Rapid Understorey Restoration of a Red Pine Plantation in Algonquin Provincial Park

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

This thesis was entirely written by the author, but the contributions of Professor Stephen Murphy and Jason Phoenix should be acknowledged. Dr. Murphy prepared the initial research proposal and experimental design which were presented to Algonquin Provincial Park. Jason Phoenix was the researcher who implemented the restoration treatments and performed initial monitoring. Algonquin Park Staff were involved in the initial planning of this restoration experiment.

Abstract

The understorey of a forest provides essential habitat for wildlife, increases habitat heterogeneity, and enhances soil nutrients through the nutrient-rich litterfall of understorey vegetation. In red pine plantations (a common plantation type in southern Ontario and Algonquin Park), the understorey is sparse to non-existent. This is due to the dense overstorey reducing light, and the thick layer of fallen pine needles, which suppresses vegetation growth and acidifies the soil.

This thesis documents the results of a four-year (2021-2024) collaboration between the University of Waterloo and Algonquin Provincial Park to restore the understorey of a 50-year-old red pine plantation. The treatments tested were tree thinning and herbicide application to invasive species (applied to the entire site), applied nucleation and windthrow guards. Applied nucleation is the planting of trees and/or shrubs scattered throughout an area, to mimic natural succession. Beaked hazel shrubs and yellow birch trees were used as the applied nuclei in this experiment. Windthrow guards are when an herbaceous species is planted around a woody species to reduce wind damage. The term windthrow guard was coined for this project, as this method has not been tested in scientific literature before. Canada goldenrod was used as the windthrow guard in this experiment. The five treatment types used were: (1) control: received no additional treatments other than the tree thinning and herbicide application, (2) applied nucleation, (3) windthrow guard, (4) applied nucleation with windthrow guard, (5) unthinned: received to restoration treatments and acts as a reference state for statistical analysis.

To determine the effectiveness of the windthrow guards and overall restoration success, mortality surveys of the transplanted trees and shrubs were performed. To determine the short-term outcomes of this experiment and establish monitoring best practices, vegetation surveys, insect trapping and identification and soil sampling and testing for pH, NO_3, P and K occurred. A Cochran's Q test with a post-hoc pairwise McNemar test with a Bonferroni correction was run on the mortality data. Differences in native and non-native vegetation community metrics (species richness, Simpson diversity and Hill-Shannon diversity) between the different treatment types were assessed with generalized linear mixed models with a post-hoc Tukey-adjusted estimated marginal means. Two-way mixed ANOVA's were used to assess the impact of treatment type and sampling time on insect community metrics (species richness, Simpson diversity and Shannon diversity). Kruskal-Wallis tests with a post-hoc Dunn test were run on each year of soil data, due to changes in sampling strategy, to determine the impacts of treatment type on each of the soil parameters (pH, NO_3, P, K).

There was a high transplant survival rate (over 80% in 2024). There were differences in the mortality of transplants between 2022 and 2024 in the applied nucleation treatment type ($p = 0.04$), but not the applied nucleation with windthrow guard plot type ($p > 0.05$) indicating that windthrow guards may reduce the mortality of transplanted trees and shrubs. The only statistically significant results of vegetation metrics between the reference and restored treatment types were in nonnative species richness and Hill-Shannon diversity in 2024 ($p > 0.05$), however, post-hoc tests did not show statistically significant differences of either of the metrics between the different treatment types. There were no statistically significant two-way interactions or main effects of treatment type or sampling time on any of the arthropod metrics ($p > 0.05$).

These results suggest that the understorey restoration had an acceptable outcome, and the area is recovering as expected, without many short-term changes in most monitoring metrics. The thesis ends with recommending further monitoring to assess the impacts of restoration impacts on the long-term site trajectory.

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Land Acknowledgement

I acknowledge that I live and work on the Territory of the Neutral, Anishinaabeg and Haudenosaunee Peoples. The University of Waterloo is situated on the Haldimand Tract, the territory granted to the Six Nations that includes six miles on each side of the Grand River. The research site for this experiment, as well as the majority of Algonquin Park, is situated on the traditional territory of 10 Algonquin communities, some of whom refer to themselves as the Algonquins of Ontario. This land was sold to the British government without consulting the Algonquin people, who never surrendered their claim to the land. Indigenous people have been here since time immemorial, and as a settler, I am grateful for the opportunity to live, work and study on these lands. Conservation is inherently rooted in colonialism, as it involves the removal of Indigenous people from their land. Until this is widely acknowledged, true reconciliation cannot occur.

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List of Abbreviations

UN Unthinned

AN Applied Nucleation

WG Windthrow Guard

ANWG Applied Nucleation with Windthrow Guard

Chapter 1

1.1 Restoration

Ecological restoration can be defined as "a solutions-based approach that engages communities, scientists, policymakers, and land managers to repair ecological damage and rebuild a healthier relationship between people and the rest of nature" (Gann et al., 2019). It has been prioritized in recent decades due to biodiversity decline and land/ecosystem deterioration and requires strategies and solutions that are both ecologically efficient and cost-effective (Piaia et al., 2020). One of the main factors determining whether an ecological restoration project will be approved and/or funded is the project's cost, and limited funding is a major barrier to this. (Brancalion et al., 2019). Reducing costs is essential for large-scale ecological restoration (Brancalion et al., 2019). To remedy this, lower-cost strategies, as well as methods with a higher effort-to-cost ratio, must be developed and tested to understand their effectiveness (Brancalion et al., 2019).

1.1.2 Forest Restoration

At a landscape (large) scale, a patchwork mosaic of different types of landcover would support more ecological diversity and encourage beneficial interactions between types of landcover (Jamison et al., 2023; Mayor et al., 2025). One landcover or ecosystem type which has commanded a large focus in the field of restoration ecology is forests (Fagan et al., 2020; FAO, 2018). Since 1990, there has been a global loss of 178 million hectares of forests (Global Forest Resources Assessment 2020, 2020).

There are multiple drivers of forest loss, and they can vary regionally as well as with forest type (Curtis et al., 2018). Some of the main global factors driving forest loss include forestry (Hoang & Kanemoto, 2021; Pendrill et al., 2019), urbanization (Curtis et al., 2018; Hoang & Kanemoto, 2021), and commodity production (Hoang & Kanemoto, 2021; Kan et al., 2023; Pendrill et al., 2019). In temperate and boreal forests, wildfire and forestry are large drivers of forest loss (Curtis et al., 2018).

More than a billion people worldwide rely on forests as a source of food, medicine and fuel (FAO, 2018). Forests provide ecosystem services such as protection from landslides and flooding

(Seymour et al., 2022; Winkel et al., 2022), acting as carbon sinks and moderating the local climate (Lawrence et al., 2022; Seymour et al., 2022; Winkel et al., 2022). Forests also contain over three-quarters of the world's terrestrial biodiversity (FAO, 2018). Due to the ecosystem and social services provided by forests, as well as the amount of forest loss which has occurred, restoring them has become a priority. However, forest restoration efforts face issues and barriers. These include differing restoration goals between stakeholders and practitioners (Höhl et al., 2020), land use and policy changes (Fagan et al., 2020), project funding (or lack thereof) (Höhl et al., 2020), the scaling up of restoration techniques, as most of them have only been tested at small scales (Holl et al., 2024), and the gaps between academics and practitioners (Holl et al., 2024).

1.2 Pine Plantations

A major reason for forest ecosystem degradation is due to conversion to forest plantations. Plantation forests can be defined as "intensively managed, composed of 1 or 2 species, even-aged, planted with regular spacing and established mainly for productive purposes" (Global Forest Resources Assessment 2020, 2020). Plantation forests make up three percent of the global forest area (Global Forest Resources Assessment 2020, 2020). In southern Ontario, two million hectares are forested (Parker et al., 2008), and plantation forests make up 12% of the total forest cover (Falconer & Nol, 2021). The majority of these plantation forests are red pine (*Pinus resinosa*) plantations (Falconer & Nol, 2021; McLaughlin et al., 2010; Parker et al., 2008).

The establishment of red pine plantations in Ontario began in the 1920s, as a way to combat deforestation caused by earlier settlers clearing land for agriculture (LandOwner Resource Center, 1999). This mainly occurred on public land, as it was due to initiatives of the provincial government (LandOwner Resource Center, 1999). In the 1960s, reforestation on private land became a large focus, resulting in pine plantations established on both private and public lands (LandOwner Resource Center, 1999). Red pine has a number of commercial uses, including interior and exterior home finishings, and structural timber (Government of Ontario, 2024). Logging has occurred in the Algonquin Park area since before it was designated as a park or protected area (Strickland, 2018). Algonquin Park is the only Provincial Park in Ontario that still allows forestry activities (including logging) to occur (Lane et al., 2024). These activities

can only occur in the Recreation/Utilization zones of the Park, which consists of 65.3% of the Park area (roughly 500 000 hectares) (Lane et al., 2024). Red pine trees are the fourth most-logged species in the Park (Strickland, 2018), and 1 000 000 red pine seedlings are estimated to be needed to fulfill the goals of the Parks 2021-2031 forest management plan (Lane et al., 2024).

Unmanaged red pine plantations create a dense monoculture, where all the trees are the same age and species, and planted equidistantly apart, creating little to no habitat heterogeneity or diversity (Hartman et al., 2005). Additionally, the branches of pine plantations shade out light to the understorey, and the thick layer of pine needles (up to six centimetres) acidifies the soil and can suppress vegetation growth (Hartman et al., 2005). Spatiotemporal diversity in forests allows for more habitat types for plant and lichen groups, which in turn support more dependent organisms (Haeussler et al., 2021). For example, the species diversity of ground beetles (family Carabidae) was lower in pine plantations when compared to more diverse forests (Jouveau et al., 2022). The lack of overstorey diversity in pine plantations reduces the overall diversity in these areas (Bertacchi et al., 2016; Medina-Torres et al., 2022).

As plantations age, they generally become more diverse (Bertacchi et al., 2016; Medina-Torres et al., 2022). The increase in diversity is thought to be due to treefall gaps introducing more spatial heterogeneity, decreases in generalist species and more time for the colonization of native species (Onaindia et al., 2013). Red pine trees do not regenerate well in plantations (Goldblum, 1998; Hartman et al., 2005), indicating that in some cases natural succession may eventually cause a conversion from a plantation to a mixed wood forest. However, if only natural succession occurs, without any restoration intervention, a long time frame is needed for change from a plantation to a mixed wood forest (Onaindia et al., 2013). Succession from a conifer plantation to a mixed hardwood forest would occur on a time scale of decades to centuries. For example, Zhu et al. (2023) found the community composition of a naturally regenerated forest and a pine plantation were significantly different after over 60 years of passive restoration, meaning that the pine plantation had not recovered yet. The vast majority of the species present in the plantation were generalists, not specialists, and it is thought that seed distributors of specialist species had gone locally extinct, severely hampering the ability of passive recovery to succeed (H. Zhu et al., 2023). Additionally, natural forests can act as seed sources for plantations

if they are located nearby, making plantation location an important factor in the possibility of natural succession succeeding (Simões et al., 2024; Wang et al., 2022). Active management is often needed to accelerate biodiversity in pine plantations, due to the number of factors hampering natural succession (Azor et al., 2015).

1.2.1 Forest Understories

The understorey of young forests (about 100 years old or younger) is formed by a combination of species already present, germination from the seed bank, and species migrating from adjacent forests (Bossuyt et al., 2001). The above-ground biomass of the herbaceous layer is less than 1% of the forest biomass, but it contains, on average, over 80% of the plant species richness of a forest (Gilliam, 2007). The understorey of the forest has a disproportionate impact on temperate forest communities, given its comparatively low biomass (Gilliam, 2007). In evergreen forests, the understorey area contributes to the overall function of the forest (Chastain et al., 2006) and has a net aboveground primary productivity that is over half of that of the trees (Nilsson & Wardle, 2005). When simulations were run with and without an understorey area, the simulations with an understorey stored significantly higher amounts of carbon and nitrogen than the simulations without an understorey (Chastain et al., 2006). Understorey vegetation litterfall has higher nutrient concentrations than overstorey litterfall, so the presence of understorey litterfall increases the nutrients which are decomposed into the soil (Gilliam, 2007; Nilsson & Wardle, 2005; Tappeiner & Alm, 1975). Understorey vegetation also increases the total amount of organic matter on the forest floor (Gilliam, 2007; Nilsson & Wardle, 2005; Tappeiner & Alm, 1975). Additionally, there is a positive relationship between understorey vegetation species richness and density and microbial markers such as biomass, respiration and decomposition activity (Hedwall et al., 2017).

1.2.2 Pine Plantation Understories

Pine plantations result in habitat degradation on many scales when compared to a naturally regenerating forest (Azor et al., 2015). The growth of a well-developed and species-rich understory is not promoted by forest plantation environments (Aubin et al., 2008). Pine plantations affect the properties of topsoil (Marcos et al., 2010). Compared to natural deciduous forests, coniferous plantations have been demonstrated to have more acidic soil (Hızal et al.,

2013), and altered levels/concentrations of soil nutrients such as nitrogen (Hızal et al., 2013; Scott & Messina, 2010), potassium (Hızal et al., 2013; Scott & Messina, 2010), phosphorus (X. Zhu et al., 2021), and calcium (Duffy et al., 2024; Hızal et al., 2013). Litter from coniferous trees is acidic, so as it decomposes it acidifies the soil (Turk et al., 2008). Additionally, litter from pine trees takes a long time to decompose due to the high lignin content in the needles, resulting in thick layers of litterfall on the forest floor (Duffy et al., 2024; Hartman et al., 2005). Litter in pine plantations also contains less moisture than litter in deciduous forests (Scott & Messina, 2010). Additionally, the thick layer of needles can prevent dropped seeds from ever reaching the soil, limiting germination abilities and suppressing vegetation growth (Duffy et al., 2024; Hartman et al., 2005).

