10.1371/journal.pone.0101413

Roem, W. J., Klees, H., & Berendse, F. (2002). Effects of nutrient addition and acidification on plant species diversity and seed germination in barren. *Journal of Applied Ecology*, *39*(6), 937–948. https://doi.org/10.1046/j.1365-2664.2002.00768.x

Rydgren, K., Hagen, D., Rosef, L., Pedersen, B., & Aradottir, A. L. (2017). Designing seed mixtures for restoration on alpine soils: who should your neighbours be? *Applied Vegetation Science*, *20*(3), 317–326. https://doi.org/10.1111/avsc.12308

Sawtschuk, J., Bioret, F., & Gallet, S. (2010). Spontaneous Succession as a Restoration Tool for Maritime Cliff-top Vegetation in Brittany, France. *Restoration Ecology*, *18*(SUPPL. 2), 273–283. https://doi.org/10.1111/j.1526-100X.2010.00653.x

Scherrer, P., & Pickering, C. M. (2006). Recovery of Alpine Herbfield on a Closed Walking Track in the Kosciuszko Alpine Zone, Australia. *Arctic, Antarctic, and Alpine Research*, *38*(2), 239–248.

Timothy. D. J. (2015). *Tourism and Trails: Cultural, Ecological and Management Issues.* Bristol, UK: Channel View Publications.

Tracy, S. R., Black, C. R., Roberts, J. A., & Mooney, S. J. (2011). Soil compaction: a review of past and present techniques for investigating effects on root growth. *Journal of the Science of Food and Agriculture*, *91*, 1528–1537. https://doi.org/10.1002/jsfa.4424

Vander Mijnsbrugge, K., Bischoff, A., & Smith, B. (2010). A question of origin: Where and how to collect seed for ecological restoration. *Basic and Applied Ecology*, *11*, 300–311. https://doi.org/10.1016/j.baae.2009.09.002

Vergeer, P., Van Den Berg, L. J. L., Baar, J., Ouborg, N. J., & Roelofs, J. G. M. (2006). The effect of turf cutting on plant and arbuscular mycorrhizal spore recolonisation: Implications for barren restoration. *Biological Conservation*, *129*, 226–235. https://doi.org/10.1016/j.biocon.2005.10.038

Wainwright, C. E., Staples, T. L., Charles, L. S., Flanagan, T. C., Lai, H. R., Loy, X., … Mayfield, M. M. (2018). Links between community ecology theory and ecological restoration are on the rise. *Journal of Applied Ecology*, *55*(2), 570–581. https://doi.org/10.1111/1365-2664.12975

Wilkinson, D. M. (2001). Is Local Provenance Important in Habitat Creation? *Journal of Applied Ecology, 38,* 1371-1373.

[redacted] Conclusions and moving forward

Overall, a variety of biotic and abiotic factors contribute to the low natural recovery potential at the closed sections of trail at both Skyline and Mica Hill locations. Stresses at both locations have led to elevated substrate compaction, surface temperatures, moisture content, and pH as well as lowered nutrient content. At Mica Hill, this has led to a shift in the vegetation community whereby the dominant vegetation on the trail does not match the rest of the ecosystem. At Skyline, trampling intensity has led to the disappearance of almost all vegetation on the trail and subsequently removed any form of seed bank.

The implementation of restoration treatments on the Skyline trail demonstrates that restoration of abiotic environmental components can improve vegetation cover and quality, in the year treatments were installed. Improvement of abiotic conditions were in the form of soil amendments via commercial topsoil addition and biodegradable erosion control matting. Thus, nutrients were reintroduced into the system, water-holding capacity was increased, compaction was reduced, and the erosion control mat allowed for some buffer protection against high winds and heavy rains. The addition of topsoil and an erosion control mat was able to improve plant cover and quality when compared to a passive control treatment. Soil ecology plays an integral role in restoration ecology and provides many feedbacks between aboveground and belowground processes (Heneghan et. al., 2008). Abiotic restoration methods, particularly related to soil ecology, can be an effective restoration tool in a headland barrens environment.

