

The effect of ecosystem change, restoration, and plant diversity on thermally imaged surface temperature

by

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A thesis  
presented to the University of Waterloo  
in fulfillment of the  
thesis requirement for the degree of  
Doctor of Philosophy  
in  
Social and Ecological Sustainability

Waterloo, Ontario, Canada, 2020

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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 contribution

This thesis consists of five chapters (including an Introduction, three data chapters, and a Conclusion). L. Jonas Hamberg is the primary author of all chapters and is exclusively responsible for their preparation, under the supervision of Dr. Stephen Murphy. Chapter 2 has been published as a peer-reviewed article in the journal *Ecological Indicators*. L. Jonas Hamberg was the first author of this article and designed the methodology, performed the analysis, and wrote the manuscript. As part of the process of preparing the manuscript for publication, the committee (Dr. Murphy, Dr. Robinson, Dr. Fraser, and Dr. Trant) were co-authors and provided language edits and suggestions on analysis. Ecological sampling data in Chapter 2 was provided by the Nature Conservancy of Canada and was sampled by their field-staff.

The research presented in Chapters 3 and 4 were conducted as part of an NSERC-CRD grant with Dr. Stephen Murphy as PI and with post-doctoral researcher Dr. Paul Richardson. Dr. Richardson designed the majority of the topsoil transfer experiment, with input from L. Jonas Hamberg. Test-transfers of soil were overseen by L. Jonas Hamberg. Topsoil transfer was undertaken by the staff of Clearview Nursery and overseen by Dr. Richardson, L. Jonas Hamberg, and, to a lesser extent, Andrew Moraga. All vegetation sampling, soil sampling, and tree measurements used in these chapters were overseen by L. Jonas Hamberg and performed by him and research assistants Rohit Verma, Payton Landsborough, Yasmeen Taghreed, Krista Birtles, Gillian Glover-Sams, Drew Welsh, Sara Ghods, Thanya Aat, Hiba Adel ElMiari, Leticia Walter, Erik Ford, Madison Haller, and Javid Said-zadé. The UAV was piloted by Rohit Verma, overseen by L. Jonas Hamberg. The manual extraction of temperature data from thermal images in the FLIR Thermal Studio software in Chapter 4 was overseen and verified by L. Jonas Hamberg, with part of the extraction of data done by research assistants Sara Ghods and Leticia Walter. Dr. Derek Robinson and Benjamin Meinen collected the optical imagery and elevation data used in chapter 4. Benjamin Meinen created the digital elevation model used for elevation measures in Chapter 4.

## Abstract

The objective of this dissertation was to test and quantify the hypothesis from ecosystem thermodynamics: that the surface temperature of a terrestrial ecosystem changes with the state of the ecosystem in general, and with plant species diversity in particular. Daytime surface temperature of vegetated terrestrial ecosystems has been hypothesized to decrease with increased biomass and diversity as they in turn increase transpiration, respiration, physical thermal inertia, and productivity, thereby reducing the portion of energy re-emitted as thermal radiation. The hypothesis is tested, and the results and applications are discussed, within the context of ecosystem restoration.

I investigated the relationship between ecosystem surface temperature and time since restoration, type of restoration methods, and changes in ecological attributes, including plant species diversity, in two projects restoring temperate wooded ecosystems.

The first restoration studied, described in Chapter 2, was a 500+ ha project restoring farmland to oak woodland, spread over 31 fields. Thermal images from 4 space-based instruments along with 12 years of in-situ sampled ecological data of the project were analyzed and compared. Significant decreases in daytime summer surface temperature (4.5 °C in 12 years), and summer diurnal temperature variation (5 °C in 8 years) over time since restorations were found. The study also found a significant relationship between increased plant species diversity and decreased surface temperature when controlling for plant cover and other vegetation attributes. Native plant species had a more pronounced relationship with surface temperature than exotic ones. The results from this study supported the hypothesis, quantified its effect, and showed how thermal imaging from space-based instruments may be used to assess the progress of restoration and the increase in species diversity.

The second project studied, in Chapters 3 and 4, was of application of multiple overlapping restoration treatments to experimental plots at two former farmland sites already planted with trees, and one abandoned gravel pit. The main experimental restoration treatment was the transfer of topsoil from a donor forest. On top of the topsoil, further additions were made of woody debris, shrub plantings, and shade shelters. In-situ sampled ecological differences between experimental restoration treatments and controls were assessed in Chapter 3 and thermal differences in Chapter 4.

Results from Chapter 3 indicated that most native forest plant species survived within the topsoil as it was transferred from the forest, and re-sprouted in the experimental plots on the second season after the transfer. The plant species community of topsoil recipient plots was significantly different from both recipient and donor site control plots as it contained both the transferred plant species and the species already present at the recipient sites. There was no significant effect on the plant community found from the woody debris, shrub plantings, or shade shelter treatments.

Since significant ecological differences were found in Chapter 3, it would be possible to find thermal differences in the same plots in Chapter 4 if the hypothesized relationship was true. To overcome the spatial and temporal resolution limitations of space-based instruments, an Unmanned Aerial Vehicle (UAV)-borne thermal camera was utilized to image the sites four times per day over 9 days. It was found that topsoil recipient plots in the gravel pit site, as well as the smaller and flatter of the two reforestation sites, had significantly lower temperatures than controls (7.0 °C and 2.0 °C respectively) and that the difference in temperature peaked at 2 pm. In the larger and more heterogeneous reforestation site, the normalized surface temperature was only significantly lower than controls (0.7 °C) at 8 pm. There was a significant negative correlation between native forest plant species and surface temperature at all three sites.

The combined evidence from the two projects studied was that temperate wooded ecosystems undergoing restoration decreased in their daytime surface temperature and diurnal temperature variation over time, and compared to controls. This decrease is partly due to increasing species diversity. These results support the existence and the direction of the hypothesized, and before this mostly untested, relationship between ecosystem change (including species diversity) and temperature. It also provides examples of the relative significance and quantity of the same relationship using more ecological, and thermal imaging, data over a longer time-span than in previous studies. Both Chapters 2 and 4 found evidence for native and later succession forest plant species having a stronger effect on surface temperature than non-native and ruderal plant species. The dissertation makes a case for thermal imaging to be used both to evaluate and monitor the progress of restoration, and to quantify the ecosystem service of thermal buffering provided by restoration. It also demonstrates the relative strengths and limitations of space-based and UAV-borne thermal imaging instruments.

## Acknowledgments

This research was made possible through the Ontario Trillium Scholarship and the NSERC-CRD funded project, which was the focus of chapters 3 and 4. I am also very thankful to have received financial support through the R. S. Dorney Memorial Fellowship. I would like to thank the Nature Conservancy of Canada and their southern Ontario staff for access to sites and sampling data used in Chapter 2. Thank you to Mary Gartshore who provided her expertise on restoration by high-diversity seeding and helped me learn to identify tallgrass prairie and oak woodland plants. Thanks also to NASA Jet Propulsion Laboratory and the ECOSTRESS science team, including Dr. Christine Lee and Dr. Josh Fisher, for allowing early adopter access to thermal imaging data and for supporting and promoting my work.

My supervisor Dr. Stephen Murphy has supported and encouraged me throughout the Ph.D. Dr. Roydon Fraser put me on the path of thermodynamics and connected me with the world of thermal imaging in the form of the ECOSTRESS mission. Dr. Derek Robinson fostered my curiosity and understanding of the usefulness of UAV platforms. He also invested a lot of time and energy into helping me learn how to craft my research into a story. Dr. Andrew Trant encouraged and taught me a lot with his knowledge of ecology and statistics.

Thanks to my many amazing research assistants that helped me identify plants and measure trees in weather from sunny and 30 °C to deep snow and -15 °C, without complaints and always with curiosity. Special thanks to Rohit Verma who flew the UAV and to Payton Landsborough who taught me a lot about plant identification. My assistants included: Yasmeen Taghreed, Krista Birtles, Gillian Glover-Sams, Drew Welsh, Sara Ghods, Thanya Aat, Hiba Adel ElMiari, Leticia Walter, Erik Ford, Madison Haller, and Javid Said-zadé. And my most trusted assistant – Bybz, the field-car that got as muddy, snowy, and covered in burrs as me but never gave up.

Deep thanks to my lab-mates and friends Dr. Michael McTavish and Jessica Turecek who both provided sage advice on statistical methods, and to Stephanie Cruz-Maysonet for her work helping me organize the project and sampling. Thanks to my dear friend Dr. Heather Cray for her support and to my master's advisor Dr. Karin Limburg for encouraging my interest in restoration ecology.