Pine plantations also cause a reduction in the amount of light reaching the understorey (Jayawardhane & Gunaratne, 2020; Tartarino et al., 2020) which impacts the community composition present (Buskey et al., 2020; Chevaux et al., 2022; Majasalmi & Rautiainen, 2020) as some species require more light than others. These factors (among others) result in understorey vegetation communities with less species richness, abundance, cover and diversity in pine plantations when compared to natural hardwood forests (Duffy et al., 2024; Medina-Torres et al., 2022). Due to the contributions of soil nutrients by understorey vegetation (Gilliam, 2007; Nilsson & Wardle, 2005; Tappeiner & Alm, 1975), this decrease in understorey vegetation could result in less nutrient availability in a plantation environment. This impact on the soil directly effects the vegetation community present, as soil factors such as pH, moisture and nutrient concentrations govern the plant species which can grow and regenerate in these areas (Cardoso et al., 2013; Prober & Wiehl, 2012).

1.3 Restoration of Pine Plantations

Restoration actions can be grouped into two categories: passive restoration and active restoration (Gann et al., 2019). Passive restoration is akin to natural succession, where the area is left to recover on its own, however, this only works in areas where there is high potential for unassisted ecosystem or habitat recovery (Gann et al., 2019). As passive restoration can occur without any intervention, it is not technically in line with the definition of restoration (Chazdon et al., 2024).

Active restoration is a more appropriate choice for areas where there is little to no ability for natural regeneration, and more intervention is necessary (Gann et al., 2019).

1.3.1 Tree Thinning

Tree thinning has direct impacts on the forest plant community. It can have positive results on the remaining trees in the forest community as they can increase in diameter, height (Parker et al., 2001; Zhang et al., 2016), and volume (Parker et al., 2001) after thinning occurs. The understorey plant community is also impacted by tree thinning (Venier et al., 2014). Generally, right after thinning activities, there is a decrease in understorey plants due to mechanical damage from the machinery and falling trees (Venier et al., 2014). However, understorey vegetation can regrow from surviving roots, rhizomes and residual stems, and vegetation expansion can occur from plants which survive in crevices and bases of tree stumps (Venier et al., 2014).

Tree thinning results in changes in the understorey microclimate and microenvironment, including increased habitat heterogeneity and light levels (Li et al., 2020; Parker et al., 2001; Yang et al., 2023). These changes to the microclimate and microenvironment have a multitude of benefits to the understorey. Benefits include increases in understorey vegetation diversity (Ares et al., 2010; Dang et al., 2018; Li et al., 2020), species richness (Jang et al., 2021; Yang et al., 2023), vegetation cover (Ares et al., 2010; Jang et al., 2021; Yang et al., 2023), density, and aboveground biomass (Cheng et al., 2017). Even small gaps from thinning can help to increase forest heterogeneity by creating size and age variation in the forest community (Rouvinen & Kouki, 2011).

In the long term, after tree thinning, there are mixed results of the impacts of thinning on understorey vegetation. Yang et al. (2023) found species richness was highest 14 years after thinning, and Dang et al. (2018) found species diversity was higher in thinned plots 11 years after thinning activities. Conversely, Jang et al. (2021) found that understorey species richness peaked 3-5 years after thinning and declined 23 years after thinning. He and Barclay (2000) found thinning had minimal effects on understorey vegetation, with the only effects being on the two most abundant species. In the long term, after thinning occurs, regenerating trees begin to

take up the previously empty space, and additional thinning may be required, as the new tree growth may eventually decrease herb and shrub cover (Ares et al., 2010; Jang et al., 2021).

1.3.1.1 Benefits of Deadwood

After tree thinning occurs, the trees can be left on the ground instead of being removed. This material can be referred to as deadwood, logging slash, or coarse woody debris. Coarse woody debris in the forest can help to prevent ecosystem degradation due to environmental extremes and also improve forest productivity (Debeljak, 2006). Leaving deadwood on the forest floor instead of removing it contributed to the release of low amounts of nitrogen, phosphorus and organic carbon onto the forest floor (Burzyńska & Sztabkowski, 2023). Deadwood also helps to reduce the amount of hillslope soil erosion and runoff on skid trails (trails left from the logging equipment) for the first one to two years after logging disturbance (Robichaud et al., 2020). Many rare species in the Fennoscandia forest area responded positively to the presence of fresh, coarse, woody debris (Koivula & Vanha-Majamaa, 2020). Dry wood and wood cavities can provide nesting habitat to some species of native bees (Da Silva Carneiro et al., 2024) and also provide microhabitats to insects which specialize in dead wood survival (Bouget & Duelli, 2004). Additionally, some lichen and moss species are epixylic, meaning they prefer to grow on dead wood, and leaving slash provides a habitat for them (Haeussler et al., 2021). As demonstrated, there are a myriad of benefits to leaving deadwood after logging.

1.3.2 Herbicide Application

Herbicide application is a common invasive plant management strategy. Ecosystems can be threatened by invasive species (Mervosh & Gumbart, 2015). Invasive plants can alter the native plant community to the extent that additional active restoration may be required to restore the ecosystem to pre-invasion levels (Reid et al., 2009). Due to this, invasive plant removal should not be the only strategy or tool used in a restoration project (Reid et al., 2009).

The removal of invasive plant species can cause changes to the understorey community of forests (Buskey et al., 2020). As long as commonly used forest herbicides are applied according to the labelled instructions, there are few long-term impacts on understorey species richness, diversity and herbaceous cover as the vegetation can recover from herbicide

application (Briggs et al., 2024). Multiple studies have found positive impacts of herbicide application on the vegetation community. Applying a glyphosate-based reduced the cover of invasive vegetation species by an average of 97% when tested at sites experiencing a variety of environmental conditions (Hinkson et al., 2024). Herbicide application was found to increase the species richness of herbaceous plants and reduce the cover of woody plants in the understorey (Schafer et al., 2014). The removal of invasive shrubs with herbicides increased the basal area and number of stems of the overstorey (Ulyshen et al., 2022).

The impacts of herbicide application on insects are less straightforward. Briggs (2024) found the effects of herbicide application on insects were variable, with some groups having conflicting responses to herbicide application and others not having many negative impacts. Ulyshen (2022) found that invasive shrub removal with herbicide increased the bee community to reference levels, and the butterfly population also increased (although it was not at reference levels by the end of the study). Conversely, herbicide application did not have an impact on an endangered butterfly population when applied to degraded prairie land in Oregon (Schultz & Ferguson, 2020). The impacts of herbicide application on insect communities are likely taxa-specific (Briggs et al., 2024). Herbicide applications can have overall positive impacts when used correctly, due to habitat improvements which can occur when invasive vegetation species are removed.

1.3.3 Applied Nucleation

Measures such as planting species that don't generally occur in pine plantations can help to accelerate succession (Onaindia et al., 2013). Applied nucleation is a restoration technique which involves planting clusters of woody vegetation (often trees) scattered throughout a cleared area (Corbin & Holl, 2012). These clusters are the nuclei, and they will expand as new seedlings are recruited, which mimics natural succession/regeneration (and passive restoration) more closely than tree plantations (Campanhã Bechara et al., 2021; Corbin & Holl, 2012). Applied nucleation has positive effects on the forest community, including increased recruitment and higher density of understorey vegetation (Campanhã Bechara et al., 2021), improving seedling abundance, survivorship and size (De Oliveira Bahia et al., 2023), as well as accelerating the natural recovery of vegetation species (Holl et al., 2024).

Applied nucleation is less expensive than planting tree plantations (Campanhã Bechara et al., 2021; Holl et al., 2020; Zahawi et al., 2013) and can be as, or more, effective than planting plantations (Zahawi et al., 2013). Applied nucleation has mainly been used in the context of planting trees in a clear-cut area (Corbin & Holl, 2012) and not in terms of planting shrubs in a pine plantation. However, it may be useful in ecosystems which have stressful abiotic conditions (Corbin & Holl, 2012), which could include an unmanaged pine plantation. Additionally, it was found that a combination of restoration techniques (such as nucleation combined with another treatment) had a greater impact on the plant communities (De Oliveira Bahia et al., 2023). Variation in the effectiveness of this technique is at least also partly due to geographical and biological factors of the restored areas (De Oliveira Bahia et al., 2023).

1.3.4 Windthrow Guards

One technique that could be used in combination with applied nucleation is windthrow guards. Windthrow is a term which generally refers to trees being uprooted by windstorms. However, this term could also be extended to smaller plants, such as shrubs. Wind is a natural disturbance factor in forests, which has large ecological consequences, such as tree death, canopy gaps (Bouget & Duelli, 2004; Taeroe et al., 2019), and increased heterogeneity of the forest floor (Venier et al., 2014). In research forests in Japan, it took 30 years for the forest canopy to recover from a severe windthrow event (Li et al., 2023). Windthrow damage is often seen in tree stands which are forty-plus years old (Türk et al., 2023). Windthrow guards are when an herbaceous plant is planted in a ring around a woody plant to protect it from wind damage or windthrow. Theoretically, the herbaceous plant should direct wind upwards rather than allowing the wind to hit the woody shrub directly.

Similar ideas have been used before, such as the idea of "nurse plants," which have been successful in ecological restoration projects (Dona & Galen, 2007; Endo et al., 2008; Gómez-Aparicio et al., 2004; Padilla & Pugnaire, 2006; Vloon et al., 2022). Nurse plants are pre-existing shrubs which are left when tree planting occurs instead of being removed (Gómez-Aparicio et al., 2004). Gómez-Aparicio et al. (2004) found that seedling survival under shrubs was more than double when compared to seedlings in open areas and Endo et al. (2008) found that nurse plants

resulted in higher rates of seedling establishment and survival. The use of nurse plants has also resulted in increased vascular plant species richness and increased recruitment of the nurse plant species (Vloon et al., 2022). Forest restoration in Uganda found that when a higher proportion of banana plants and remnant shrubs were present in a restoration area, the restored forest had increased positive attributes such as increased compositional similarity to old-growth forests (Ssekuubwa et al., 2022). It is thought that nurse plants help to buffer against stressors by introducing beneficial soil microbes, protecting seedlings from herbivores (Callaway, 1995), and providing a more sheltered microclimate (Dona & Galen, 2007).

1.4 Indicators

Indicators can be defined as "specific, quantifiable measure of attributes that directly connect longer-term goals and shorter term objectives." (Gann et al., 2019). A restoration project should include specific and measurable indicators which can be monitored to determine whether the ecological goals of the project are being met (Gann et al., 2019). As responses to restoration can be species-specific, indicators need to be chosen with a high degree of thought and care (Strobl et al., 2020), and should also be chosen with the ecosystem type in mind (Gatica-Saavedra et al., 2017). Chosen indicators for restoration should be easily measured, change over time, and be good surrogates for ecological processes and ecosystem services (Suganuma & Durigan, 2015). As seen in Table 1, there are many potential indicators that have been identified.

Table 1. An incomplete list of commonly used indicators in forest restoration.

Potential Indicator	**Can Indicate**	**References**
Insect/Arthropod Community Composition	Habitat disturbance or change	Akutsu et al., 2007; Chowdhury et al., 2023; Noreika et al., 2020
	Habitat Structure	Humphrey et al., 1999; Maleque et al., 2006; Maritano, 2020
	Ecosystem Diversity	Borges et al., 2021; Chowdhury et al., 2023; Maleque et al., 2006
	Soil Quality	Chowdhury et al., 2023
Bee Community Composition	Plant Pollinator Interactions; Plant Reproduction	Da Silva Carneiro et al., 2024

Vegetation Cover	Soil Quality Changes; Succession	Park et al., 2024; Piaia et al., 2020
Vegetation Species Richness & Diversity	Nutrient Availability; Habitat Disturbance Increased Vegetation Cover; Recovery of diversity	Johansen & Evju, 2024; Park et al., 2024; Suganuma & Durigan, 2015
Vegetation Community Changes	Site/Habitat change; Nutrient availability	Palik & Kastendick, 2023; Strobl et al., 2020
Basal Area	Forest recovery; Carbon sequestration, Effects of thinning	Palik & Kastendick, 2023; Suganuma & Durigan, 2015
Canopy Cover	Ground cover; Understorey plant community development; Understorey species diversity; Seed dispersal	Piaia et al., 2020; Tartarino et al., 2020; Zahawi et al., 2013
Diameter at Breast Height (DBH)	Forest Density; Forest structure	Campanhã Bechara et al., 2021; Suganuma & Durigan, 2015
Soil Organic Carbon	Plant production and growth; Vegetation restoration; Overstorey composition	Qiu et al., 2018; Zhao et al., 2023
Soil pH	Overstorey composition; Microbial activity; Nutrient availability; Plant composition	Cardoso et al., 2013; Costantini et al., 2016; Hızal et al., 2013
Soil Nitrogen	Understorey vegetation composition; Native plant diversity	Prescott et al., 2000; Prober & Wiehl, 2012
Soil Phosphorus	Litter quality; Overstorey composition; Native plant diversity	Prober & Wiehl, 2012; Zhu et al., 2021
Soil Potassium	Water availability use; Vegetation recovery	Qiu et al., 2018; Sardans & Peñuelas, 2015
Soil Microbial Markers (biomass, respiration and decomposition)	Understorey species richness and density; Nutrient availability	Chandra et al., 2016; Hedwall et al., 2017

Diversity measures can provide information about how individuals of species are structured across space and time, however, it is important to keep in mind that these measures provide only part of the picture, as the factors influencing where species are located are complex and multifaceted (Mittelbach & McGill, 2019). They can provide fine resolutions to species diversity data (Qiao et al., 2024). Diversity measures operate with assumptions about the data, so they must be used with those assumptions in mind (Qiao et al., 2024). Simpson's diversity represents diversity differences at different sites with higher accuracy, when compared to other diversity models/measures (Morris et al., 2014) and seem to perform better at a smaller scale (such as plot level) (Qiao et al., 2024). Using multiple diversity measures can provide a more overall understanding of species diversity occurring (Morris et al., 2014).