The restoration treatments also demonstrate that the restoration of biotic environmental components can improve vegetation cover and quality. Improvement of biotic conditions were in the form of native, genetically similar transplants and seed. Vegetation cover was immediately increased, and a version of seed bank was restored. When transplants were directly added to the trail without any topsoil amendments, many were able to survive and improved when compared to a passive control treatment. Therefore, the reason natural recovery fails on the trail at Skyline is not because mature plants cannot survive the harsh trail conditions, but rather natural recovery fails because seeds are not staying on the trail and harsh trail conditions do not allow for successful seed germination and seedling establishment. Biotic restoration methods that bypass the germination phase can be an effective restoration tool in a headland barrens environment.

While abiotic and biotic restoration can aid in restoration separately, they are more effective when used simultaneously. The addition of transplants or seeds into topsoil with an erosion control mat both had greatly improved vegetation cover and quality in the first year. There is potential for improved growth in these treatments over the next several years. The use of seeding in combination with topsoil amendments may be a very successful treatment in subsequent years as it improves soil quality and reintroduces vegetation cover without the growing requirements for *ex situ* transplants. Thus, improvement of abiotic conditions via topsoil amendments can allow for successful seed germination and seedling establishment on the trails at Skyline.

With sufficient knowledge of an ecosystem and restoration methodology, restoration outcomes are often successful (Murcia & Aronson, 2014). In such a way, insight gained from collected data and analyses from Chapter 2, directly contribute to the successful implementation of treatments at Skyline as analysed in Chapter 3. The increased compaction and lowered nutrient content were addressed through topsoil addition with biodegradable erosion controls. The added topsoil was uncompacted and high in nutrients. The erosion control mats stabilized the added topsoil which lacked a stabilizing root network. The absence of a seed bank and reduced vascular plant cover at Skyline was addressed through seed and transplant addition. The stabilized topsoil allowed for improved growth of seedlings and transplants which, in turn, will be able to create that root network which will theoretically continue stabilize and utilize the topsoil as the erosion control mats begin to decompose. This particular study was able to take knowledge gained from the ecosystem in addition to theoretical knowledge and examples from past studies in similar environments and apply it towards a restoration project in a logical and practical manner.

However, there is often a gap between science and applied restoration where there can be conflicting interests between restoration practitioners and scientific research (Clark et. al., 2019; Miller et. al., 2017). This gap is the result of the, often, lengthy amount of time needed to properly develop and test theories while many restoration projects require immediate action and, subsequently, there is insufficient empirical data to make informed restoration and conservation decisions (Cadotte et. al., 2017). As an example, most restoration outcomes are described within a time span of less than 5 years (Le Roy et. al. 2018) which can make it difficult to determine the long-term effects of restoration practices.

In an era where the effects of climate change are becoming unmistakable, there are increasing calls to protect, remediate, and restore the environment (Prober et. al., 2019). Particularly, environments unique in niche, time, and space are more vulnerable to the changing climate and thus require distinct management strategies (Kling et. al., 2020). While this study does not directly address climate change, its research implications may provide support to future restoration and rehabilitation projects in an uncommon habitat, undertaken as a result of climate change. To add, as this study will persist for the next decade or more, research outcomes may be affected.

This study provides supporting evidence for factors contributing to trail degradation, with specific details for barrens habitat. It also provides the basis of a long-term restoration study aimed at identifying potential restoration strategies in the barrens of Nova Scotia. Long-term monitoring is crucial to measuring the success of the implemented treatments (Prach et. al., 2019). Empirical evidence gathered from this study as it moves forward can inform future restoration programs in the province and, in the face of climate change, can potentially provide guidance on restoration practices in stressful environmental conditions.

4.1 Literature cited

Cadotte, M. W., Barlow, J., Nuñez, M. A., Pettorelli, N., & Stephens, P. A. (2017). Solving environmental problems in the Anthropocene: the need to bring novel theoretical advances into the applied ecology fold. *Journal of Applied Ecology*, *54*(1), 1–6. https://doi.org/10.1111/1365-2664.12855

Clark, L. B., Henry, A. L., Lave, R., Sayre, N. F., González, E., & Sher, A. A. (2019). Successful information exchange between restoration science and practice. *Restoration Ecology*, *27*(6), 1241–1250. https://doi.org/10.1111/rec.12979