Lastly, I would like to thank my parents, my family, and my friends who all supported me and helped me relax and unwind with games, laughs, and food. To my partner, unpaid field assistant, and co-academic, Harshita Yalamarty, I love you so much, and I am eternally grateful for your support through this journey and beyond.

## Dedication

**To my mormor (grandmother), Sigrid Barbro Ljusberg, an exceptional role-model. At age 11, she left her family to continue her education as there were no schools for women beyond this age in her home-town. At age 38, with two children at home, she took the overnight train weekly to gain a university education in history. In her 50's, she joined local politics and took an active part in the successful lobbying for the first university in her home town of Sundsvall. At age 79, she went back one final time to take classes on politics, history, and astronomy. Through her life and work, she taught me the values and joy of constant curiosity, sharing of knowledge, and life-long learning, without subject boundaries.**



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# Chapter 1. Introduction

*“The intermediate forms assumed by solar energy until it falls to terrestrial temperatures can be fairly improbable, so that we can easily use the transition of heat from the sun to earth for the performance of work<sup>1</sup>” – Ludwig Boltzmann, 1886*

## 1.1. Objective

The objective of this dissertation was to test and quantify the hypothesis from ecosystem thermodynamics that a terrestrial ecosystem’s surface temperature changes with the state of the ecosystem in general, and with plant species diversity in particular. The hypothesis is tested, and the results are quantified and discussed, within the context of restoring temperate wooded ecosystems. The discussion is especially focused on applying thermal imaging to restoration to evaluate and monitor its progress and to quantify the thermal buffering ecosystem service it provides. Two restoration projects are studied in this dissertation, one in Chapter 2, and the other in Chapters 3 and 4. Chapters 2 and 4 explore the objective directly, through the use of different technologies (space-based versus Unmanned Aerial Vehicle (UAV)-borne thermal imagers), sites, and restoration methods (Table 1.1), but asking similar questions<sup>2</sup>:

1. Does restoration of wooded ecosystems decrease its daytime summer surface temperature, and if so, by how much, either over time (Chapter 2) or by type of restoration method (Chapter 4)?
2. When during the day is the thermal buffering effect of the restoration the most pronounced (Chapter 4) and could restoration increase relative nighttime temperatures, thereby decreasing diurnal temperature variation (Chapter 2)?
3. What is the relationship between plant species diversity (native and exotic) and surface temperature, when taking into account total plant cover and other ecological attributes?

Chapter 3 tests the relative ecological success of different restoration methods, including forest topsoil transfer, in moving the restored sites towards a state more like reference forest sites. Through this testing, Chapter 3 provides the necessary in-situ measured ecological data for comparisons to be made with surface temperature data in Chapter 4. Chapter 3 is tied to the main objective in that if there were no ecological differences between treatment and control plots found in that chapter, then there would be no reason to expect finding surface temperature differences in Chapter 4 experiments, and therefore no way of answering the dissertation questions.

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<sup>1</sup> Work, in Boltzmann’s writing and in thermodynamics in general is defined as any process in a system which exerts force on its surroundings. This includes any work by an ecosystem, including photosynthesis and transpiration.

<sup>2</sup> Chapter 4 combines the first and second dissertation question into one combined question in the chapter.

**Table 1.1: Details of the two projects studied, illustrating how they attempt to answer similar questions at different sites and with different thermal imaging instruments.**

<b>Sites and data collection details</b>	<b>Project 1 (Chapter 2)</b>	<b>Project 2 (Chapter 3 &amp; 4)</b>
Location	Norfolk County, Ontario, near Long Point and Lake Erie	Simcoe County, Ontario, partially on the Niagara escarpment
Site conditions before restoration method tested was implemented	Active agriculture	Former agriculture sites planted with trees (2 sites) and gravel pit
Type of restoration treatment tested	Oak woodland restoration by high diversity seeding of mid and late succession species	Topsoil transfer from forest, woody debris, shrub planting and shade shelters (described in Chapter 3)
Period of implementation of restoration	First site (out of 31) initially restored in 2006, last site initially restored in 2013	Trees planted in 2015 and 2016, topsoil transfer in fall 2017, additional treatments fall 2017 – spring 2018
Ecological data collector	Nature Conservancy of Canada trained staff (2007-2018)	Author and research assistants in 2019 (described in Chapter 3)
Ecological attributes sampled and measured	Abundance of plant species, ground cover, canopy cover and number of woody stems	Abundance of plant species, ground cover, diameter and height of trees (described in Chapter 3)
Thermal imaging platforms and instruments	Landsat 5 (TM), 7 (ETM+) and 8 (TIRS) satellites and the ECOSTRESS instrument on the International Space Station	UAV quadcopter with DJI Zenmuse XT2 - FLIR Tau 2, uncooled microbolometer (Chapter 4 only)
Thermal imagery collected and used	53 thermal images during summer (May 31st to September 31st), over 17 years, one image covered all sites	Multiple imaging flights per day, over 9 days in July and September, 2019, totaling 98 flights, collecting 1080 images (Chapter 4 only)
<b>Tests of dissertation questions</b>	<b>Chapter 2</b>	<b>Chapter 4</b>
1. Testing and quantifying temperature change due to restoration	Linear mixed effects model and repeated measures correlation of 31 field sites compared before and after restoration over 17 years (2002-2018)	ANOVAs on linear mixed effects models of surface temperature of topsoil recipient plots versus controls at 3 sites
2. Comparing differences in temperature change at different times of the day	Linear mixed-effects model and repeated measures correlation of day and nighttime surface temperature of 19 sites of different age since restoration during growing season 2018	ANOVAs, as above - comparing surface temperature difference of plot-types at 12, 2, 4 and 8 pm in June and September
3. Relationship of surface temperature with plant species diversity and other ecological attributes	Multiple linear regression of ecological data from 59 2x2 m sampling plots in 3 sites under active and passive restoration against normalized surface temperature over 12 years	Correlation analyses of plant data from 150 2x0.5 m sampling plots, and size of all planted trees in 25 12.5 x 10 m plots against normalized surface temperature in July 2019



## 1.2. Why focus on restoration of temperate wooded ecosystems?

Increased loss and degradation of forests, and other wooded ecosystems<sup>3</sup>, due to human development, has led to a global decrease in biodiversity and ecosystem services such as carbon sequestration, soil protection, pollination, and water filtration (Chazdon, 2008; Costanza et al., 2014; De Groot et al., 2002; Löf et al., 2019; Sutton et al., 2016). Temperate forests and woodlands are threatened by invasive species, tree disease, and climate change, and have been cleared for agriculture and mono-culture tree plantations in many parts of the world (Millar and Stephenson, 2015; Newton and Featherstone, 2005). Specific types of temperate forests and woodlands are especially threatened. For example, less than 1 % of the historical extent of oak woodland, the focus of Chapter 2, remains in Ontario (Bakowsky and Riley, 1994).

Translated into economic terms, ecosystem services provided by temperate forests are estimated to have been worth \$US 9.4 trillion per year in 2011 (Costanza et al., 2014), which would equate to 13 % of global GDP in the same year (World Bank, 2020).

Ecosystem restoration is the process of assisting and guiding the recovery of degraded or destroyed ecosystems (Clewel et al., 2004). When implemented successfully, restoration can change the ecosystems successional trajectory (Choi et al., 2008; Suding and Gross, 2006; Walker and del Moral, 2009) and increase the speed of its recovery (Dobson et al., 1997; Suding, 2011); thereby returning many of the sought after ecosystem services (Benayas et al., 2009; Chazdon, 2008; De Groot et al., 2013; Perring et al., 2015). Due to the increasing need to recover ecosystem services, and the potential of restoration to do so, there have been multiple calls for scaling up restoration efforts massively (DellaSala et al., 2003; Löf et al., 2019; Suding et al., 2015). Prominent among these calls are the 2011 Bonn Challenge, where nations committed to restore 350 million ha of degraded and deforested terrestrial ecosystems by 2030, (Suding et al., 2015), and the United Nations declaration of 2021-2030 as the ‘Decade on Ecosystem Restoration’ (Aronson et al., 2020; Young and Schwartz, 2019).

## 1.3. Why does cooling and thermal buffering matter in restoration?

A vegetated ecosystem decreases its surface<sup>4</sup> temperature by transforming and storing some of the incoming solar energy through transpiration, respiration, and production of new biomass (Bonan, 2008 chap. 16-18; Schneider and Kay, 1994), which decreases the proportion re-emitted as radiated thermal

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<sup>3</sup> The term ‘wooded ecosystems’ are used to refer to both closed canopy forests and more open woodland, such as the oak woodland restoration project presented in Chapter 2.