Species richness can be defined as "the number of unique species found in a local community" (Mittelbach & McGill, 2019). However, species richness values are correlated with sample size and tend to have issues representing the actual abundance of rare species, as sampling captures them infrequently or not at all, causing species richness value to be underestimated (Gwinn et al., 2016; Mittelbach & McGill, 2019; Roswell et al., 2021). There are methods to correct this underestimation (Gwinn et al., 2016; Oliveira et al., 2024). Changes in species richness values can be used in a variety of ways in ecological monitoring, such as assessing the impact of disturbance and testing the impacts of management options (Gwinn et al., 2016).

Hill numbers can be defined as "the effective number of species" (Ricotta & Feoli, 2024). Using different exponents while calculating Hill numbers gives results which are proxies for the "traditional" metrics of species richness, Shannon diversity and Simpson diversity (Roswell et al., 2021). For example, using an exponent of,1 results in a Hill-Shannon diversity metric, which is fairly robust to bias (Qiao et al., 2024) while still being able to account for species that are very rare or very common (Roswell et al., 2021). However, Hill numbers behave linearly, and ecological processes and relationships do not always behave in a linear fashion (Ricotta & Feoli, 2024). Hill numbers are commonly used in ecology, and as they result in values that are on the same scale, they are more easily comparable (Roswell et al., 2021).

Species richness and diversity measures are often used as proxies for restoration outcomes (De Oliveira Bahia et al., 2023; Wortley et al., 2013). Insect and vegetation species richness and diversity metrics are commonly used in ecological monitoring (Evju et al., 2020; Ruiz-Jaen & Mitchell Aide, 2005). They are cost-effective to survey, operate on relatively short time scales, react to habitat change, and have large impacts on the ecosystem itself (Akutsu et al., 2007; Chowdhury et al., 2023; Hodkinson & Jackson, 2005; Johansen & Evju, 2024; Strobl et al., 2020). Recovery timelines for species richness and diversity differ (Suganuma & Durigan, 2015), and using both of these species metrics could provide a clearer picture of the state of the ecosystem.

Chemical soil parameters can be used as proxies of ecosystem processes such as nutrient cycling and decomposition and can also directly indicate nutrient availability for vegetation (Costantini et al., 2016). Chemical soil parameters are also the most commonly used class of soil indicator for monitoring forest restoration (Gatica-Saavedra et al., 2023). Soil chemical indicators respond more quickly to forest restoration efforts when compared to biological or physical components of soil (Allek et al., 2023). When operating on a short-term scale, soil properties which can recover comparatively quickly, such as pH, nitrogen, potassium and phosphorus, are suitable options for soil testing (Cardoso et al., 2013; Costantini et al., 2016). These measurements are also comparatively inexpensive and simple to test (Cardoso et al., 2013; Gatica-Saavedra et al., 2023). Nitrogen and phosphorus can be colimiting in forest ecosystems, so the availability of both is important to understorey vegetation diversity (Hedwall et al., 2017). Nitrogen, phosphorus and potassium availability and cycling have positive impacts on vegetation biodiversity recovery (Qiu et al., 2018).

Restoration projects that use nodal or nucleation techniques are attempting to mimic natural succession more closely, as well as use techniques that are simpler and more cost-effective (Campanhã Bechara et al., 2021; De Oliveira Bahia et al., 2023). Transplant survival, growth and reproductive success are commonly used indicators to determine the outcome of these restoration activities (Doyle et al., 2023; Menges, 2008). Restoration projects that involve conifer or pine plantations tend to involve increasing habitat heterogeneity as well as mitigating

the negative impacts of pine needles on the understorey and the soil (Jayawardhane & Gunaratne, 2020; H. Zhu et al., 2023).

A forest restoration project involving thinning a pine plantation, using herbicide to remove invasive plant species, and then using nodal techniques in the understorey could have impacts on the habitat which include changes to species richness, diversity, and chemical soil parameters (Chandra et al., 2016; De Oliveira Bahia et al., 2023; Hinkson et al., 2024; Jayawardhane & Gunaratne, 2020; Li et al., 2020; Yang et al., 2023; Zhu et al., 2021). Therefore, a project that is studying the effectiveness of understorey restoration techniques in a pine plantation over four years would want to use indicators that are simple, cost-effective, and indicate natural succession and habitat heterogeneity. Some indicators that fit this criterion are species diversity and species richness of vegetation and insects, chemical soil parameters and the mortality of the nuclei species.

1.5 Research Context

This project is a four-year collaboration between Algonquin Provincial Park and the University of Waterloo. The first two years of this project (2021 & 2022) were completed by a previous researcher, and the last two years of this project (2023 & 2024) are being implemented by the current researcher. This project is studying rapid understorey restoration techniques of a red pine plantation located on the shore of Cache Lake in Algonquin Provincial Park, Ontario, Canada. The pine plantation being used by this project was planted in the early 1970's. It had been unmanaged until this project began, meaning that the trees had not been thinned. This resulted in the trees growing to be tall and slender, as there was competition for light. Lack of tree thinning also resulted in the branches of the trees growing close together, limiting the amount of light reaching the understorey and covering the ground with a thick layer of pine needles. These factors make vegetation in the understorey sparse to non-existent.

1.6 Research Objectives

As two major barriers to some restoration projects are the time that projects require (on the scale of decades) and the cost of projects, this project assessed techniques that are cost and time effective. This project determined the feasibility and success of various understory restoration techniques at the pine plantation in the short term. It also determined which of these methods (if any) will accelerate the timeline for understorey regeneration. The treatments tested were tree thinning and herbicide application alone, as well as tree thinning and herbicide application in conjunction with applied nucleation and windthrow guards. The windthrow guards will surround the nucleation species in this experiment, theoretically protecting them from wind damage by diverting the wind-up and away. The restoration technique of a windthrow guards has never been formally tested in any scientific literature and will fill the knowledge gap of whether windthrow guards reduce mortality of transplanted shrubs and trees. The species used for the applied nucleation treatment are beaked hazel (*Corylus cornuta*) and yellow birch (*Betula alleghaniensis*). These species were chosen partially due to availability. There was a colony of shrubs/trees located close to the site in a utility corridor. As the plants may be damaged or killed during power line maintenance, some were used in this experiment. The species used for the windthrow guard treatment is Canada goldenrod (*Solidago canadensis*). There was also a large patch of Canada goldenrod located near the site, making its use logistically feasible. Additionally, all these species possess characteristics that should allow them to survive in the understorey of a red pine plantation.

1.7 Research Questions

1. Are there short-term changes to the vegetation and arthropod species richness and diversity metrics due to restoration efforts?
2. Are there short-term changes to chemical soil parameters due to restoration efforts?
3. Are windthrow guards effective at reducing the mortality of transplanted shrubs and trees?

Chapter 2

2.1 Methods

The first two years of this experiment mainly focused on implementing the restoration treatments and devising an early monitoring plan. The second two years of this project primarily focused on revising the monitoring plan and conducting statistical analysis of the preliminary results of this project.

2.1.1 Study Area[1]

This experiment was conducted in Algonquin Provincial Park in Ontario, Canada (Figure 1). The coordinates of the centre of the site are 45 33’08.7” N, 78 35’42.0” W. This area received an average of 840.4 mm of rainfall, and 345.2 cm of snowfall a year from 1981 to 2010 (Government of Canada, 2024). The daily average temperature ranged from -11.7 degrees Celsius (°C) in January to 18.1 °C in July from 1981 to 2010 (Government of Canada, 2024). The slope of the area is classified as rolling to hilly, with a coarse-textured soil, and the dominant soil type is classified as orthic podzol (IUSS Working Group WRB, 2022).

[1] For more detailed information about the history of the site, experimental grid set-up, application of the restoration treatments and monitoring that occurred during the first half of this project, please refer to Phoenix, 2022.



Figure 1. a) A map with a white box showing the location of the research site. The box is not to scale (Google Earth, n.d.-b) b) A map showing the outline of the research site borders (Google Earth, n.d.-a).

2.1.2 Experimental Setup and Design

Prior to the set-up of the experimental grid, the area was thinned by Algonquin Park Staff between December 2020 and February 2021. After thinning occurred, invasive species management occurred; glyphosate was applied in June and August 2021, and triclopyr was applied in May 2022.

In October of 2021, the previous researcher set up the main experimental grid. The site is composed of four blocks, each with a ten-metre buffer zone around it. Each block is divided into an eight-by-eight grid of treatment plots, and each plot is eight metres by eight metres. The entire site is 158 metres by 158 metres.

The treatments were assigned to the plots using a randomized block design. There were four treatment types:

1. Control (C, received no treatments other than the thinning and herbicide application that the entire site received);
2. Applied nucleation (AN, received either an eastern beaked hazel or yellow birch transplant);
3. Windthrow guard (WG, received a ring of 6 Canada goldenrod stems);
4. Applied nucleation with windthrow guard (ANWG, received an applied nucleation transplant with a windthrow guard planted around it).

In May of 2023 four unthinned plots which received no thinning, herbicide application or other treatments were added to the experiment to act as a reference state for statistical comparison by the current researcher. The unthinned plots are also eight metres square and are located adjacent to block number one (Figure 2).

This is a small-scale restoration project, as there is only one site, and it is comparatively small (roughly 25000 m^2). Additionally, there is only one ecosystem type, as the project is not large enough to contain multiple ecosystem types within a patchwork mosaic. However, there are multiple spatial scales at play, as restoration efforts occurred in a roughly 25000 m^2 area, and

monitoring efforts occurred in multiple $8m^2$ plot scales to $1m^2$ quadrat scales. Operating on various scales is common for forest restoration (Dodson & Peterson, 2010), as monitoring often cannot feasibly occur on the same scale as restoration efforts themselves. Additionally, this project is operating on a relatively short timescale, as four years is not a lot of time in the timescale of ecological change.

In the short-term, the goals of this project are to determine monitoring best practices, collect and analyze baseline data, and to determine the effectiveness of the restoration treatments in the short term (which can be indicated by transplant survival). In the long term (20 to 30 years and beyond), the establishment of a tolerant hardwood and hemlock forest, the main forest type on the west side of the park (Lane et al., 2024) is likely. This is indicated by the adjacency of the plantation to a forest of this composition, and the presence of sugar maple, red maple, balsam fir, choke cherry and black cherry seedlings present in the understorey.

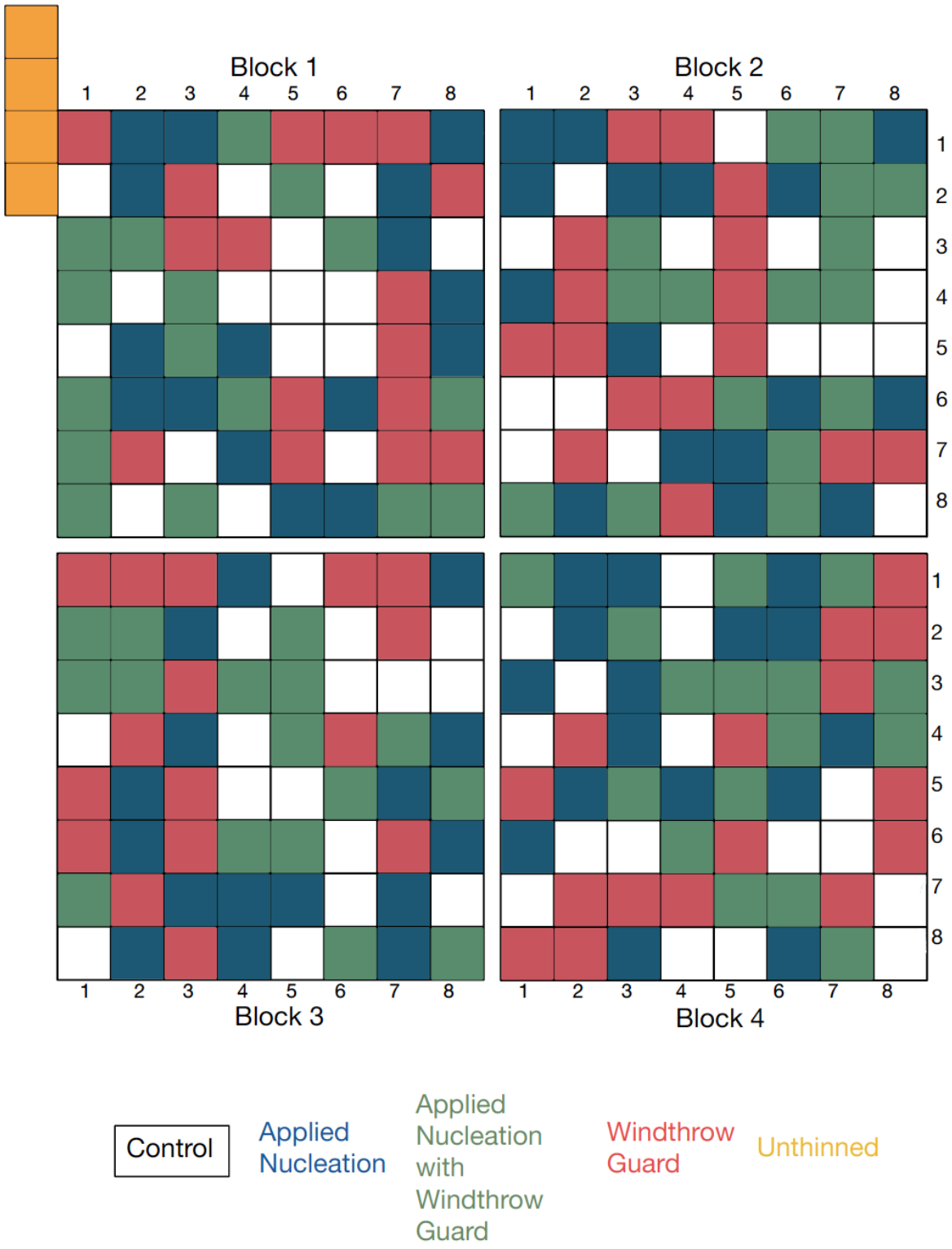


Figure 2. A schematic of the research site showing which treatment types were assigned to each plot. The figure is not to scale, each box is approximately eight metres by eight metres.