Heneghan, L., Miller, S. P., Baer, S., Callaham, M. A., Montgomery, J., Pavao-Zuckerman, M., … Richardson, S. (2008). Integrating Soil Ecological Knowledge into Restoration Management. *Restoration Ecology*, *16*(4), 608–617. https://doi.org/10.1111/j.1526-100X.2008.00477.x

Kling, M. M., Auer, S. L., Comer, P. J., Ackerly, D. D., & Hamilton, H. (2020). Multiple axes of ecological vulnerability to climate change. *Global Change Biology*, *00*, 1–16. https://doi.org/10.1111/gcb.15008

Le Roy, M., Sawtschuk, J., Bioret, F., & Gallet, S. (2018). Toward a social-ecological approach to ecological restoration: A look back at three decades of maritime clifftop restoration. *Restoration Ecology*, 1–11. https://doi.org/10.1111/rec.12849

Miller, B. P., Sinclair, E. A., Menz, M. H. M., Elliott, C. P., Bunn, E., Commander, L. E., … Stevens, J. C. (2017). A framework for the practical science necessary to restore sustainable, resilient, and biodiverse ecosystems. *Restoration Ecology*, *25*(4), 605–617. https://doi.org/10.1111/rec.12475

Murcia, C., & Aronson, J. (2014). Intelligent Tinkering in Ecological Restoration. *Restoration Ecology*, *22*(3), 279–283. https://doi.org/10.1111/rec.12100

Prach, K., Durigan, G., Fennessy, S., Overbeck, G. E., Torezan, J. M., & Murphy, S. D. (2019). A primer on choosing goals and indicators to evaluate ecological restoration success. *Restoration Ecology*, *27*(5), 917–923. https://doi.org/10.1111/rec.13011

Prober, S. M., Doerr, V. A. J., Broadhurst, L. M., Williams, K. J., & Dickson, F. (2019). Shifting the conservation paradigm: a synthesis of options for renovating nature under climate change. *Ecological Monographs*, *89*(1), 1–23. https://doi.org/10.1002/ecm.1333

Appendix

Table 0.1. Total estimated costs of items used in the setup of the restoration treatments including, cost, preparation time, and application time. While preparation and application estimates are in single person hours, most of the work was conducted in groups or with a partner. Preparation of TerraFix coir mats involved cutting rolls to appropriate sizes, transplant preparation involved germination and growth in a greenhouse, and seed preparation involved collection in the field, and seed cleaning/extraction.

Item	**Amount**	**Estimated total cost**	**Estimated preparation time (for one person)**	**Estimated application time (for one person)**
Scotts® Pro Blend Top Soil	400 bags	2500.00 $	NA	300 hours
APB5 Anchoring Bulk Pins, Dewitt	1500 pins	360.00 $	NA	Done with mat application
TerraFix Coir Mat 400	4 rolls	900.00 $	4 hours	4 hours
Transplants	1180 individuals	NA	7 months	32 hours
Seeds	400 g	NA	80 hours	1 hour

Table 0.2. Model selection output for seed bank emergence. AIC values were calculated using gls linear models with the lme_4 package. WAIC values were calculated using log likelihood values using the loo package. WAIC SE is the standard error for each WAIC value.

Model	**Df**	**AIC**	**ΔAIC**	**WAIC**	**WAIC SE**
ID + Block	8	71.7	0.0	977.8	251.7
ID	7	74.7	3.0	1098.6	449.4
Intercept	2	85.2	13.5	1766.2	606.9

Table 0.3. 95% HDI posterior distribution outputs for comparisons of seed bank emergence study. Values are the log scale.

Comparison	**2.5% HDI**	**Mean**	**97.5% HDI**
Positive Control – Negative Control	1.6076	1.8254	2.0428
Mica Hill Ref – Mica Hill Trail	-1.3064	-1.1197	-0.94214
Skyline Ref – Skyline Trail	0.26011	0.51108	0.76722
Mica Hill Ref – Negative Control	0.24828	0.49984	0.76229
Mica Hill Trail – Negative Control	1.3985	1.6196	1.8426
Skyline Ref – Negative Control	0.27993	0.53215	0.78846
Skyline Tail – Negative Control	-0.26464	0.021069	0.29969

Table 0.4. Model selection output for transplant health index. AIC values were calculated using polr linear models with the MASS package. WAIC values were calculated using log likelihood values using the loo package. WAIC SE is the standard error for each WAIC value.