<sup>4</sup> Surface is referred to in this dissertation as the surface as seen by the thermal imager from above, which in a closed canopy forest means the canopy, or in the case of an more open woodland, or area undergoing restoration is a mix of the ground vegetation layer, soil, and canopy.

energy (sensible heat)<sup>5</sup>. In one study of a temperate mature forest ecosystem, 49 % of the energy absorbed by the forest was transformed into latent heat through evapotranspiration<sup>6</sup> and 2.5 % was transformed and stored through respiration and production of new biomass (Gosz et al., 1978).

The relative decrease of extreme or peak temperature by vegetation, referred to as thermal buffering (e.g. Aerts et al., 2004; Lin et al., 2020; Wagendorp et al., 2003) or temperature regulation (Costanza et al., 1997), has been considered an ecosystem service under the wider umbrella of climate regulation (Betts et al., 2018; Costanza et al., 1997; Lin et al., 2020). However, while it has been discussed within the context of urban heat island effects (e.g. Jenerette et al., 2011; Livesley et al., 2016; Rafiee et al., 2016) and green-roofs (e.g. MacIvor et al., 2016; Simmons et al., 2008), **the service of thermal buffering provided by vegetated ecosystems has not yet been explored or quantified within the field of restoration ecology**. Many large reviews and studies on ecosystem services affected by restoration have not included thermal buffering, or similar concepts, at all (e.g. Barral et al., 2015; Benayas et al., 2009; Birch et al., 2010; Bullock et al., 2011; Palmer and Filoso, 2009); even though the value of thermal buffering to restoration funders and stakeholders could be high.

Areas affected by human development, such as quarries, pits, roads, and farm-fields, can all have increased daytime summer temperature compared to forests and other vegetated natural ecosystems under the same weather conditions (Holbo and Luvall, 1989; Lin et al., 2020; Tuff et al., 2016). These developed areas can also increase local temperatures into the edges of adjacent areas (e.g. Delgado et al., 2007; Robinson et al., 2009; Tuff et al., 2016). If the adjacent areas are natural ecosystems, including forests, then plant and animal life could be adversely affected if temperatures rise beyond their tolerance-limit (e.g. Hijmans and Graham, 2006; Jiguet et al., 2007; Tuff et al., 2016). If the adjacent areas are urban instead, then the increased peak temperatures can decrease human well-being (Douglas, 2012; Heaviside et al., 2017). Conversely, if the restoration of ecosystems can re-establish the cooling and buffering effects of more natural terrestrial ecosystems, and size of this effect can be quantified, it can become an effective argument for more restoration in both rural and urban areas.

Exploring the relative cooling and thermal buffering effect of different ecosystem restoration methods could also lead to new methods for evaluating and monitoring the progress of restoration of wooded ecosystems. Through measurements from flux-towers, two studies have shown that temperature buffering ability increases as forests increase in age and height (Lin et al., 2017b) and decreases in forests

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<sup>5</sup> Heat is energy in transfer, and sensible heat is the transfer of energy that can be measured as a temperature change. Latent heat is the energy that is transferred through a phase change, due to a shift in volume or pressure – including the change of water from liquid to vapor (Cengel and Boles, 2008 chap. 3).

<sup>6</sup> Evapotranspiration refers to both transpiration from plants and evaporation from soil. For forested ecosystems the active transpiration of plants tends to dominate over passive evaporation, and can make up over 90% of the total of evapotranspiration during the growing season (Jasechko et al., 2013; Wilson et al., 2001).

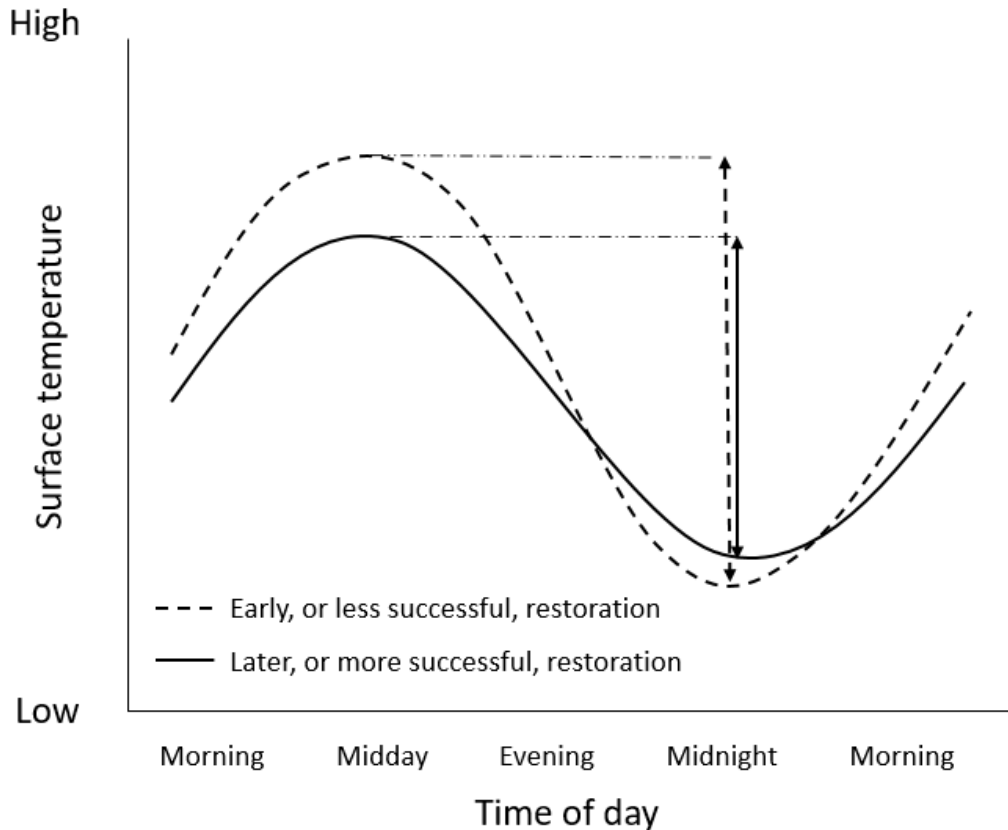
that are considered more degraded (Lin et al., 2017a). An earlier study using an airborne thermal infrared scanner also showed that younger recovering forests and plantations were warmer than older forests (Holbo and Luvall, 1989). Through thermal imaging from UAV, one study has shown that relative surface temperatures of diseased trees increase in semi-naturalized tree plantations (Smigaj et al., 2019). Water stress has also been shown to increase surface temperatures in monoculture tree plantations (e.g. Iizuka et al., 2018; Ludovisi et al., 2017). With the results from these studies in mind, it seems plausible that the relative progress of a forest restoration project could be evaluated and monitored by its ability to cool and buffer temperatures through mapping and comparisons over space and time. Aside from a single one-year study conducted using hand-held infrared thermometers (Aerts et al., 2004), the use of surface temperature for monitoring and evaluation has not been explored within the field of restoration ecology.

#### **1.4. Why does it matter when the cooling and buffering occurs?**

The surface temperatures of ecosystems decrease as incoming energy from the sun is stored and transformed. Some of that energy is only stored for a short period – minutes to hours – either as physically stored thermal inertia<sup>7</sup> (Jayalakshmy and Philip, 2010) or as chemical energy stored in the respiration process. The respiration process uses energy taken up during the day, releasing some of the remaining energy as sensible heat during the night. With an increase in biomass, the capacity to physically and chemically store energy increases, and more of it is released later in the day, and into the evening and night, leading to increasing temperatures away from the daytime peak (Chen et al., 1999; e.g. Lin et al., 2020). In this way, recovery after ecosystem restoration could be evaluated not only by how much it lowered high temperatures during the day, but also on how it raised low temperatures at night – or, taken together, by the range of diurnal variation in surface temperature (Figure 1.1), as explored in the second experiment of Chapter 2.

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<sup>7</sup> Thermal inertia is the relatively slow conduction of energy through a material, which is based on the material's heat capacity and its conductivity (Jayalakshmy and Philip, 2010; Kondepudi and Prigogine, 2014).



**Figure 1.1: Illustration of hypothesized difference in diurnal temperature variation of a forest ecosystem early after restoration versus later. The same illustration also works for comparing the difference between same-aged projects where the restoration effort was more or less successful. Projects and sites can be compared by subtracting temperature variations at the same two points in time (lines with arrows), near their daily maximum and minimum temperatures.**

It should not be assumed that the diurnal pattern is always that surface temperature is buffered most at the time of peak incoming radiation, or highest overall temperatures. For example, in a very hot and arid climate, where plants would shut down their activity during the mid-afternoon to preserve water (Chaves et al., 2016) it may be that more cooling occurs later in the day and evening when transpiration resumes. Exploring when during the day thermal buffering was most pronounced for our temperate forest site in southern Ontario is part of the contribution of Chapter 4.