2.1.3 Beaked Hazel

One of the applied nucleation species used in this experiment was eastern beaked hazel. This species is tolerant to both light and shade, can grow up to ten feet tall (C. G. Armstrong et al., 2018), and have wide mono or single-layer crowns, which allows for more light absorption in the darker forest understorey (Markgraf et al., 2023). Beaked hazel shrubs can reproduce asexually (Tappeiner, 1971) and sexually (C. G. Armstrong et al., 2018). They are monoecious dicots (meaning they produce separate male catkins and female flowers on the same plant). (C. G. Armstrong et al., 2018). Beaked hazel produces catkins in the early spring and the hazelnuts are ripened in late summer or fall (Prevéy et al., 2020). These hazelnuts are large and wingless (Tappeiner, 1971). Hazelnuts are eaten and dispersed by a variety of animals including squirrels and deer (C. G. Armstrong et al., 2018; Tappeiner, 1971).

2.1.4 Yellow Birch

The other applied nucleation species used in this experiment was yellow birch. This tree species grow well in habitats with moderately sized canopy gaps (Bognounou et al., 2023; Ducey et al., 2023), soil scarification (Bognounou et al., 2023; Nolet et al., 2023), and well-drained soils (Bognounou et al., 2023). Yellow birch trees normally grow between 60 to 70 feet tall, but can reach 100 feet tall (Hosie, 1979). Yellow birch trees are monoecious dicots like beaked hazel shrubs (Perala & Alm, 1990). They have small and light seeds that are dispersed by wind (Canham & Murphy, 2016; Kern et al., 2019; Perala & Alm, 1990) and ripen in early fall (Perala & Alm, 1990). Yellow birch regeneration occurs most frequently on deadwood or decaying wood, followed by mineral soil (Lambert et al., 2016).

2.1.5 Canada Goldenrod

In this experiment, Canada goldenrod was used as the windthrow guard. Goldenrod is highly light-tolerant (Goldberg & Werner, 1983), which was useful in this experiment, as the tree thinning results in a large amount of light reaching the understorey. Goldenrod can also adjust to lower light conditions (which will occur as seedling and sapling trees grow) by growing more branching leaves near the top of the plant, to intercept more sunlight (Abrahamson & Gadgil, 1973). Canada goldenrod can survive in a wide range of soil textures and fertility levels, but it is typically found in areas with moist soil (F. Zhang & Wan, 2017). Canada goldenrod can

reproduce asexually through rhizomes as well as through seeds (Z. Zhang et al., 2022). The flowers of Canada goldenrod are pollinated by insects as they are self-incompatible (Coladonato, 1993). Canada goldenrod seeds are very small (Brisson et al., 2020) and are dispersed by wind (Z. Zhang et al., 2022).

2.1.6 Data Collection

Data collection occurred in the summers of 2021-2024, with 2021-2022 serving as a pilot study to test various approaches. The previous researcher performed data collection in 2021 and 2022, and I performed data collection in 2023 and 2024.

In the pilot study (2021-2022), the original researcher used a W-shaped transect to perform data collection activities. Expanding on this, I wanted to address the fact that this original transect did not include any of the applied nucleation with windthrow guard plot types. I added 4 applied nucleation with windthrow guard plot types to create an amended W-shaped transect (Figure 3).

The applied nucleation with windthrow guard plots were added as close to the original transect as possible. Although the unthinned plot types are not within the W-shaped transect, they were also surveyed/sampled at the same time as the plots in the W-shaped transect. When the W-shaped transect is mentioned, it is important to note that data collection activities also occurred in the unthinned plots. This occurred in 2023 and 2024, as the unthinned plot type was added to the project in 2023.

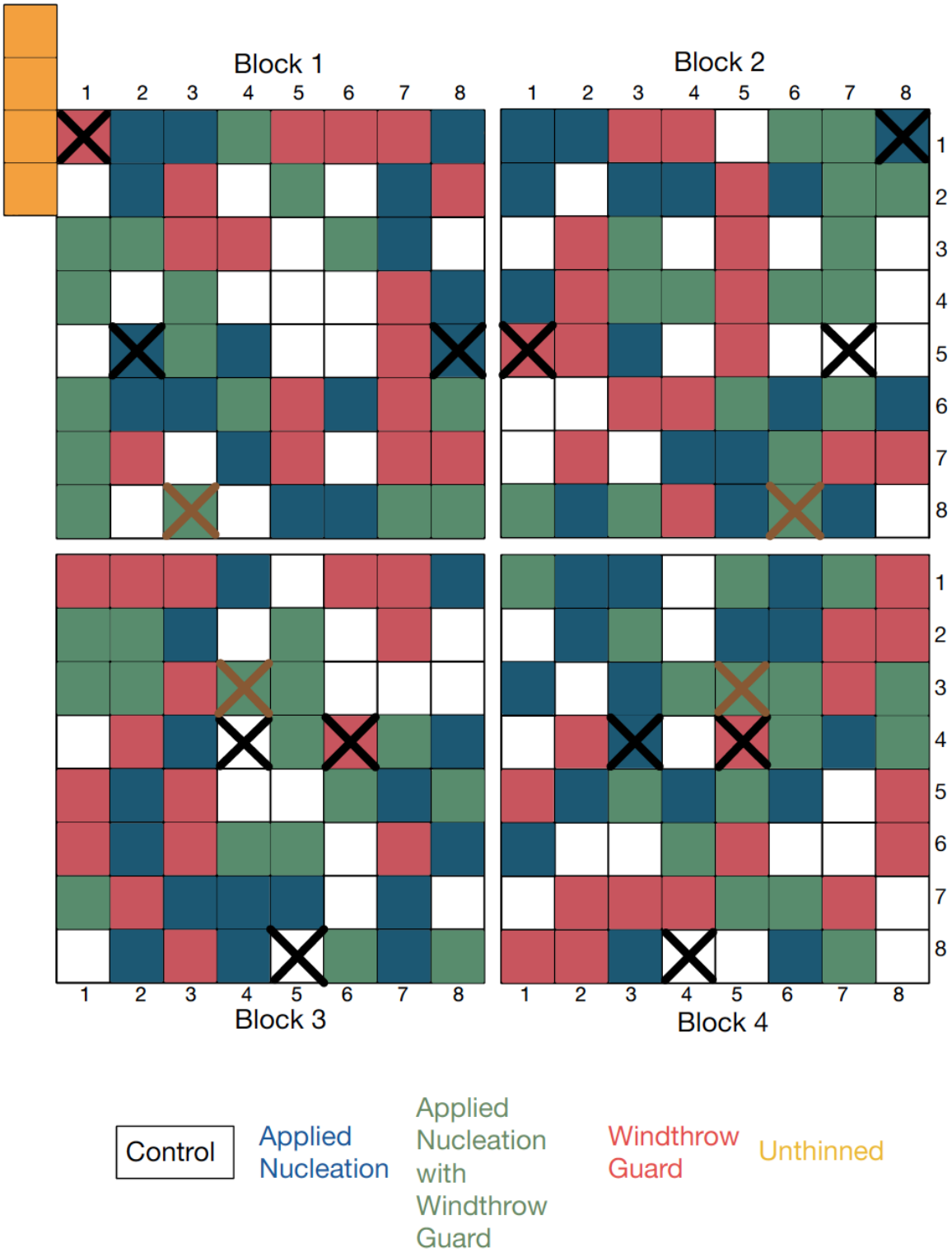


Figure 3. A schematic of the research site showing the treatment type assigned to each plot and the plots of the W-shaped transect. The black X's show which plots were a part of the original transect, the brown X's show the added plots. The diagram is not to scale, each box is approximately eight metres by eight metres.

2.1.6.1 Mortality Surveys

Mortality surveys occurred once a year from 2022-2024 and occurred in the applied nucleation, windthrow guard and applied nucleation with windthrow guard plot types. The original researcher and I used the same survey strategy, with no changes. Whether the applied nucleation transplant (if present) was alive or not was recorded (zero being dead and one being alive), as well as the number of living goldenrod stems (if present). If no living goldenrod stems were present, a zero was recorded. Originally, it was thought that all the applied nucleation transplants were beaked hazel shrubs. However, in 2024, the I realized that most of the transplants had been misidentified. The living transplants were identified to be either beaked hazel or yellow birch. Due to the extreme visual similarity of the two species without leaves, the dead transplants could not be definitively identified.

2.1.6.2 Vegetation Surveys

Vegetation surveys were performed once in 2022, three times in 2023 and four times in 2024. The survey in 2022 was performed by the original researcher, and I conducted the 2023 and 2024 surveys. The lesson learned from the original researcher was it was logistically impossible to do vegetation surveys in each plot – it was too time consuming. In 2023, I revised the survey strategy to focus on the W-shaped transect plots and further refined it in 2024 as the time required was still too long. In 2024, the survey area was changed to occur in a one metre squared quadrat, instead of the entire eight metre squared plot (Figure 4). The quadrat was placed in the middle of the plots which received no transplanted vegetation (unthinned and control), and north of the middle in plots which received transplanted vegetation (applied nucleation, windthrow guard, applied nucleation with windthrow guard) as the transplanted vegetation was in the middle of these plots. This was done to avoid preferential selection of areas of the plot. There was one exception to this quadrat placement, in one of the W-shaped transect plots (385). This was to adjust to a ruffed grouse (*Bonasa umbellus*) nest constructed in the middle of the plot. To minimize disturbance to the mother incubating her eggs, the vegetation surveys took place in the bottom right corner of the plot.

Relative to the pilot studies, there was now a smaller area to be surveyed. This meant that additional plots outside of the W-shaped transect could be surveyed in 2024. They were selected

using a random number generator to generate the three-digit code associated with a plot. A random number generator was used to prevent bias that could occur by human selection of plots to survey. If a W-shaped transect plot ID was generated, it was not resurveyed, and a new plot ID number was generated instead.

Although the size of the surveyed area changed, the actual vegetation surveying strategy did not. No matter the size of the plot, vegetation was identified to the species level (except for grasses, sedges and mosses, which were identified to the family level) and the percentage of the ground each species covered was grouped into Daubenmire cover classes (Daubenmire, 1959) and recorded**.** Transplanted vegetation was not included in the vegetation surveys.



Figure 4. a) Image of the 1-metre square quadrat used for vegetation survey b) Image of the 1-metre square quadrat used for vegetation surveys.

2.1.6.3 Arthropod (Insect and Spider) Trapping and Identification

Arthropod trapping occurred twice a year in 2023 and 2024. Fisher® 4oz Polypropylene Specimen Containers were used as pitfall traps, and Blue Vane Pollinator traps were used as pollinator traps (Figures 5 a and b). The pitfall traps were filled a quarter full of 100% ethanol in 2023 and 70% ethanol in 2024. The change in ethanol percentage occurred due to bycatch slugs dissolving/rotting in the higher ethanol percentage. The pitfall traps were dug to be the approximate size and shape of the container and were left in the ground with the lid on for at least seven days. This was done to eliminate bias from the arthropods exploring something new

in their habitat. The pollinator traps were filled with water, and approximately three drops of unscented detergent were added to break the surface tension. When the pollinator traps were set out, the lids were removed from the pitfall traps, and both traps were left this way for approximately 48 hours and then collected. The insects and spiders were placed in labelled jars with 70% ethanol to preserve them. They were identified to the family level with a Leica S6E Stereo Microscope (Figures 5 c and d).