Model	**AIC**	**ΔAIC**	**WAIC**	**WAIC SE**
Treat + Species + Week + Treat*Species + Treat*Week + Species*Week + Treat*Species*Week + Site	Does not converge	NA	15838.3	110.0
Treat + Species + Week + Treat*Species + Treat*Week + Species*Week + Treat*Species*Week	16004.2	0.0	16006.3	108.2
Treat + Species + Week + Treat*Species + Treat*Week + Species*Week	16013.5	9.3	16014.6	107.6
Treat + Species + Week + Treat*Week + Species*Week	16017.2	13.0	16018.5	107.8
Treat + Species + Week + Treat*Week	16496.8	492.6	16497.0	97.7
Treat + Species + Week	16492.2	488.0	16492.4	97.4
Treat + Week	17356.6	1352.4	1737.2	79.0
Week	17410.7	1406.5	17411.1	76.8
Intercept	17479.9	1475.7	17479.8	73.5

Table 0.5. Raw data from substrate nutrient analysis for Skyline samples.

Distance down trail (m)	Type	Nitrogen (%)	pH (pH Units)	Buffer pH (pH Units)	Organic Matter (%)	P2O5 (kg/ha)	K2O (kg/ha)	Calcium (kg/ha)	Iron (ppm)	Manganese (ppm)	Zinc (ppm)	CEC (meq/100 g)
20	**Trail**	0.2	4.51	7.32	6.5	29	209	1062	321	46	8.1	10
20	**Control**	0.96	4.07	6.5	31	46	378	1774	162	45	10.35	19.3
60	**Trail**	0.19	4.43	7.26	4.8	41	145	180	325	7	1.58	7
60	**Control**	0.37	4.12	6.92	11.3	65	160	675	335	21	4.36	11.7
140	**Trail**	0.3	4.9	7.35	8.8	58	247	1856	322	80	12.85	13
140	**Control**	1.23	4.15	6.91	38.4	124	410	1132	272	86	9.44	13.8
180	**Trail**	0.15	4.63	7.05	5.4	9	139	240	243	10	0.77	8.9
180	**Control**	0.26	4.41	7.34	7	21	232	991	306	41	6.89	10.1
220	**Trail**	0.14	4.58	6.88	5.5	10	77	33	133	2	0.2	9.2
220	**Control**	0.56	4.13	6.83	17.8	42	191	1164	357	42	8.37	13.9
260	**Trail**	0.18	4.51	7.09	5.7	7	210	716	343	14	4.32	10.6
260	**Control**	0.89	4.12	6.51	24.3	68	311	1520	332	35	11.34	18.3
300	**Trail**	0.57	5.23	7.01	15.2	23	286	992	352	31	5.73	12.2
300	**Control**	0.92	4.25	6.78	23.4	88	337	1332	328	70	6.85	15.7
330	**Trail**	0.3	4.39	6.94	7.6	23	223	239	397	20	1.57	10
330	**Control**	0.64	4.75	7.14	18.1	87	323	2205	315	174	16.77	15.2
350	**Trail**	0.17	4.54	6.7	7.1	14	50	62	151	3	0.26	10.7
350	**Control**	0.84	4.66	6.8	18.8	77	232	929	267	54	8.46	13.8
410	**Trail**	0.94	5.07	7.16	15.8	63	319	1775	217	103	21.35	15.6
410	**Control**	0.88	4.67	7.19	20.3	149	340	1743	271	147	11.26	14.5

Table 0.5. Continued.