Humans, non-human animals, and plants all have rhythms of activity during the day and night (circadian, or otherwise), and it matters not just how much thermal buffering happens but also when it happens. A relative increase in nighttime temperature during the growing season could be a benefit for some organisms that cannot regulate their own temperature well, such as ectothermic lizards (Kearney et al., 2009; Tuff et al., 2016). For humans in urban settings, cooling is most sought after during summer

afternoons, as evidenced by air-conditioning usage (e.g. Salamanca et al., 2013). If it can be shown that restored ecosystems not only cool average surface temperatures but that they do so at times when cooling is needed the most, then it adds further to the argument for the thermal buffering service of restoration, whether urban or rural.

### **1.5. Why explore the relationship between surface temperature and diversity?**

Increasing biodiversity is often considered both a goal and a measure of restoration project success (Benayas et al., 2009; Bull et al., 2013; Bullock et al., 2011). Studies based on the thermodynamics of ecosystems have hypothesized that species diversity, along with increased trophic, structural, and functional complexity, could further increase the cooling and temperature buffering of ecosystems (Fraser and Kay, 2004; Kay, 2000; Schneider and Kay, 1994). If we can establish that temperature buffering increases not just with biomass but also with plant species diversity, then thermal imaging becomes a much more interesting and useful tool for ecosystem evaluation and monitoring.

The mechanisms through which increased biodiversity could decrease and buffer surface temperatures are indirect. Increased plant diversity has been shown to be able to increase and stabilize both transpiration (Baldocchi, 2005) and productivity (Cardinale et al., 2007; Hooper et al., 2005; Tilman et al., 2001), which are the principal processes of ecosystem energy transformation and cooling. This increase and stabilization in transpiration and productivity with increasing diversity has been ascribed to, among others, an increase in nutrient cycling (Loreau, 1995), and an increase in variations in root depth, life-span, leaf assemblage (overlap and shape), and phototropism (Baldocchi, 2005; Schneider and Kay, 1994). This dissertation contributes by testing the hypothesis brought forward by Schneider and Kay (1994) – that diversity decreases surface temperatures – which so far remains mostly untested (Avelar et al., 2020; Cushman, 2015). The dissertation further explores if later succession native plant species have a stronger cooling effect than more ruderal exotic plant species. The theoretical argument for native and later succession plant species having a stronger effect on surface temperature is that they have evolved to be more effective at energy uptake in their native range. They may also increase structural and trophic complexity more than exotic and ruderal species, which further increases energy uptake. If it is true that native and later succession species cool more, then this also strengthens the use-case for thermal imaging of restoration, as the goal of many forest restoration projects is not just an increase in species diversity, but specifically an increase in native species diversity (Stanturf et al., 2014).

The tests on the relationship between temperature and diversity were done by combining thermal imaging and in-situ measurements of plant species diversity and other ecological data. I have so far not come across any other comparison of thermal imagery from space or UAV with in-situ sampling of plant diversity. The closest study used surface temperature taken from data loggers, compared with forest sites

of different species diversity, mainly old-growth forests, and tree plantations, and found a significant cooling in the more diverse and complex old-growth forests (Norris et al., 2012). Another study used thermal imaging but used structural complexity as measured remotely as a proxy for diversity (Avelar et al., 2020). Unfortunately, there seems to be a lack of collaboration between thermal remote sensing scientists and field ecologists.

While the results and applications discussed in this dissertation are done so mainly within the context of restoration, I also believe the findings can be expanded and adapted further into related areas of research and practice, such as conservation, ecosystem management, and systems ecology.

## **1.6. Dissertation outline**

This dissertation aims directly, or indirectly, to answer the questions outlined at the beginning of this chapter and discuss the implications of these answers. It consists of three manuscript data chapters and a concluding chapter:

Chapter 2 presents the first study of the dissertation, which was comprised of three connected experiments. For the three experiments, we<sup>8</sup> utilized thermal imaging and in-situ sampled ecological data from a 500+ ha oak woodland restoration project, spread over 31 fields in Norfolk County, Ontario, Canada. Experiment 1 compared the surface temperatures of 31 former agricultural fields before and after undergoing restoration for a total of 16 years, using thermal images from three Landsat satellites and the ECOSTRESS instrument on the International Space Station. Experiment 2 compared daytime thermal images from Landsat 8 and ECOSTRESS against nighttime imagery only available from ECOSTRESS, during 2018. Experiment 3 compared in-situ sampled ecological data, including plant species diversity data, against thermal data for 3 restoration fields (more details in Table 1.1). This chapter is published in the journal *Ecological Indicators* as Hamberg et al. (2020).

Chapter 3 describes the implementation of an experiment (and second study of the dissertation) applying multiple restoration treatments methods to plots in two former farm-field sites already planted with trees, and one gravel pit site in Simcoe County, Ontario. The most prominent treatment applied and tested was that of topsoil transfer from a nearby deciduous forest. The experiment also included the addition of woody debris, shrub plantings, and shade-shelters on top of the added topsoil. Multiple types of control plots were also established at both topsoil donor and recipient sites. The plant diversity, plant community composition, and tree growth of the topsoil and control treatments were compared, through ANOVA and Analysis of Deviance Wald Chi-Square tests on linear mixed regression models, and through PERMANOVA tests on Non-Metric Multidimensional Scaling models. These tests were

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<sup>8</sup> These chapters uses the third person ‘we’, as committee members have provided guidance and are, or will be, co-authors of these studies.

conducted to ascertain if the topsoil transfer and additional methods were useful for improving the trajectory of the restoration towards a native late-succession forest species composition. The tests were also conducted to see if there were ecological differences that could be investigated through thermal imaging in the following chapter.

Chapter 4 compares the surface temperature of topsoil recipient, and control, plots of the three experimental sites described in Chapter 3. To compare and quantify the surface temperature at different times of the day and of the growing season, we utilized thermal images from a UAV taken 4 times per day over 9 days total in July and September 2019. We then used correlation analyses to explore the relationship between the normalized surface temperature of all treatment and control plots and the in-situ sampled ecological data from Chapter 3, to test if increased species diversity, and other ecological attributes, correlated significantly with decreased surface temperatures as hypothesized (more details in Table 1.1).

Chapter 5 concludes the dissertation by considering what the results of the studies mean for how we think about the surface temperature of ecosystems undergoing restoration, and how thermal imaging from both space-based instruments and UAV could potentially be utilized by restoration and conservation managers. It also outlines potential paths of research to further test and deepen our understanding of how ecosystem diversity, function, and structure affect temperature.

## **Chapter 2. Surface temperature as an indicator of plant species diversity and restoration in oak woodland**

### **2.1. Introduction**

A goal in many restoration projects of wooded ecosystems is to establish a highly diverse mix of native vegetation (Ruiz-Jaén and Aide, 2005; Suding et al., 2015). However, recovery may take decades, and issues such as low viability of planting stock or seeds, changes to the surrounding landscape or climate, disease, or invasive species can put the ecosystem on an unwanted trajectory (Halme et al., 2013; Suding, 2011). To identify these issues before they become irreversible, and adjust restoration procedures to reach a goal of high native species diversity requires consistent monitoring (DeLuca et al., 2010; Wortley et al., 2013).

The most common form of monitoring and evaluation of terrestrial restoration efforts is through in-situ sampling of vegetation (e.g., ground and canopy-cover, composition, species diversity) and soil factors that are used as indicators of the current state of the ecosystem and when repeated offer insight about the trajectory of the ecosystem (Ruiz-Jaén and Aide, 2005; Wortley et al., 2013). However, as restoration projects increase in their number, size, and complexity (DeLuca et al., 2010; Suding et al., 2015), in-situ sampling can become too costly and time-consuming (Henry et al., 2019; Reif and Theel, 2017).

Remote sensing offers one approach to overcome the spatial and cost limitations of field sampling (DeLuca et al., 2010; Reif and Theel, 2017). Satellite imagery can be used to quantify patterns of plant diversity, but to do so directly requires very high-resolution imagery ( $\leq 5$  m), which is expensive (Nagendra et al., 2013; Turner et al., 2003). In addition to diversity, a compilation of spectral bands has been used to estimate vegetation cover (e.g. Leaf Area Index) and photosynthetic activity (e.g. Normalized Difference Vegetation Index) in scientific studies of restoration, but implementation of these methods by restoration practitioners remains uncommon (Nagendra et al., 2013; Reif and Theel, 2017).