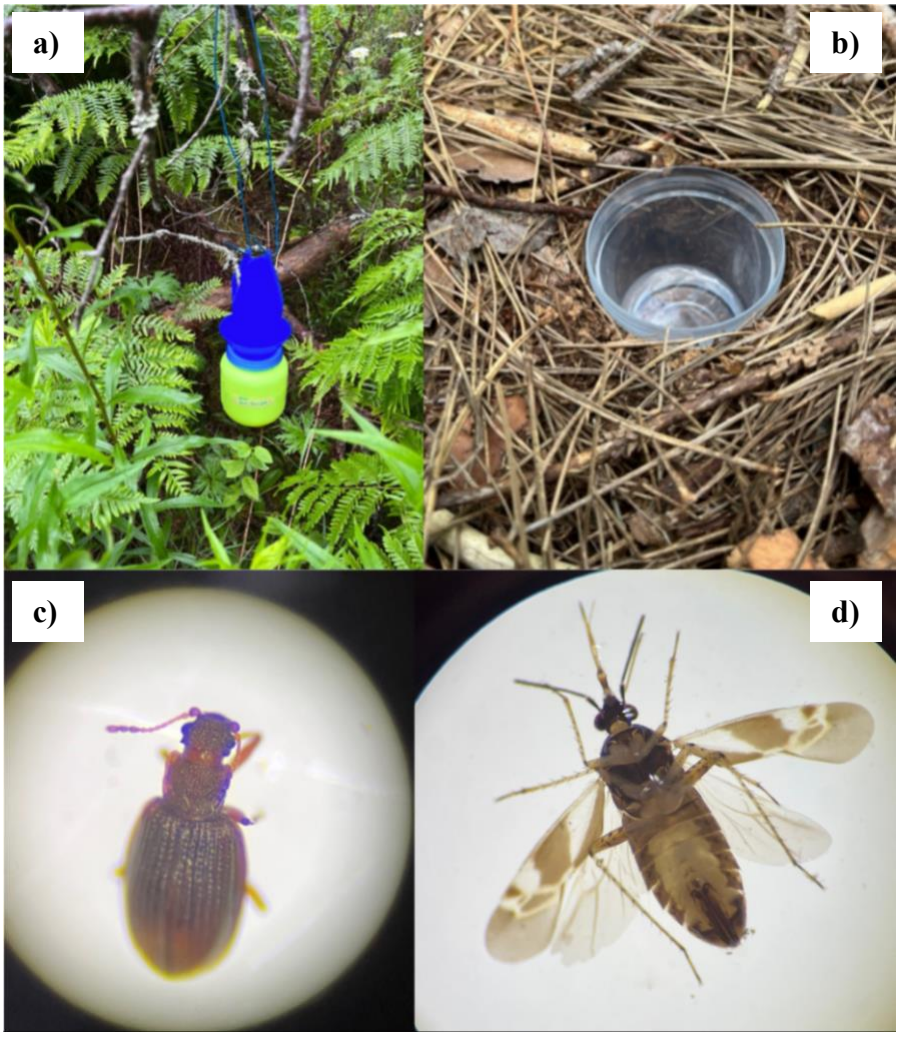


Figure 5. Images of a) a pollinator trap b) a pitfall trap c) a minute brown scavenger beetle (family Latridiidae) under the microscope d) a plant bug (family Miridae) under the microscope.

2.1.6.4 Soil Sampling and Testing

Soil sampling was conducted annually from 2021 to 2024. In 2021, samples were taken randomly by the original researcher. In 2022, samples were taken in the original W-shaped

transect by the original researcher. In 2023 and 2024, I collected samples in the amended W-shaped transect.

Although the exact plots which were sampled differed over the years, the sample collection strategy remained consistent. A 10-centimetre-deep soil sample was taken in roughly the middle of each plot and stored in a labelled paper bag and stored in a refrigerator between sample collection and sample testing. I performed all the soil testing. The samples were tested for pH using a Hanna HI-98129 pH meter and for nitrate-nitrogen (ppm), phosphorus (ppm), and potassium (ppm) using a LaMotte SMART 3 Soil Colorimeter according to the manufacturer's instructions (LaMotte Company, 2022, Figure 6). Nitrate-nitrogen was converted to nitrate by multiplying the nitrate-nitrogen concentration by 4.4, according to the manufacturer's instructions (LaMotte Company, 2022).



Figure 6. Testing three soil samples for phosphorus concentrations (ppm) with the LaMotte soil testing kit.

2.1.7 Statistical Analysis

I used R statistical software version 4.5.0 (R Core Team, 2025).

2.1.7.1 Mortality

A Cochran's Q test (Cochran, 1950) was run on the mortality data (package RVAideMemoire, Herve, 2025). The mortality of the transplants was assessed with separate Q tests for the applied nucleation and applied nucleation with windthrow guard treatment types. A post hoc pairwise McNemar test (package rcompanion, Mangiafico, 2025) with a Bonferroni correction for multiple comparisons was then performed on statistically significant results. The Bonferroni correction controls for false positives when performing multiple statistical tests by adjusting probability values (R. A. Armstrong, 2014).

2.1.7.2 Vegetation

Generalized linear mixed models (Bolker et al., 2009) (GLMMs) (packages lme4, glmmTMB, Bates et al., 2025; Brooks et al., 2025) were used to assess differences in species richness, Simpson Diversity (Simpson, 1949) and Hill-Shannon (Hill, 1973) diversity (package hillR, Li, 2023) of native and non-native vegetation between the different treatment types. Due to the change in vegetation survey size, data from 2024 was assessed in separate models from the data from 2022 and 2023. Additionally, separate models were built for native and non-native vegetation metrics, allowing for the assessment of impacts on both types of vegetation metrics to be evaluated separately.

Models with 2022 and 2023 data were assessed for best fit with random effects of year, plot ID, and both plot ID and year. Using these random effects allows us to account for the fact that data from the same plot that is resampled, and data from the same year is more likely to be similar than data from different plots and/or years. Models with the 2024 data used the random effect of plot ID, as there was only 1 year of data. All models had the fixed effect of treatment type. The fit of different appropriate distributions was assessed with the fitdistrplus package (Delignette-Muller & Dutang, 2015). The most parsimonious model was selected using AIC and BIC values. Model fit was assessed with quantile-quantile and residual plots (packages qqplotr, DHARMa, ggplot2; Almeida et al., 2023; Hartig et al., 2024; Wickham et al., 2025). When the output of the GLMM showed statistically significant differences between vegetation diversity metrics in the reference and treatment plot types, post-hoc comparisons using Tukey-adjusted estimated marginal means were performed (package emmeans, Lenth et al., 2025).

2.1.7.3 Arthropods

Two-way mixed ANOVAs (Guarino et al., 2009) were used to assess the impact of treatment type and year on species richness, Shannon-Weiner diversity and Simpson diversity values. A separate two-way mixed ANOVA (package rstatix, Kassambara, 2023) was used to assess impacts on each diversity and richness metric individually. Before the ANOVA test was run, the data was tested to ensure it met the assumptions of the test. The normality of each metric was assessed using a normal QQ plot (package ggpubr, Kassambara, 2023), the homogeneity of variance was assessed with a Levene test (package rstatix, Kassambara, 2023) and the homogeneity of covariance (package rstatix, Kassambara, 2023) was assessed with Box's test of equality and covariance matrices. Outliers were also identified but were not removed. The ANOVA table was extracted, and the Greenhouse-Geisser sphericity correction was automatically applied to any factors violating the assumption of sphericity. Sphericity assumes that the variance between each paired group is equal, and when this assumption is violated, the chance of false positives increases (Salkind, 2010). The Greenhouse-Geisser sphericity correction corrects for this issue (Salkind, 2010)

2.1.7.4 Soil

A non-parametric Kruskal-Wallis test (Kruskal & Wallis, 1952) was run for the within-year comparisons of treatment type on each of the soil parameters (pH, NO_3, P, K). Separate within-year comparisons were performed due to the changes in which plots were sampled, as well as the addition and sampling of the unthinned plot types. For results of Kruskal-Wallis tests that were statistically significant, a post-hoc Dunn test (package rstatix, Kassambara, 2023) was run with a Benjamini-Hochberg adjustment for multiple comparisons.

2.2 Results

2.2.1 Mortality

In 2024, 53/64 transplants in the applied nucleation treatment were still alive. Of the surviving transplants, 2/53 were beaked hazel shrubs and 51/53 were yellow birch trees. In 2024, 57/64 of the applied nucleation with windthrow guard transplants were still alive. Of the surviving transplants, 2/57 were beaked hazel shrubs and 55/57 were yellow birch trees.

The mortality of transplants in the applied nucleation plot type was statistically significantly different over the 3 years of data collection (2022, 2023 and 2024), as assessed by the Cochran's Q test ($Q_2 = 11.46$, $p = 0.003$). There were statistically significant differences in transplant mortality in the applied nucleation treatment type between 2022 and 2024 ($p = 0.04$) as assessed by pairwise McNemar tests with a Bonferroni correction (Figure 7).

The mortality of transplants in the applied nucleation with windthrow guard plot type was also statistically significantly different over the 3 years of data collection, as assessed by Cochran's Q test ($Q_2 = 6.3$, $p = 0.04$). However, there were no statistically significant differences in transplant mortality between the applied nucleation with windthrow guard treatment types in any of the paired years ($p > 0.05$) as assessed by pairwise McNemar tests with a Bonferroni correction (Figure 7).

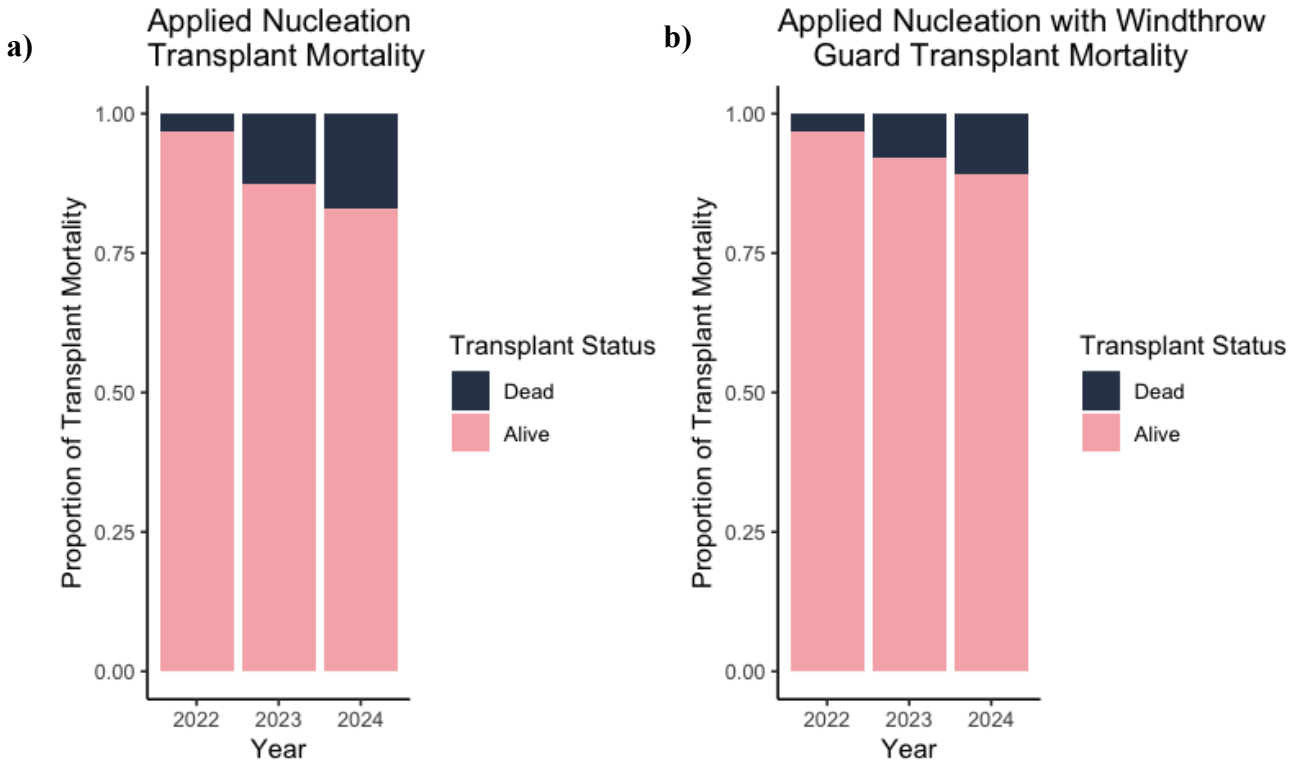


Figure 7. Proportion graphs of transplant mortality of the a) applied nucleation and b) applied nucleation with windthrow guard plot types.

2.1.2 Vegetation

A total of 129 different vegetation species (90 native and 39 non-native) were identified during this experiment. Native and non-native species differentiation was determined using information in multiple field guides (Chambers et al., 1996; Cobb et al., 2005; Newcomb, 1977; Petrides, 1972). The most commonly observed species were wild red raspberry (*Rubus idaeus*), present in 208/344 of vegetation surveys, followed by mayflowers (*Epigaea repens*), present in 204/344 of the vegetation surveys. Both of these species are native. The most commonly observed non-native species were orange hawkweed (*Hieracium aurantiacum***)**, present in 70/344 of the vegetation surveys, and heath/common speedwell (*Veronica officinalis*), present in 65/344 vegetation surveys.