Distance down trail (m)	Type	Magnesium (kg/ha)	Sodium (kg/ha)	Sulfur (kg/ha)	Aluminum (ppm)	Boron (ppm)	Copper (ppm)	Base sat. K (%)	Base sat. Ca (%)	Base sat. Mg (%)	Base sat. Na (%)
20	**Trail**	361	68	17	1008	0.5	0.33	2.2	26.6	15.1	1.5
20	**Control**	542	104	17	293	0.5	0.33	2.1	23	11.7	1.2
60	**Trail**	91	46	32	1430	0.5	0.18	2.2	6.4	5.4	1.4
60	**Control**	254	57	20	829	0.5	0.15	1.5	14.4	9.1	1.1
140	**Trail**	659	56	9	748	0.5	0.24	2	35.8	21.2	0.9
140	**Control**	400	52	27	243	0.5	0.23	3.2	20.6	12.1	0.8
180	**Trail**	110	63	44	2054	0.5	0.11	1.6	6.7	5.1	1.5
180	**Control**	458	81	16	953	0.5	0.17	2.4	24.6	18.9	1.8
220	**Trail**	9	30	86	2717	0.5	0.27	0.9	0.9	0.4	0.7
220	**Control**	316	68	22	791	0.5	0.24	1.5	20.9	9.5	1.1
260	**Trail**	269	79	21	1434	0.5	0.25	2.1	16.9	10.6	1.6
260	**Control**	500	80	22	601	0.5	0.38	1.8	20.8	11.4	0.9
300	**Trail**	326	70	16	1040	0.5	0.66	2.5	20.3	11.1	1.2
300	**Control**	496	67	23	597	0.5	0.55	2.3	21.3	13.2	0.9
330	**Trail**	149	53	41	1559	0.5	1.24	2.4	5.9	6.2	1.2
330	**Control**	553	97	16	542	0.5	0.55	2.2	36.1	15.1	1.4
350	**Trail**	15	32	83	3143	0.5	1.02	0.4	1.4	0.6	0.7
350	**Control**	363	75	17	729	0.5	0.69	1.8	16.8	10.9	1.2
410	**Trail**	919	132	15	665	0.76	0.89	2.2	28.4	24.5	1.8
410	**Control**	736	119	23	640	0.57	0.65	2.5	30	21.1	1.8

Table 0.5. Continued

Distance down trail (m)	Type	Base sat. H (%)	LR CaCO3 (t/ha to pH 6.5)
20	**Trail**	54.6	10
20	**Control**	62.1	24
60	**Trail**	84.5	12
60	**Control**	74	18
140	**Trail**	40.1	9
140	**Control**	63.4	18
180	**Trail**	85	14
180	**Control**	52.3	11
220	**Trail**	97.1	17
220	**Control**	67.2	19
260	**Trail**	68.8	14
260	**Control**	65.1	24
300	**Trail**	64.9	12
300	**Control**	62.3	20
330	**Trail**	84.4	17
330	**Control**	45.1	12
350	**Trail**	96.9	20
350	**Control**	69.3	18
410	**Trail**	43.1	11
410	**Control**	44.6	12

Table 0.6. Raw data from substrate nutrient analysis from Mica Hill samples.

Section	Distance down trail (m)	Type	Nitrogen (%)	pH (pH Units)	Buffer pH (pH Units)	Organic Matter (%)	P2O5 (kg/ha)	K2O (kg/ha)	Calcium (kg/ha)	Iron (ppm)	Manganese (ppm)	Zinc (ppm)	CEC (meq/100 g)
A	50	Trail	0.01	4.32	8	1	28	50.00	25	69	0.00	0.20	0.3
A	50	Control	0.86	3.86	6.69	43.4	82	280	343	60	7	2.78	12.5
B	40	Trail	0.03	4.26	7.83	1.8	74	50.00	29	182	1	0.22	1.6
B	40	Control	0.5	3.72	6.81	47.8	41	313	495	22	26	1.79	12.5
C	60	Trail	0.03	4.73	7.86	1.8	22	50.00	170	347	69	0.24	1.7
C	60	Control	0.96	5.12	7	22	39	182	3327	268	142	1.91	18
C	110	Trail	0.01	4.3	7.99	1.2	28	50.00	26	101	2	0.20	0.2
C	110	Control	0.93	3.69	6.61	55.8	99	295	633	41	23	6.05	14.5
D	30	Trail	0.49	4.25	7.54	10.2	74	78	282	322	5	1.59	5
D	30	Control	2.33	3.85	6.51	49.1	107	106	376	174	1	1.2	13.7
D	70 (side trail)	Trail	0.22	3.79	7.16	10.3	33	98	240	84	1	2.16	8.2
D	70 (side trail)	Control	0.82	3.71	6.45	53.6	40	272	960	34	1	5.79	17.6
D	130	Trail	0.02	4.27	7.86	2.7	35	50.00	49	186	1	0.51	1.4
D	130	Control	0.88	3.71	6.34	43.9	11	134	593	54	2	2.54	16.5
D	230	Trail	0.01	4.12	7.92	1.5	44	50.00	14	87	0.00	0.20	0.8
D	230	Control	0.48	3.69	7.02	22.2	31	180	258	71	1	1.76	9.5
D	330	Trail	0.04	3.97	7.79	2.1	29	50.00	44	113	1	0.68	2
D	330	Control	0.12	3.84	7.5	9.2	25	166	431	15	16	3.85	6.1
D	430	Trail	0.05	4.56	7.71	2.1	66	50.00	21	267	0.00	0.25	2.5
D	430	Control	0.62	3.73	6.88	23.4	21	192	125	53	1	1.12	10.4