Less common still is the use of remotely sensed thermal imaging to monitor and assess ecosystem restoration projects. Contemporary reviews of remote sensing of ecosystems and monitoring of restoration fail to mention thermal imaging or do so only briefly (e.g., de Almeida et al., 2019; Pettorelli et al., 2018; Reif and Theel, 2017). Yet, thermal data can be used to measure or estimate important ecosystem characteristics such as canopy temperature (Luvall et al., 1990) and evapotranspiration and water stress (Fisher et al., 2017).

Thermal imaged relative surface temperature may also have potential as an indicator of relative change in plant species diversity. Plants have some capacity to regulate their leaf temperature using



evolved traits such as stomatal conductance, geometry, and absorptivity (Michaletz et al., 2015). An increase in plant species diversity increases the diversity of functions (e.g., nutrient uptake), structures (e.g., root-depth and xylem structure), and phenology (e.g. leaf-out timing) in an ecosystem (Baldocchi, 2005; Isbell et al., 2009). These variations can increase the ratio of incoming solar energy used by the ecosystem both through evaporation and photosynthesis, and decrease the ratio of energy reflected or re-emitted as heat (Baldocchi, 2005; Jorgensen and Svirezhev, 2004; Kay, 2000; Schneider and Kay, 1994). This means that on a warm clear summer day without strong winds and all other factors being equal, more diverse ecosystems should be cooler than less diverse ecosystems (Fraser and Kay, 2004).

An increase in diversity of function and structure has also been theorized to allow for an ecosystem to retain solar radiation for longer, both through physical and chemical heat storage; moderating daily thermal fluctuations (Norris et al., 2012; Ulanowicz and Hannon, 1987). By retaining the heat and releasing a larger portion of it at night, the more diverse ecosystem remains warmer at night and cooler during the day (i.e. less diurnal temperature variation) relative to a less diverse, but otherwise similar, ecosystem (Fraser and Kay, 2004; Ulanowicz and Hannon, 1987).

Ecosystem development has been shown to correlate positively with cooler temperatures, as measured with airborne thermal sensors (Luvall et al., 1990). Ecosystems with more structural complexity and trophic interactions heat up slower than less complex ecosystems when measured with hand-held thermometers (Aerts et al. 2004). Studies of flux-net tower measurements have found that mature forest buffer temperature change most effectively and that this capacity decreases with degradation of the forest (Lin et al., 2017a). Similarly, thermal imagery from an Unmanned Aerial Vehicle has been used to detect stress due to disease in trees by patterns of increased heat (Smigaj et al., 2019). However, thermal imaging has not been applied using satellite data over time on ecosystems undergoing restoration. Thermal imaging data has also not been compared to in-situ sampled plant species diversity and cover data.

To overcome these gaps, and answer calls to expand the toolset of remote sensing indicators for restoration monitoring and evaluation (Anderson et al., 2017; de Almeida et al., 2019; Reif and Theel, 2017), we sought to answer the following three questions:

1. Is there a difference in ecosystem surface temperature before and after restoration and if so, what is the rate of change over time since restoration commenced?
2. To what extent does diurnal temperature variation change with time since initial restoration?
3. Accounting for biomass and shade conditions, what, and how large, is the effect of increasing plant species diversity on surface temperature?

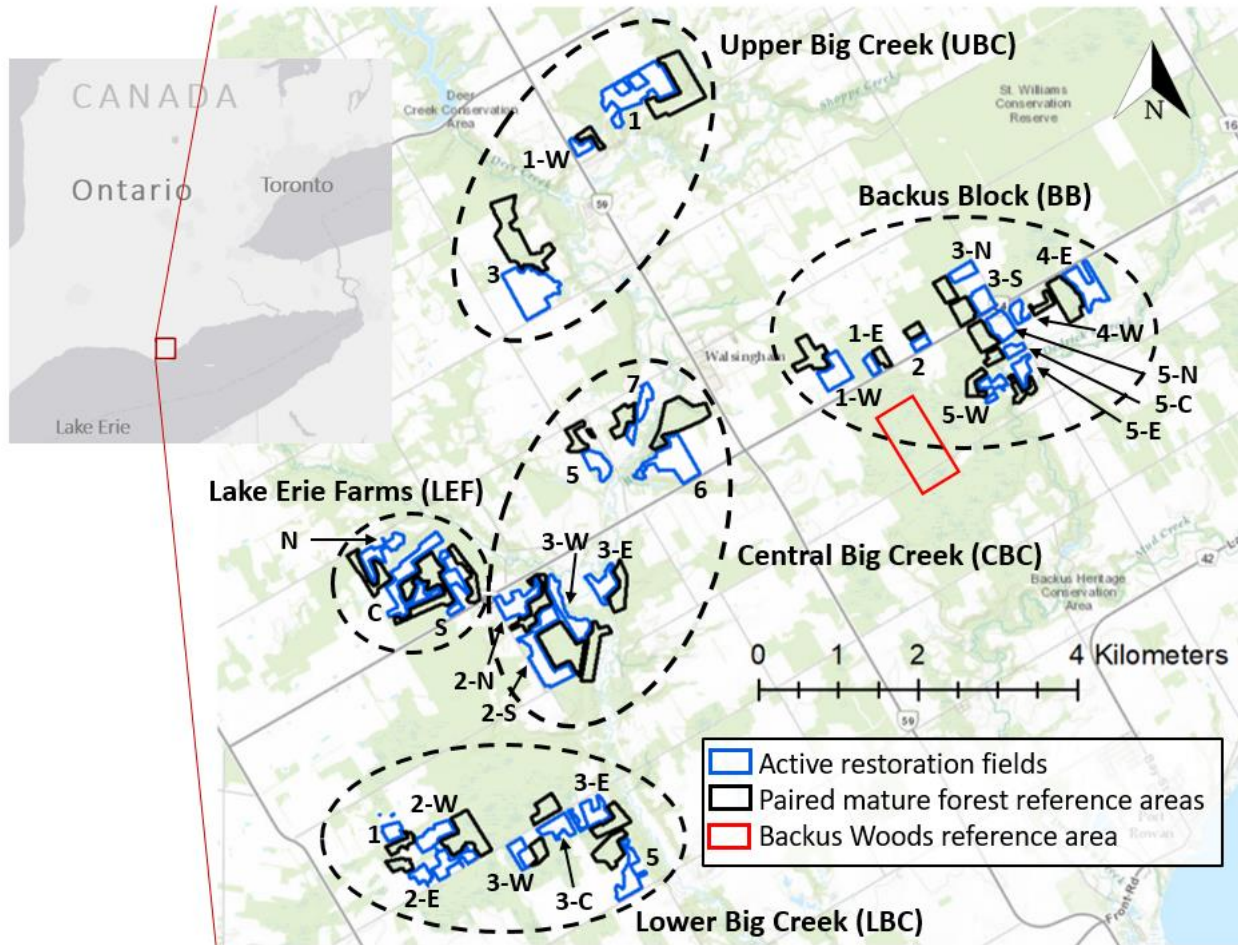
To answer these questions, we quantify surface temperature from thermal satellite imagery across a set of fields undergoing active restoration (through seeding), and passive restoration (unseeded), and normalize the temperature against stable areas made up of nearby mature forests. We compare satellite-based measurements to ground-surveys to determine the efficacy of satellite-based thermal data for predicting changes in ecosystem diversity as a function of surface temperature change. Furthermore, we use ground and canopy cover data to control for effects of biomass or shade.

## **2.2. Methods**

### **2.2.1. Study Area**

A large-scale restoration project was established in Norfolk county, southern Ontario in 2006 by the Nature Conservancy of Canada (NCC). The goal of the project was to restore 700+ ha of mainly agricultural fields to prairie, wetland, mixed forest, and oak woodland (Figure 2.1). The project is spread over 34 properties, some of which contain multiple fields.

The majority of the fields are on eolian fine sand to sandy loam soil, with some edges of fields on variable alluvial deposits. Fields LBC 3-E and LBC 5 are on clay soil (OMAFRA, 2015). The area is flat except for sand dunes and smaller stream valleys (OMNRF, 2019). Preparation for restoration included earth moving to recreate microhabitat, plowing, spraying of herbicides, and fencing to reduce grazing by deer. All fields were seeded with a high diversity seed mix containing between 85 and 106 species of native herbaceous plants, prairie grasses, shrubs, and trees, dominated by oaks (Gartshore, Unpublished; Henry et al., 2019; Inlow, 2010, pp. 23–29; Nature Conservancy of Canada, 2006).