The information for each GLMM run, including the distribution and the random effect(s) is in Table 2.

Table 2. GLMM summary tables for the vegetation metrics, showing the dependent variable, the years the data is from and the model that fits each set of data the best.

Dependent variable/Indicator	Year(s) data is from	Distribution	Fixed Effect	Random Effect(s)
Native Simpson Diversity	2022 & 2023	Ordered Beta	Treatment Type	Year
Nonnative Simpson Diversity	2022 & 2023	Ordered Beta	Treatment Type	Year
Native Simpson Diversity	2024	Ordered Beta	Treatment Type	Plot ID
Nonnative Simpson Diversity	2024	Ordered Beta	Treatment Type	Plot ID
Native Species Richness	2022 & 2023	Poisson	Treatment Type	Year & Plot ID
Nonnative Species Richness	2022 & 2023	Poisson	Treatment Type	Year & Plot ID

Native Species Richness	2024	Poisson	Treatment Type	Year & Plot ID
Nonnative Species Richness	2024	Poisson	Treatment Type	Year & Plot ID
Native Hill-Shannon Diversity	2022 & 2023	Gamma	Treatment Type	Year & Plot ID
Nonnative Hill-Shannon Diversity	2022 & 2023	Log Normal	Treatment Type	Year
Native Hill-Shannon Diversity	2024	Normal	Treatment Type	Plot ID
Nonnative Hill-Shannon Diversity	2024	Gamma	Treatment Type	Plot ID

The species richness of native and non-native vegetation in 2022 and 2023 was not statistically significantly different in any of the treatment plots when compared to the unthinned plots ($p > 0.05$). The species richness of native vegetation in 2024 was also not statistically significantly different than the unthinned plots ($p > 0.05$). However, there was a statistically significant difference in the species richness of nonnative vegetation in the applied nucleation with windthrow guard, control and windthrow guard plots when compared to the unthinned plots ($p > 0.05$, Table 3).

Table 3. GLMM outputs of the fixed effects from the species richness of nonnative vegetation from 2024. The unthinned plot type was used as the reference. Significant p-values are bolded. Treatment Types: AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

Treatment Type	Estimate	Std. Error	z value	p-value
AN	1.3	0.67	1.9	0.05
ANWG	1.7	0.66	2.6	**0.01**
C	1.6	0.67	2.3	**0.02**
WG	1.7	0.65	2.6	**0.01**

Although species richness was higher in the applied nucleation with windthrow guard, control and windthrow guard treatment types when compared to the unthinned treatment type, these differences are not statistically significant as assessed with post-hoc pairwise comparisons of Tukey-adjusted estimated marginal means ($p > 0.05$, Table 4).

Table 4. Post-hoc Tukey-adjusted estimated marginal means results of the GLMM assessing differences in non-native vegetation species richness in 2024. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

Treatment Type Comparisons	Estimate (Log)	Estimate (Exponentiated)	Standard Error	Z ratio	p-value
UN - AN	-1.3	0.27	0.67	-1.90	0.30
UN - ANWG	-1.7	0.18	0.66	-2.60	0.08
UN - C	-1.6	0.21	0.67	-2.30	0.13
UN - WG	-1.7	0.19	0.65	-2.60	0.08
AN - ANWG	-0.41	0.66	0.40	-1.00	0.84
AN - C	-0.27	0.76	0.41	-0.66	0.96
AN - WG	-0.38	0.69	0.38	-0.98	0.86
ANWG - C	0.14	1.20	0.40	0.34	1.00
ANWG - WG	0.04	1.00	0.34	0.10	1.00
C - WG	-0.10	0.91	0.39	-0.26	1.00

The Simpson diversity of native and nonnative vegetation in 2022 and 2023 was not statistically significantly different in any of the treatment plots when compared to the reference plots ($p > 0.05$). The Simpson diversity of native and nonnative vegetation in 2024 was not statistically significantly different in any of the treatment plots when compared to the reference plots ($p > 0.05$).

The Hill-Shannon diversity of native and nonnative vegetation in 2022 and 2023 was not statistically significantly different in any of the treatment plots when compared to the reference plots ($p > 0.05$). The Hill-Shannon diversity of native vegetation in 2024 was not statistically significantly different in any of the treatment plots when compared to the reference plots ($p > 0.05$). The Hill-Shannon diversity of nonnative vegetation was not statistically significantly different in the applied nucleation plots when compared to the reference plots ($p > 0.05$).

However, the Hill-Shannon diversity of nonnative vegetation was statistically significantly different in the control, windthrow guard and applied nucleation with windthrow guard plot types when compared to the reference state ($p < 0.05$, Table 5).

Table 5. GLMM outputs of the fixed effects from the Hill-Shannon diversity of nonnative vegetation from 2024. The unthinned plot type was used as the reference. Significant p-values are bolded. Treatment Types: AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

Treatment Type	Estimate	Std. Error	z value	p-value
AN	0.48	0.27	1.7	0.08
ANWG	0.64	0.27	0.3	**0.02**
C	0.58	0.28	2.1	**0.03**
WG	0.70	0.27	2.6	**0.01**

Although species richness was higher in the control, windthrow guard and applied nucleation with windthrow guard treatment types when compared to the unthinned treatment type, these differences are not statistically significant as assessed with post-hoc pairwise comparisons of Tukey-adjusted estimated marginal means ($p > 0.05$, Table 6).

Table 6. Post-hoc Tukey-adjusted estimated marginal means results of the GLMM assessing differences in non-native vegetation Hill-Shannon diversity in 2024. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

Treatment Type Comparisons	Estimate (Log)	Estimate (Exponentiated)	Standard Error	Z ratio	p-value
UN - AN	-0.48	0.62	0.28	-1.80	0.41
UN - ANWG	-0.64	0.53	0.28	-2.30	0.15
UN - C	-0.58	0.56	0.29	-2.10	0.23
UN - WG	-0.70	0.50	0.27	-2.60	0.08
AN - ANWG	-0.16	0.86	0.20	-0.78	0.94
AN - C	-0.10	0.90	0.21	-0.51	0.99
AN - WG	-0.22	0.81	0.19	-1.10	0.79
ANWG - C	0.052	1.10	0.21	0.25	1.00
ANWG - WG	-0.061	0.94	0.18	-0.33	1.00
C - WG	-0.11	0.90	0.20	-0.56	0.99

2.1.3 Arthropods

A total of 545 individuals from 111 different taxa were trapped and identified throughout this project. The most common arthropod family observed was hoverflies (Syriphidae), with 56 individuals found in 36 traps. Ants (Formicidae) closely followed, with 53 individuals found in 20 traps. Slugs were not included in this data or the modelling, but some were identified, and all were nonnative.

There were no statistically significant two-way interactions between treatment type and sampling time on arthropod species richness ($F_{(12,30)} = 1.011$, $p > 0.05$). Multiple pairwise t-tests with a Bonferroni correction for multiple comparisons revealed that neither the main effect of sampling time nor the main effect of treatment type had a statistically significant impact on arthropod species richness ($p > 0.05$, Figure 8).

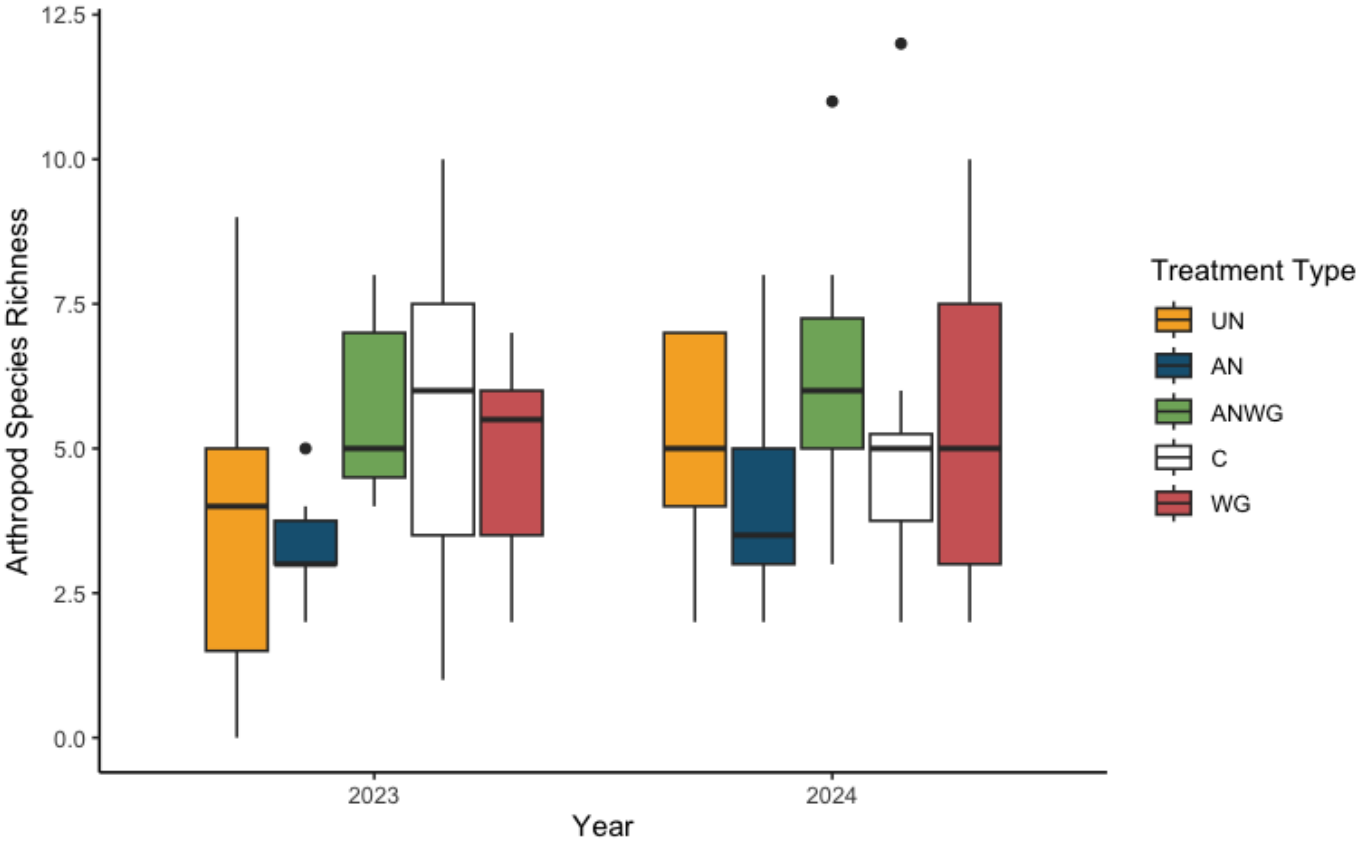


Figure 8. Box Plot of Arthropod Species Richness per year separated by treatment type. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

There were no statistically significant two-way interactions between sampling time and treatment type on arthropod Simpson diversity ($F_{(7.58,18.96)} = 0.61$, $p > 0.05$). Multiple pairwise t-tests with a Bonferroni correction for multiple comparisons showed that the main effects of treatment type and sampling time on arthropod Simpson diversity were not significant ($p > 0.05$, Figure 9).

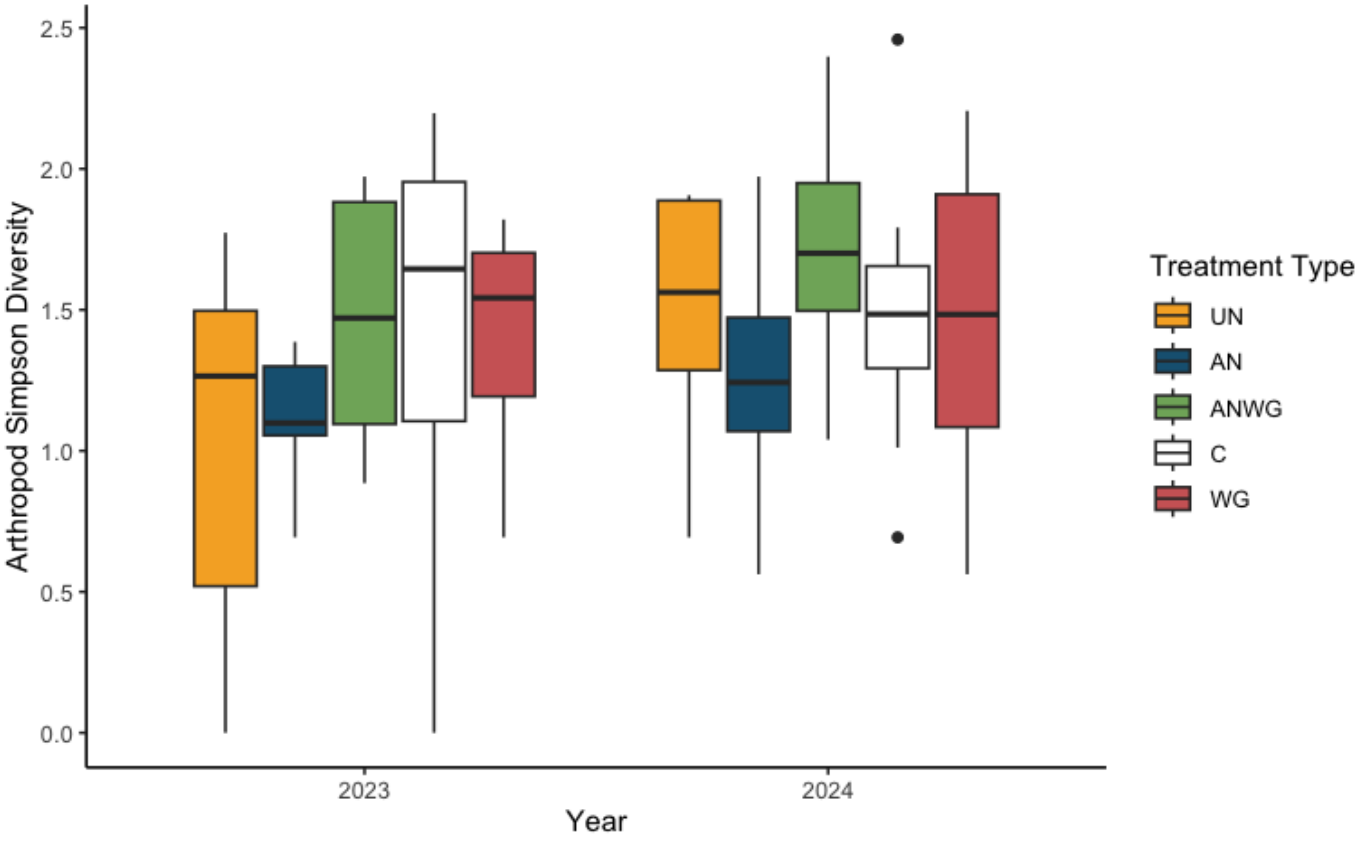


Figure 9. Box Plot of Arthropod Simpson Diversity per year separated by treatment type. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

There were no statistically significant two-way interactions between treatment type and sampling time on arthropod Shannon-Weiner diversity ($F_{(12, 30)} = 0.80$, $p > 0.05$). Multiple pairwise t-tests with a Bonferroni correction for multiple comparisons showed that the main effects of treatment type and sampling time on arthropod Shannon-Weiner diversity were not significant ($p > 0.05$, Figure 10).