Table 0.6. Continued

Section	Distance down trail (m)	Type	Magnesium (kg/ha)	Sodium (kg/ha)	Sulfur (kg/ha)	Aluminum (ppm)	Boron (ppm)	Copper (ppm)	Base sat. K (%)	Base sat. Ca (%)	Base sat. Mg (%)	Base sat. Na (%)
A	50	Trail	13	19	6	125	0.50	0.18	13.1	23.2	19.8	14.7
A	50	Control	176	42	14	493.00	0.50	0.28	2.4	6.9	5.9	0.7
B	40	Trail	18	18	10	517	0.50	0.10	1.9	4.6	4.7	2.5
B	40	Control	302	73	17	73	0.50	0.32	2.7	9.9	10.1	1.3
C	60	Trail	31	21	9	568	0.50	0.10	1.6	24.3	7.5	2.6
C	60	Control	302	91	11	1023	0.50	0.17	1.1	46.3	7	1.1
C	110	Trail	15	16.00	4	124	0.50	0.10	4.2	27.5	26.3	8.9
C	110	Control	314	64	12	221	0.50	0.18	2.2	10.9	9	1
D	30	Trail	89	53	13	537	0.50	0.44	1.7	14.2	7.5	2.3
D	30	Control	137	78	31	742	0.50	0.17	0.8	6.9	4.2	1.2
D	70 (side trail)	Trail	175	36	7	371	0.50	0.10	1.3	7.3	8.8	0.9
D	70 (side trail)	Control	554	105	8	75	0.50	0.3	1.6	13.6	13.1	1.3
D	130	Trail	31	16.00	6	296	0.50	0.69	1.8	8.5	9.2	2.2
D	130	Control	361	56	6	254	0.50	0.15	0.9	9	9.1	0.7
D	230	Trail	12	16.00	7	318	0.50	0.10	2.7	4.4	6.5	3.7
D	230	Control	175	54	13	440	0.50	0.10	2	6.8	7.7	1.2
D	330	Trail	35	32	9	355	0.50	0.10	1.7	5.4	7.2	3.4
D	330	Control	192	41	10	60	0.50	0.17	2.9	17.6	13	1.5
D	430	Trail	16	16.00	32	1279	0.50	0.21	1.3	2.1	2.6	1.3
D	430	Control	186	53	9	435	0.50	0.10	2	3	7.5	1.1

Table 0.6. Continued

Section	Distance down trail (m)	Type	Base sat. H (%)	LR CaCO3 (t/ha to pH 6.5)
A	50	Trail	29.2	
A	50	Control	84.1	21
B	40	Trail	86.3	3
B	40	Control	76.1	19
C	60	Trail	64.1	2
C	60	Control	44.5	13
C	110	Trail	33.2	
C	110	Control	76.9	23
D	30	Trail	74.3	7
D	30	Control	86.9	24
D	70 (side trail)	Trail	81.6	14
D	70 (side trail)	Control	70.3	25
D	130	Trail	78.3	2
D	130	Control	80.3	27
D	230	Trail	82.7	1
D	230	Control	82.3	16
D	330	Trail	82.2	3
D	330	Control	65.1	8
D	430	Trail	92.6	4
D	430	Control	86.4	18