**Figure 2.1: Map of NCC restoration fields that fulfill the criteria for the study (blue) and paired mature forest normalization areas (black). The red rectangle represents the most mature deciduous forest in the region, and it was used to normalize images for the map in Experiment 1. Green represents forested areas. Names of sub-areas and numbering of fields are the same as used for NCC property maps. Cardinal directions are used for separate fields within NCC properties.**

The first NCC property site to undergo restoration in the project was Lake Erie Farms. The site is composed of three fields and is the only site in the project to be in-situ sampled. During restoration one area in each field was left unseeded and represents a passive restoration area. The areas surrounding the Lake Erie Farm restoration fields are mainly mature (50+ years) mixed hardwood forest, dominated by oak (*Quercus spp.*) and maple (*Acer spp.*) with a closed canopy (Henry et al., 2019; Nature Conservancy of Canada, 2006). The majority of the property is on well-drained Plainfield fine sand soil (Nature Conservancy of Canada, 2006). While only these three fields have been systematically surveyed, discussions with NCC staff and the practitioner responsible for the seeding and upkeep (personal communications, Liv Monck-Whipp 2019 and Mary Gartshore 2016) and personal observations in 2016

and 2017 indicated that most restoration fields in the project were increasing in both biomass and plant species diversity.

### **2.2.2. Experiments**

We designed three experiments that each corresponds with, and seek to answer, one of the three questions posed:

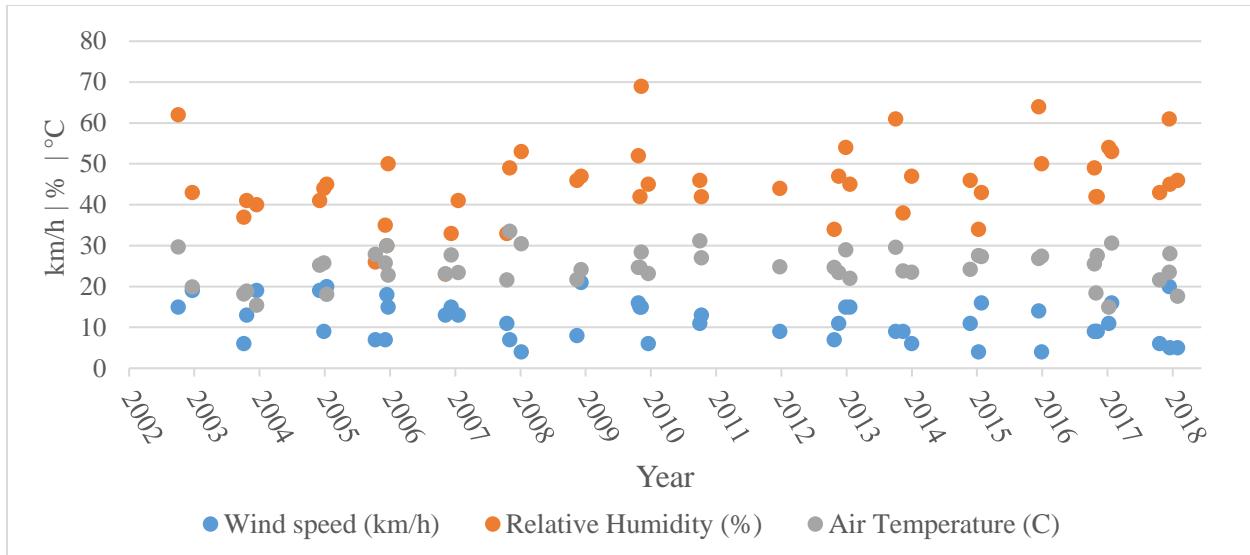
***Experiment 1: Temperature change over time since restoration.*** The relative mid-day land-surface temperature of 31 agricultural fields being restored to woodland by NCC was compared, both before and after restoration, over the period 2002-2018 using Landsat and ECOSTRESS imagery.

***Experiment 2: Comparison of diurnal temperature variation and time since restoration.*** The difference of surface temperature between day and night for three pairs of ECOSTRESS and Landsat images was compared for 19 restoration fields in 2018, through a space-for-time substitution, with years since restoration as the independent variable.

***Experiment 3: Comparison of vegetation and temperature.*** Relative land surface temperature data from three Landsat satellites were compared to vegetation data, including plant species diversity data, sampled at ground level for three restoration fields at the Lake Erie Farms property from 2007 to 2018.

### **2.2.3. Site and imagery selection**

To allow for sufficient photosynthetic activity, the criteria for including satellite imagery in the study were that imagery had <20% cloud cover over the study area (full extent of Figure 2.1) and that it was captured between May 31<sup>st</sup> and September 31<sup>st</sup> of 2002 to 2018. Weather data (Figure 2.2), taken from the Delhi weather station (Environment Canada, 2019), 26 km north of the southern-most restoration site, was used to exclude any image taken when surface temperature measurements of vegetation would not accurately reflect the energy use of the ecosystem. To avoid a large cooling effect on leaves from wind, imagery was only selected if wind-speeds were below 25 km/h (Grace, 1988). To avoid transpiration and photosynthesis slow-down imagery was only selected if air-temperatures were between 10 and 40 °C and relative humidity was below 75% (Körner, 2006).



**Figure 2.2: Air temperature, relative humidity and wind-speed measured within 30 minutes from imagery taken for all daytime imagery used in the experiments.**

### 2.2.3.1. Site selection for Experiment 1 - Temperature change over time

To be included in this experiment, restoration fields had to have at least 5 years of data (initially restored by 2013 latest), to limit the effect of variable weather conditions and short-term disturbances. Fields had to be at least two hectares in size and a minimum of 100 m wide on the shortest side, to decrease edge effects. Thirty-one fields fulfilled our selection criteria (Figure 2.1). To further decrease edge effects, all field polygons were reduced by a 20 m interior buffer before analysis. The total size of all fields after removing the buffer area was 294.5 ha, with a mean field size of 9.4 ha (SD 6.8 ha), the largest field was 26.8 ha and the smallest 2.6 ha.

To allow for normalization of temperature over time, each restoration field was paired with an adjacent mature forest area. Hereafter referred to as ‘normalization areas’. These areas were selected to border the restoration field, have an area within 10% of the restoration field’s size, have as similar perimeter-to-area ratio as the landscape permits, have a similar amount of edge adjacent to active agriculture fields and be composed of a closed canopy deciduous forest. Each forest area was checked visually using aerial high-resolution imagery from 2006, 2009, 2010, 2014, 2016 and 2018, and lower resolution satellite imagery from 2002 in Google Earth (Google Inc.), and adjusted to avoid canopy openings or other visible disturbances. We further checked to avoid any major differences in elevation or soil conditions using a Lidar-derived terrain model and the Ontario Soil Survey Complex (OMAFRA, 2015; OMNRF, 2019). For analyses, the forest areas were reduced by a 20 m interior buffer, similar to the restoration field.