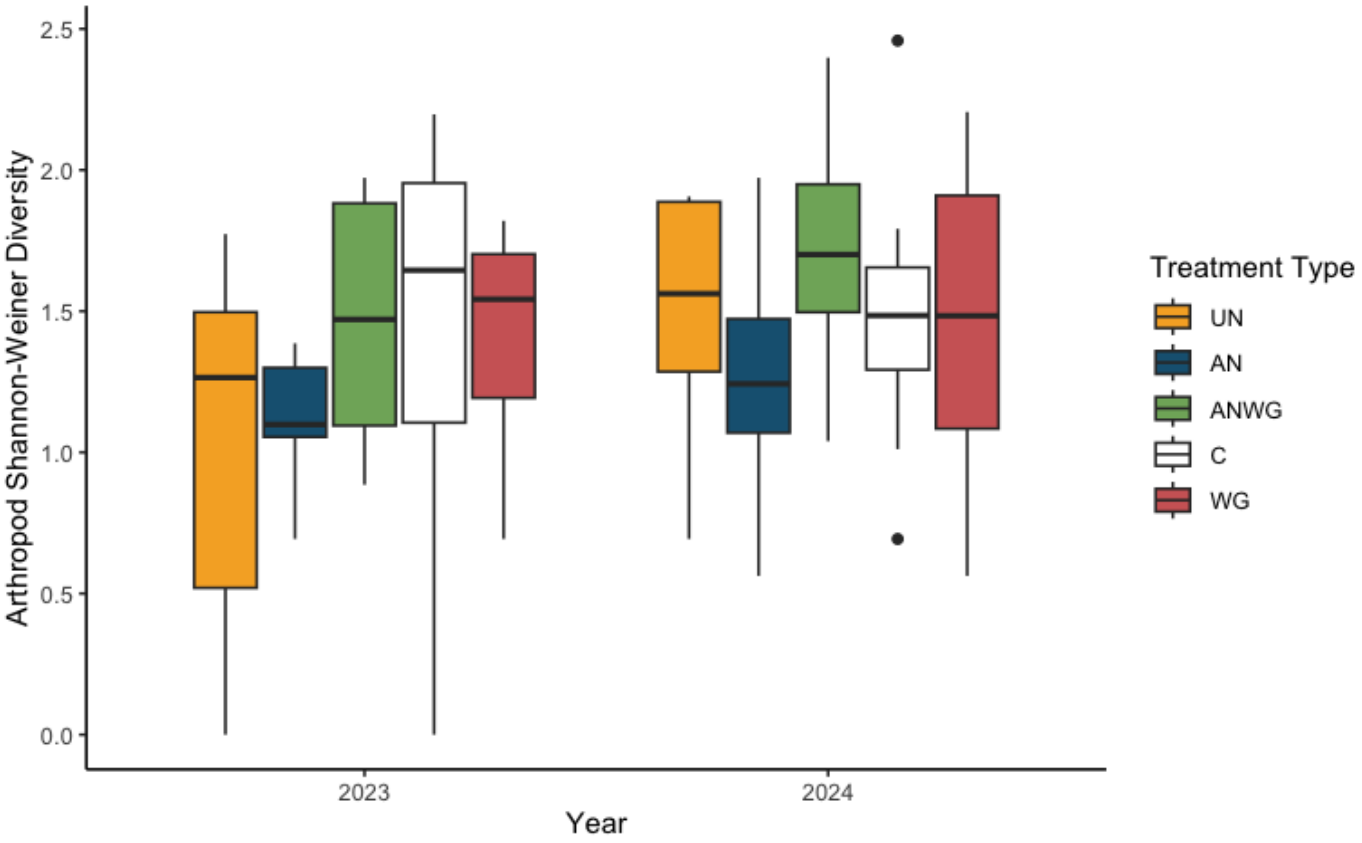


Figure 10. Box Plot of Arthropod Shannon-Weiner Diversity per year separated by treatment type. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

2.1.4 Soil

In 2021, there were no statistically significant differences between pH values and potassium concentrations (ppm) between the different treatment types ($p > 0.05$) as assessed by the Kruskal-Wallis test. Every soil sample tested in 2021 had a nitrite concentration of zero ppm. Due to this, the Kruskal-Wallis test was unable to report a p-value. However, since every sample had the same (lack of) nitrite, it can be assumed that treatment type did not have a statistically significant impact on nitrite concentrations in 2021. There were statistically significant differences in phosphorus concentrations between different treatment types in 2021 ($p = 0.03$) as assessed by the Kruskal-Wallis test. There are statistically significant differences in phosphorus concentrations between the control and windthrow guard treatments ($p = 0.04$, Table 7) as

assessed by the post-hoc Dunn's test with a Benjamini-Hochberg adjustment for multiple comparisons.

Table 7. Results of post-hoc Dunn's Test with a Benjamini-Hochberg adjustment for the differences in phosphorus (ppm) between the different treatment types in 2021. Significant p-values are bolded. Treatment Types: UN – unthinned, AN – applied nucleation, ANWG – applied nucleation with windthrow guard, C – control, WG – windthrow guard.

Treatment Type Comparisons	z statistic	p-value (adjusted)
AN – ANWG	1.23	0.46
AN – C	1.92	0.27
AN – WG	0.57	0.57
ANWG – C	1.20	0.46
ANWG – WG	-1.28	0.46
C – WG	-2.71	**0.04**

In 2022 and 2023, there were no statistically significant differences in pH, nitrate, phosphorus or potassium values among the different treatment types as assessed by the Kruskal-Wallis test ($p > 0.05$).

In 2024, there were no statistically significant differences in pH, nitrate, or potassium values among different plot types, as assessed by the Kruskal-Wallis test ($p > 0.05$). There was a statistically significant difference in phosphorus values among different treatment types, as assessed using the Kruskal-Wallis test ($p = 0.030$). However, post-hoc pairwise comparisons with Dunn's test using the Benjamini-Hochberg correction for multiple comparisons did not reveal any significant differences in phosphorus levels between treatment groups ($p \geq 0.18$).

2.3 Discussion

2.3.1 Mortality

2.3.1.1 High transplant Survival Rate

A high transplant survival rate can indicate an acceptable restoration outcome (Grossnickle, 2012; Preece et al., 2023), especially as forest restoration transplants tend to have high mortality rates (Marshall et al., 2023). Other restoration projects involving transplanting saw a survival rate of 70% (Balestri et al., 1998) or over 80% (Holl et al., 2011) as an acceptable rate of

transplant survival. Overall, the applied nucleation transplants had a high rate of survival, with 83% of the transplants in the applied nucleation treatment type surviving and 89% of the transplants in the applied nucleation with windthrow guard treatment type surviving. Although longer-term data are needed, this may suggest that the transplanting technique used, as well as the species used, are both suited for restoring the understorey habitat of a red pine plantation, as the high survival rate is an acceptable outcome for this project (Phoenix, 2022).

There are multiple benefits to a high transplant survival rate, including the reduction or elimination of additional transplanting efforts (Preece et al., 2023). This prevents the spending of additional time, money and resources on replanting efforts further down the line (Preece et al., 2023). Additionally, a high transplant survival rate can help to increase species diversity and abundance (Marshall et al., 2023). Factors such as root and shoot growth and development, as well as stress resistance impact transplant mortality (Grossnickle, 2012). The transplants in this project were sourced from nearby the research site, meaning they were already adjusted to the climactic and soil conditions of the area. This likely resulted in lower levels of transplant stress.

Generally, transplanted trees exhibit higher survival rates, while transplanted shrubs tend to have lower survival rates (Reis et al., 2025). The survival and growth of applied nucleation transplants seem to be trait-specific (e.g. slow vs. fast-growing) and species-specific (Manhães et al., 2024). Of the 110 surviving transplants (in 2024) across both the applied nucleation and applied nucleation with windthrow guard treatment types, 106 of them were yellow birch trees, and 4 were beaked hazel shrubs. Although we do not know the initial number of transplants for each species, a large difference in transplant mortality between the two species may inform future restoration decisions. However, further research would be necessary to determine the exact transplant survival rates of both species.

2.3.1.2 Windthrow Guard Benefits

The difference in transplant mortality between 2022 and 2024 was statistically significantly different for the applied nucleation treatment type, but not for the applied nucleation with windthrow guard treatment type. This may suggest that the time since the transplant occurred has a lesser impact on the applied nucleation with windthrow guard transplants. Although other

factors may be involved, the use of windthrow guards could be a contributing factor in reducing transplant death rates.

The use of windthrow guards around applied nuclei could also increase species richness and community composition, by adding both woody species and forbs (herbaceous flowering plants that are not grasses, sedges or rushes) or grasses to the area simultaneously. Different understorey groups (woody species, forbs and grasses) have differing life history traits and perform various functions within the community (Duffy et al., 2024). For example, some forb species fix nitrogen, provide habitat for pollinators, and increase forage value for herbivorous species (Vera-Velez & Lamb, 2022). The introduction of both woody species and forbs simultaneously could increase the community's functional diversity, which can assist with the recovery of ecosystem function (Manhães et al., 2024).

Additional research into species-specific responses to windthrow guards, as well as testing the success of using different species or groups with particular traits may help to identify the combination(s) of woody and herbaceous species with optimal traits and high survival rates. Further testing of different densities and numbers of windthrow guard planting, as well as various combinations of numbers and species of woody plants, would also enable the use of the most efficient and effective restoration treatments possible. For example, Koutzoukis (2023) found that the "optimal" distance of forb survival planted near sagebrush nurse plants was species-specific. Although that study addresses the possible impacts of shrubs on forbs and grass, and we are examining the effects of forbs on shrubs and trees, there are likely still species-specific optimal ranges for survival due to differing species-specific or functional attributes.

2.3.2 Vegetation

2.3.2.1 Increased Nonnative Vegetation Community Metrics in 2024

The only statistically significant difference in vegetation metrics between the treatment and reference plot types was the species richness and Hill-Shannon diversity of non-native vegetation in 2024. Although the post-hoc tests did not find any statistically significant differences between the paired groups of treatment types, the nonnative species richness and Hill-Shannon diversity were different in the unthinned/reference plot types when compared to the other treatment types.

In general, four years after restoration treatments were implemented, the species richness and Hill-Shannon diversity of non-native vegetation in the understorey had increased compared to the reference state. Non-native vegetation is often highly competitive, tolerant to disturbance, and has higher abundance in more open forest areas, when compared to mature forest areas with more canopy closure (Jenkins et al., 2025). This likely explains the higher presence of non-native vegetation in the restored (thinned and more disturbed) plots than the reference (unthinned and less disturbed) plots.

2.3.2.2 Lack of Change in Vegetation Community Metrics in Early Years Post-Restoration

The recovery of native vegetation after restoration tends to be a longer-term process, no matter the ecosystem or restoration treatment(s) used. One year after tree thinning occurred, Wen et al. (2025), found there were no changes to understorey functional groups in a masson pine plantation. Seven years after applied nucleation occurred in tropical forests, Piaia et al. (2020), found that some indicators of restoration recovery, such as forb richness and abundance, had not yet reached reference levels. For the first nine years after nurse plant establishment in the desert, Abella et al. (2023), found that non-native vegetation cover was higher than native vegetation cover underneath the nurse plants. However, years nine to twelve experienced greater native vegetation cover underneath the nurse plants (Abella et al., 2023). Although there has not been significant changes to native vegetation metrics in the first few years after restoration occurred, there may be changes in the future.