### 2.2.3.2. Site selection for Experiment 2 - Diurnal temperature variation change with time

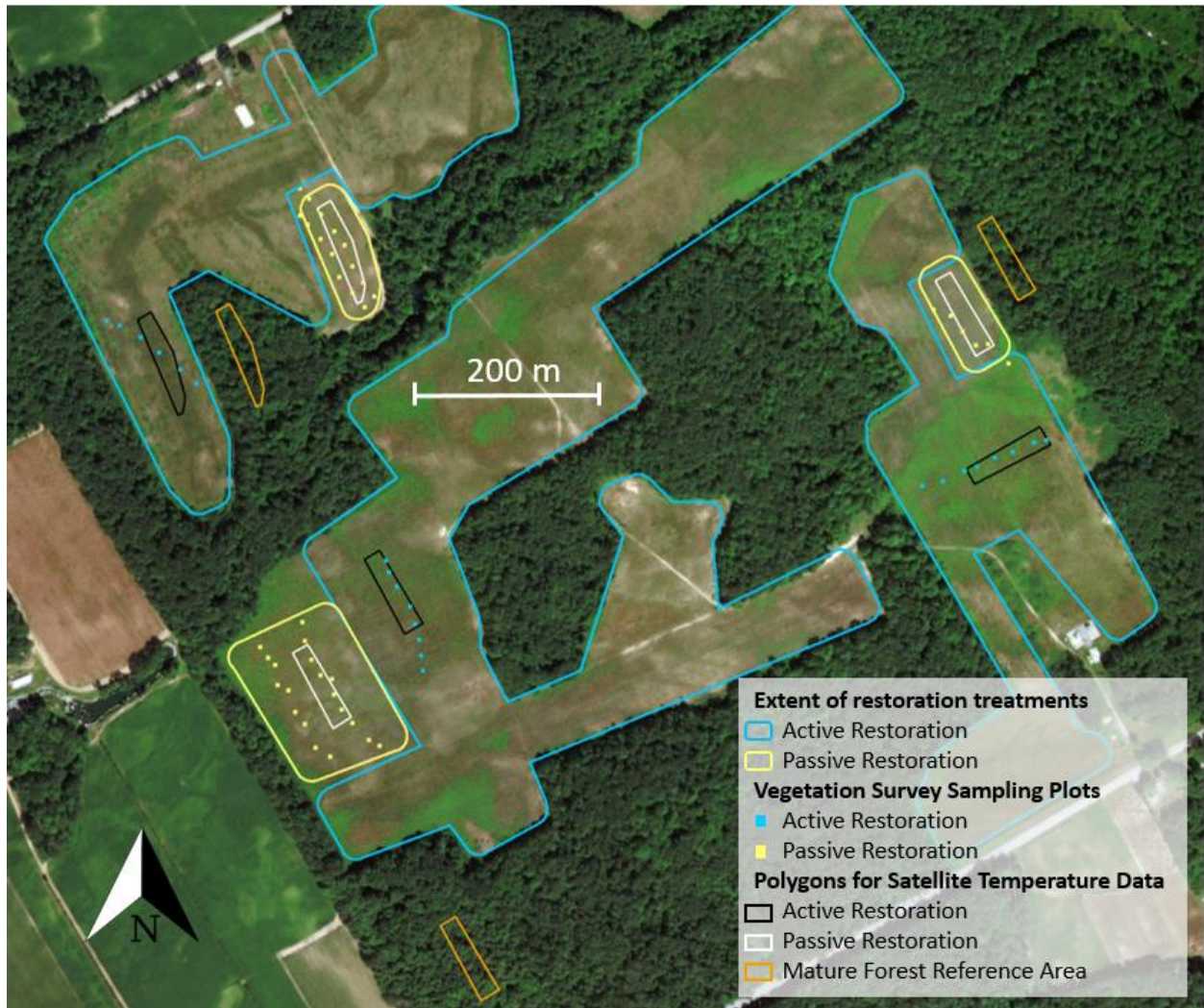
The criteria for site selection for Experiment 1 were also used for Experiment 2. Additionally, this experiment uses space-for-time substitution – i.e., different fields representing different ages since restoration, measured at the same time – and further criteria were needed to control for edge and center effects due to field size and shape differences. Temperature has been shown to be affected by the edge up to 50 meters into an oak-chestnut forest (Matlack, 1993; Robinson et al., 2009) and forests also influence prairie and open fields ecologically, through in-seeding, and thermally, through shading and its effect on wind-patterns and moisture (Dovčak et al., 2005; Gehring and Bragg, 1992). To minimize edge effects between adjacent forest and our restoration areas the internal edge buffer was increased to 70 m. Further, to give each field a more equal amount of perimeter-to-area ratio (Hunsaker et al., 1990) any area more than 140 m from the edge of the field was removed. After applying the buffers we removed all areas that were under 1 ha, resulting in 19 field areas of analysis between 1 and 10.5 ha.

### 2.2.3.3. Site selection for Experiment 3 - Comparison of vegetation and temperature

For Experiment 3, sites were selected from Lake Erie Farms, as it is the only property where vegetation monitoring has taken place. The criteria for selecting the areas of study within the property was for all areas to have a minimum of 6 sampling plots within them, and for each area to be as large as possible while keeping all sites approximately the same size, shape, and edge and distance to forest. We also checked the altitude and slope of restoration areas from Lidar-derived data (OMNRF, 2019).

All three passive restoration areas were retained and three active restoration areas of similar size (2589-2708 m<sup>2</sup>), shape and distance to forest edge were selected; one near each of the passive restoration areas. A similarly sized and configured mature forest normalization area was selected to pair with each restoration area in order to compute relative temperatures between restoration areas and mature forest areas to allow for comparisons of temperature and vegetation over time. All polygons used for thermal measurements were given a minimum 20 m internal buffer from any other treatment or ecosystem type (Figure 2.3).





**Figure 2.3: Map of the Lake Erie Farms property and restoration sites. Three paired active restoration, passive restoration and mature forest normalization areas were selected for the study, along with the nearest transect of vegetation sampling plots.**

#### **2.2.4. Vegetation sampling and measurements**

Once per year, from 2007 to 2018, trained NCC staff surveyed 137 2x2 m plots on the Lake Erie Farms property, for percentage ground cover of each plant species present. From 2011 onwards, the sampling also included a count of the number of woody stems and percentage canopy cover for each species. In total, the areas included in Experiment 3 include 59 survey plots with at least 6 plots per studied area (Figure 2.3). The mean measurements of each plot transect were used for statistical analysis. Our use of 6 plots or more for each area provides a representative sample of the larger area as determined by previous research at our study site (Henry et al., 2019).

The percentage ground cover of each plant species in each plot was used to calculate the ‘effective number of species’ or ‘Hill number’ (Table 2.1). The Hill number is based on the Simpson-dominance diversity index and accounts for both total species richness and its evenness (Chao et al., 2014). Species were defined as native or exotic to Ontario according to the Database of Vascular Plants of Canada (Brouillet et al., 2010). Ecological data from the project are available in Hamberg (2019).

**Table 2.1: Summary of diversity measurement equations, where  $\lambda$  is the diversity or evenness,  $R$  is the total species richness for the plot surveyed, and  $p$  is the proportional cover of species  $i$ .**

Simpson-dominance index	$\lambda = \sum_{i=1}^R p_i^2$
Simpson Hill number	$1 / \sum_{i=1}^R p_i^2$

In-situ sampled percentage ground cover by vegetation, number of woody stems, and percentage canopy cover, as well as relative NDVI derived from Landsat imagery were used as proxies for biomass. Total percentage canopy cover was also used an indicator for potential shading effects.

### 2.2.5. Selecting and processing satellite imagery

Satellite data were acquired from NASA/USGS satellites Landsat 5, Landsat 7 and Landsat 8, and from the ECOSTRESS thermal imaging instrument mounted to the International Space Station (ISS). Landsat imagery thermal bands from each satellite (Band 6, Band 6 VCID\_2, and Band 10 respectively) were used to estimate at-satellite brightness temperature. The red and near-infrared bands from the Landsat satellites were used to derive NDVI measurements. Landsat 7, 8 and ECOSTRESS thermal radiometric accuracy has been found to be within  $\pm 1$  °C at 27 °C when validated against ground measurements (Hook et al., 2019). To process and correct Landsat level-1 satellite data (Zanter, 2019) a Digital Number to Land Surface Temperature (DNtoLST) model was constructed using ArcGIS Desktop 10.5.1 (ESRI) Model Builder.

Landsat data is adjusted for emissivity in the DNtoLST model using NDVI threshold method (Jiménez-Muñoz et al., 2009). Soil and vegetation emissivity values were sourced from the ECOSTRESS library (Meerdink et al., 2019) and literature (Sobrino et al., 2008) based on soil maps of the site and vegetation surveys (Nature Conservancy of Canada, 2006). Atmospheric adjustment were done using the Atmospheric Correction Parameter Calculator (Barsi et al., 2005). Any area within 200 m of a cloud was removed from analysis. For a detailed description of processing methods, see Appendix 2A. With the surface temperature adjusted for emissivity and atmospheric conditions, the DNtoLST model then outputs

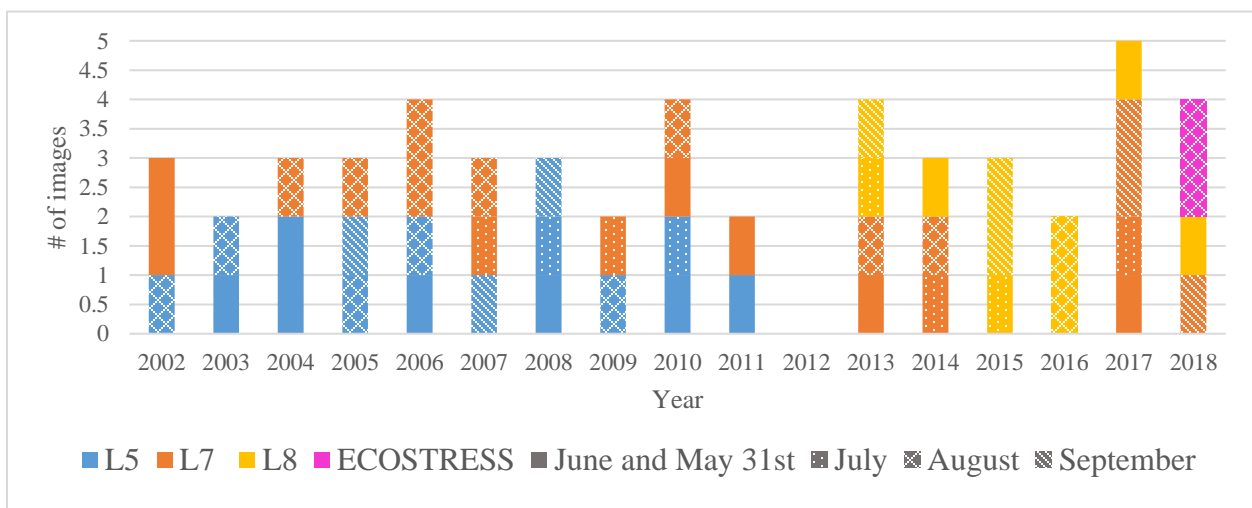


summary statistics weighted by the percentage of the pixels that were inside the polygon of interest (Conrad, 2003).