Further surveying will provide more information about the recovery of native and non-native vegetation in future years. If non-native vegetation metrics continue to increase, or native vegetation metrics do not, additional herbicide applications or other invasive vegetation removal strategies may need to be implemented.

2.3.2.3 Reproduction and Habitat Characteristics of the Most Commonly Observed Native and Nonnative Vegetation Species

Although there were only statistically significant differences in nonnative vegetation metrics, the two most commonly observed vegetation species were native. Wild red raspberry was present in 61% of the surveyed plots, and mayflowers were present in 59% of the surveyed plots.

Additionally, 70% of the observed species were native. The most commonly observed non-native species was orange hawkweed, present in 20% of the surveyed plots followed by heath/common speedwell, present in 19% of the surveyed plots. Due to the high presence of these plant species, it can be beneficial to explore the traits and life history strategies of these species, as they can provide information as to the current state of the research site.

Wild red raspberry is a native perennial deciduous shrub which generally reproduces and spreads through rhizomes (Oleskevich et al., 1996). The flowers are commonly pollinated by hoverflies and sweat bees (Subgenus Dialictus) (Hansen & Osgood, 1983). Wild red raspberry plants are an early successional species (Lautenschlager, 1999). It can tolerate a wide range of soil conditions and is often found in disturbed and open forest sites (Oleskevich et al., 1996), which can include a restored red pine plantation.

Canada mayflower is a native perennial flowering plant (Barrett & Helenurm, 1987). Mayflowers grow well in acidic soil (Coville, 1911), and as previously discussed, pine plantations have acidic soil (Turk et al., 2008). Mayflowers are a common forest understorey herb, with self-incompatible, insect-pollinated flowers which are often pollinated by solitary bees and hoverflies (Barrett & Helenurm, 1987). Due to this, there may be a relation to the high number of mayflowers and hoverflies present at the site. Ants and possibly birds disperse mayflower seeds (Clay, 1983), so the high presence of ants at the site may also be related to the high presence of mayflowers.

Orange hawkweed is an invasive perennial flowering plant from central Europe (Loomis & Fishman, 2009). Orange hawkweed can reproduce clonally, through rhizomes, as well as through apomictic seeds (Loomis & Fishman, 2009). The seeds can be dispersed by wind or animals (Guetling et al., 2023). It can survive in a wide range of habitats (Guetling et al., 2023). Orange hawkweed has allelopathic pollen, meaning that its pollen can inhibit the sexual reproduction of other plants by releasing toxins (Murphy, 2001). The significant presence of orange hawkweed at the study site may be cause for concern; further monitoring of the population should occur, and removal strategies may be an option.

Common speedwell is an invasive flowering perennial plant from Europe and western Asia (MacQuarrie & Lacroix, 2003). Common speedwell tolerates shaded environments (Dale & Causton, 1992). Common speedwell reproduces both sexually through seeds and asexually through rhizomes (Government of Ontario, 2023). Common speedwell grows best in slightly acidic to neutral soil (Falkengren-Grerup, 1998); therefore the significant presence of this species may indicate that the soil pH is becoming less acidic.

The high presence of these plants indicates that the research site is currently in a disturbed, open canopy, early successional state, that is currently not suited for the support of many understorey specialist species, or many woody species, as only one of the most common species is a shrub. The high presence of vegetation with insect pollinated flowers indicates that there is a fairly abundant pollinator community capable of pollinating these species flowers.

2.3.3 Arthropods

2.3.3.1 Lack of Arthropod Community Change in Early Years Post-Restoration

As it is still early in the successional trajectory following restoration treatments, neither treatment type nor year had a statistically significant impact on arthropod species richness, Simpson diversity, or Shannon-Weiner diversity. For example, Tudor et al (2023) found that it took ten years after restoration for insect community composition to reach reference levels in restored post-mining forests. Additionally, the long-term impacts of restoration on terrestrial invertebrates are not well studied (Borges et al., 2021). However, the use of terrestrial invertebrates as ecological indicators for restoration success has increased in recent years (Borges et al., 2021). Responses of invertebrates to restoration efforts such as herbicide application appear to be taxon-specific (Briggs et al., 2024). For example, Ulyshen et al. (2022) found that the responses of bees and butterflies to invasive shrub removal differed, with bee communities but not butterfly communities recovering to reference levels over 13 years.

Insect succession or recovery is impacted by the structural development of forests, as the age of the trees, complexity and diversity of the understorey vegetation, and availability of resources for more specialized insects (such as dead wood) changes over time (D. Allison et al., 2023). As forest succession occurs, a more diverse habitat can support a more diverse

insect/arthropod population (D. Allison et al., 2023). For example, forest attributes such as the availability of floral resources (Ulyshen et al., 2022), tree composition and age (Ulyshen et al., 2024) impact the composition of a forest pollinator community. Pollinator community attributes, such as species richness, are increased by both understorey vegetation diversity and the presence of nurse plants (Losapio et al., 2021). As vegetation diversity and habitat complexity increase over time post-restoration, the arthropod community can change in turn to populate new habitat niches as they become available.

2.3.3.2 Ecosystem Services Provided by Most Commonly Observed Arthropod Families

Considering the long-term trajectory of this project, it is beneficial to explore the early outcome that ants (Formiculidae) and hoverflies (Syriphidae) are the most observed families. Both insect families provide ecosystem services and can be sensitive to environmental change (Tiede et al., 2017; Vezsenyi et al., 2021). Ants perform a variety of functions which can assist the goals of restoration projects (De Almeida et al., 2024). Ants (and other social insects) provide essential ecosystem services and have mutualistic relationships with many types of organisms (Elizalde et al., 2020). They act as ecosystem engineers by altering the surrounding area to build their nests (Elizalde et al., 2020). Ants can also act as seed dispersers, which enables the establishment of pioneer plants (Fernandes et al., 2020). However, the ecological impacts of ants are not all positive. For example, Çakır (2019) found that the density of soil microarthropods was negatively impacted by the presence of wood ants in a semi-arid pine stand.

Hoverflies are an abundant and diverse family (Vezsenyi et al., 2021), found on every continent except for Antarctica (Reynolds et al., 2024). Hoverflies provide important ecosystem services such as pollination (T. Doyle et al., 2020). Hoverflies can travel long distances (T. Doyle et al., 2020) and some North American species are likely migratory (Clem et al., 2022; Reynolds et al., 2024). The long distances travelled by hoverflies allow them to increase the genetic diversity of plant populations, as they can move pollen over large areas (T. Doyle et al., 2020). Some hoverfly species may also act as biocontrol agents for invasive hawkweed (Hieracium) plants, as their larvae specifically target and eat hawkweed plants (Grosskopf, 2005). As the most abundant non-native species is orange hawkweed, the high presence of hoverflies may reduce the abundance of this species.

Further experiments testing the amount, types of and duration of arthropod trapping could result in the creation of a standardized protocol of arthropod trapping. This would allow for more direct comparison between restoration projects and provide more data on ecological recovery. There have been calls to increase the use of terrestrial insects in ecological monitoring, as they have unique characteristics which allow them to act as good proxies for the environment around them and are also fairly inexpensive to sample (Chowdhury et al., 2023).

2.3.4 Soil

2.3.4.1 Lack of Change in Soil Chemical Parameters in the Early Years Post-Restoration

The overall lack of statistically significant results of most of the chemical soil parameters (pH, nitrate, and potassium in 2021-2024, and phosphorus in 2022 and 2023) in the early years after forest restoration occurred is to be expected, due to the long recovery timelines of chemical soil parameters (Cardoso et al., 2013). In general, statistically difference results in soil parameters often do not occur until over 15 years post-restoration (Gatica-Saavedra et al., 2023). The soil parameters associated with coniferous plantations can persist for decades after restoration of the plantation occurs (Cerli et al., 2006).

Although the phosphorus concentrations from different treatment types in 2021 and 2024 were significantly different across the different treatment types, only the phosphorus concentrations in 2021 showed significant differences between any of the treatment types. There were statistically significant differences in phosphorus concentrations between the control and windthrow guard treatment types in 2021. Due to the long recovery timeline of soil phosphorus (Gatica-Saavedra et al., 2023), this significant difference is likely attributed to spatial variation in soil phosphorus concentrations within the site itself. In 2021, soil samples were taken in only two of the blocks, whereas from 2022 onward, samples were collected along the W-shaped transect to reduce spatial impacts.

Soil phosphorus is influenced by various factors including native plant diversity, litter quality and overstorey composition (Prober & Wiehl, 2012; X. Zhu et al., 2021). Total phosphorus present in soil has been found to increase as succession occurs from a coniferous

plantation to a deciduous forest due to the higher nutrient concentrations and faster decomposition time of deciduous leaves when compared to coniferous needles (X. Zhu et al., 2021). So, as succession continues to occur, the phosphorus at the site may increase. For example, six years after forest restoration occurred, Zhong et al. (2021), found that soil phosphorus levels had changed to be slightly closer to reference levels.

My research was tasked with establishing a single soil sampling strategy. That has been done and used in 2023 and 2024; the purpose was to learn from the 2021-2022 sampling options and settle on a strategy that means that all soil data from 2023 onwards can be compared between years.

Ideally, resampling and testing the same plots every year or every few years would eventually build up a robust data set that would allow for the study of long-term effects of these understorey restoration techniques on chemical soil parameters. As previously stated, chemical parameters of soil change on a comparatively long-time scale, so four years is not a lot of time. Additionally, testing these techniques in different types of plantation forests, which are different ages, would allow for the comparison of how effective these restoration techniques are for other species and variously aged plantations. This information would provide practitioners with valuable knowledge.

2.4 Project Limitations and Challenges

It is also important to discuss the project challenges and limitations, as well as how these issues were solved. To begin, there was only one researcher collecting data at a time, with no assistants. As was only so much a single person could do in a day, which limited both field and lab work activities.

The first challenge with fieldwork was the fading of the plot ID numbers, which were written with permanent markers on flagging tape, and attached to one of the pigtail markers that delineated the four corners of each plot. To address this, metal tags that could be engraved with the plot ID number were used to relabel every plot. This sped fieldwork up considerably.

Due to the large number of plots, it was impossible to perform vegetation surveys in every single plot, which lead to the use of the W-shaped transect.

Several issues occurred with the arthropod trapping and storage. In 2023, the arthropods were stored in 100% ethanol, which resulted in the dissolving/rotting of bycatch slugs. In 2024, 70% ethanol was used, and the slugs were removed right away, just in case. In 2024, the night before the last round of traps were collected, there was heavy rainfall. This caused some of the pollinator traps to fill with more water, and some fell to the forest floor. After all of the arthropods were identified, the data from this round of trapping was examined to see if the fallen traps caused any outliers. There were no outliers, so the data did not need to be removed or otherwise changed.

Additionally, until the summer of 2024, it was thought that all the applied nucleation transplants were beaked hazel shrubs. After it was determined that at least some of the transplants were not beaked hazel shrubs, they all had to be identified, and were found to be yellow birch trees. There was also an attempt to identify the dead shrubs, but this was not possible. Due to the unknown species distribution of the dead transplants, species specific responses to windthrow guards could not occur.

Finally, there was only so much time and budget for soil testing. This meant that additional testing for other soil parameters could not occur, as it was outside of the budget of this project. This data could have provided more information about the state of the soil and detect changes which may have occurred to soil parameters other than the ones tested. Also, some of the paper bags containing soil from 2022 were stored in a tied plastic bag in the refrigerator. This caused them to become moldy, and the paper began to rot. Due to health concerns from the mold, and samples becoming mixed due to the paper rotting, these samples had to be discarded.

2.5 Overall Experiment Conclusions

The high survival rates of the transplants are promising, as is the possible that the increased survival is due to windthrow guards. This is an indication of successful restoration and

demonstrated that windthrow guards may be effective in reducing the mortality of transplanted trees and shrubs.

In terms of forest soil, plant and arthropod community timelines, four years is not a very long time. Although the treatments do not have many statistically significant relationships with soil chemical parameters, vegetation and arthropod richness and diversity metrics, just because change has not occurred yet does not mean it will not.

Anecdotally, there has been wildlife use of the research site. Moose (*Alces alces*) were spotted at the site in 2022 and 2024, and a white-tailed deer (*Odocoileus virginianus*) was sighted in 2023. Additionally, in 2024, a ruffed grouse had a nest and successfully incubated and hatched her eggs at the site. At least two of the chicks survived for several months after hatching, as they were sighted multiple times over the summer of 2024.

This project developed a set of comparatively inexpensive and accessible forest plantation restoration strategies that are successful in at least the short term. These techniques could be adopted by Algonquin Park and other groups or private landowners to speed up restoration of red pine and other forest plantations. Due to the prevalence of red pine plantations, these techniques could speed up the restoration of large areas of land. If other projects use other species as the nuclei and windthrow guards, eventually a set of criteria for optimal or suboptimal species or traits could be developed, if mortality of transplants is tracked.

This project also allowed for the development of a robust set of sampling techniques that can feasibly be perfomed by a single person and is comparatively inexpensive. These sampling techniques could be useful in any terrestrial restoration situation where time, money and personnel are all limitations. These strategies could be adopted by Algonquin Park to continue to monitor the site for this project and could be used to establish baseline monitoring throughout the park. Eventually, this would allow for a long-term dataset that can be easily compared among years and locations. If these monitoring strategies are adopted by multiple organizations, there would eventually be a large dataset that could be compared between different areas and restoration techniques.

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