The native resolution of thermal images from Landsat 5 and 7 is 120x120 m, and 100x100 m for Landsat 8 but all Landsat level-1 thermal data are provided as resampled to 30x30 m pixels (U.S. Geological Survey, 2016). Data from each field were only used if at least 75% of pixels were available for the sampled restoration area polygon and its paired normalization area polygon.

In addition to the acquired Landsat data, land surface temperature data were acquired from ECOSTRESS (build version 5.03) for 2018 through the Early Adopter program (Hulley and Hook, 2018) and converted to geo-located grid data using the swath2grid python programming tool (Krehbiel, 2019). ISS orbit varies and therefore so does ECOSTRESS time of local imagery acquisition. Imagery from ECOSTRESS were considered as taken during the day and included in all experiments if it was captured between 10:00 and 19:00 eastern standard time, which provided a buffer following sunrise for the ecosystem to warm up and commence photosynthetic activity. ECOSTRESS imagery were considered as taken during the night, and included in Experiment 2, if taken between 22:00 and 07:00, to allow the ecosystem to cool after sunset. ECOSTRESS has a spatial resolution of 69x38 m.

Out of approximately 232 images available for the area and time from Landsat 5, 7 and 8, 48 images fulfilled the criteria for the 2002 to 2018 period. The only year without a single useable product was 2012 (Figure 2.4). Five cloud-free ECOSTRESS images between first imagery of the site in July 30 to September 21 2018 were found and included in the study. Two of these images were recorded close to mid-day and were included in all experiments. Three of the 5 useable ECOSTRESS images were taken during nighttime and were only included in Experiment 2.



**Figure 2.4: Imagery taken near noon and used for Experiments 1 and 3 by instrument year and month. L5-L8 - Landsat 5-8 satellites.**

## 2.2.6. Statistical comparison and analysis

To test the statistical significance of temperature and vegetation changes over time in all experiments in this study we fit linear mixed effects models, repeated measures correlations and multiple linear regressions (ordinary least square). All analyses were performed in R. The linear mixed effects models use the ‘lme4’ package in R, and we report confidence intervals but p-values are not appropriate for these mixed models and are therefore not reported (Bates et al., 2015). Repeated measures correlation were calculated using the ‘rmcorr’ package in R (Bakdash and Marusich, 2017). This tests for the significance of the correlation of slopes between different grouping factors while allowing for varied intercept. The r-value reported for these tests signify how correlated the slopes are, with 1 and -1 being perfect correlation. It does not signify anything about the angle of the slope, beyond it being positive or negative (Bakdash and Marusich, 2017).

The data used for multiple linear regression analyses were checked to see that they fulfill the assumptions of linearity, near-normal distributions, and equal variance by plotting values, histograms and residuals and computing Pearson correlation matrices. Presence of collinearity of ecological variables was tested using the variance inflation factor (VIF) using the ‘olsrr’ package in R (Hebbali, 2017). For linear mixed effect models and repeated measures correlations, the independence within random grouping factors are relaxed (Bakdash and Marusich, 2017; Bates et al., 2015), but all other assumptions for linear regression hold and were tested as above. In all cases, for all experiments, all assumptions were met in this study without any transformation needed of any data.

### 2.2.6.1. Normalizing temperature by paired mature forest normalization areas

As thermal imagery is taken at different days with different air temperature and weather conditions, the surface temperature change within fields cannot be compared directly over time from image to image. For Experiment 1 and 3, the relative surface temperature data for each restoration field was therefore computed as the percentage difference between the restoration field’s surface temperature and the surface temperature of its adjacent mature forest normalization area (Figure 2.1).

Mature forests change in biomass and diversity only slowly and should therefore not change nearly as much as the restoration areas from 2002-2018. Surface temperature response from the forest should therefore also remain the same and the forest should have a similar surface temperature on a day in 2002 and 2018 if solar insolation, ground water availability, wind and humidity are the same on those days. To test this assumption, we performed a linear-mixed effect model on all measurements from Landsat satellites of all mature forest normalization areas from 2002 to 2018 with each field of the 31 fields as random grouping factor. Although these measurements depend on the weather of the day they

were acquired, as long as there is no significant decreasing or increasing trend over time the assumption of the mature forest being relatively stable for normalization of temperature over time holds.

#### 2.2.6.2. Statistical analysis and mapping for Experiment 1 - Temperature change over time

To test if relative temperature (dependent variable) changed significantly with time since restoration (independent variable) we performed a linear-mixed effects model, and to test if the change was significantly correlated between fields, we performed a repeated measures correlation. To control for the same fields being measured repeatedly over time, and as each field varied in size and in their landscape context, the restoration field was used as the grouping factor in both methods. Before restoration, all 31 fields were in agricultural production. To determine if there was a change in the pattern of temperature after initial restoration, a linear effects model and repeated measures correlation were performed separately first for all 31 fields for the time from 2002 to initialization of restoration (2006 to 2013, depending on the field), when they were still agricultural fields. The same methods are then used for the same fields after initial restoration (2007-2014) until 2018. A pattern of decrease in temperature only after initial restoration would indicate that the plant growth from restoration is behind the effect.

A map was created of the mean temperature change of restoration areas between 2014 (the first year all restoration had been initialized) and 2018 to test if temperature change can be located and displayed within and between fields. All temperature data in imagery for these years was first subtracted by the temperature of the Backus Woods normalization area (Figure 1), the mean of the temperature of the three images from 2014 was then subtracted from the mean of the 4 images from 2018.

#### 2.2.6.3. Statistical analysis for Experiment 2 - Diurnal temperature variation change with time

In this experiment, we define diurnal temperature variation as the temperature difference between mid-day and late night. First, we compared mean surface temperatures to time since restoration for each restoration field within one image for daytime and nighttime ECOSTRESS and Landsat images in 2018. Second, to test if nighttime imagery showed a common trend in temperature change with time since restoration, we fit a repeated measures correlation and linear mixed-effects model with the imagery from which the measurement was extracted as random grouping factor. The same statistical methods were used separately to test the common trend temperature of daytime imagery.

To test if diurnal temperature variation decreased over time since restoration, we paired each nighttime imagery's temperature data with the nearest unused daytime imagery's temperature data by date, subtracting the nighttime temperature from the paired daytime temperature for the same field to find the diurnal variation. We then performed a repeated measures correlation with percentage diurnal temperature difference as the dependent variable, the field's time since restoration as independent variable

and imagery as grouping factor, to test the significance of the common trend. A linear mixed-effects model with the same factors was used to find the mean slope and its confidence interval.

#### 2.2.6.4. Statistical analysis for Experiment 3 - Comparison of vegetation and temperature

To determine the relative effects of plant species diversity, percentage ground cover, percentage canopy cover, stem count, and years since restoration on surface temperature measurements, a multiple linear regression model was used. Data from 2018 were excluded from this model as canopy cover and stem count was not properly recorded that year.

To test the relationship between relative surface temperature as the dependent variable and vegetation abundance, plant species diversity and time since restoration as independent variables while including 2018 data, we performed two more multiple linear regression models. As proxies for biomass, the first model used total percentage vegetation ground cover and the second used the percentage difference of NDVI between the restoration fields and their normalization area. To test for a potential difference in effect of native versus exotic plant species, we performed a simple linear regression of native and exotic diversity against relative temperature. To test which variables have the most statistical explanation power we compared the Akaike Information Criterion – corrected (AICc) using the ‘dredge’ function of the ‘MuMin’ package in R (Barton, 2020). The AICc compares different combinations of independent variables ability to statistically explain change in the dependent variable, penalizing models with higher number of variables (Brewer et al., 2016).

### 2.3. Results

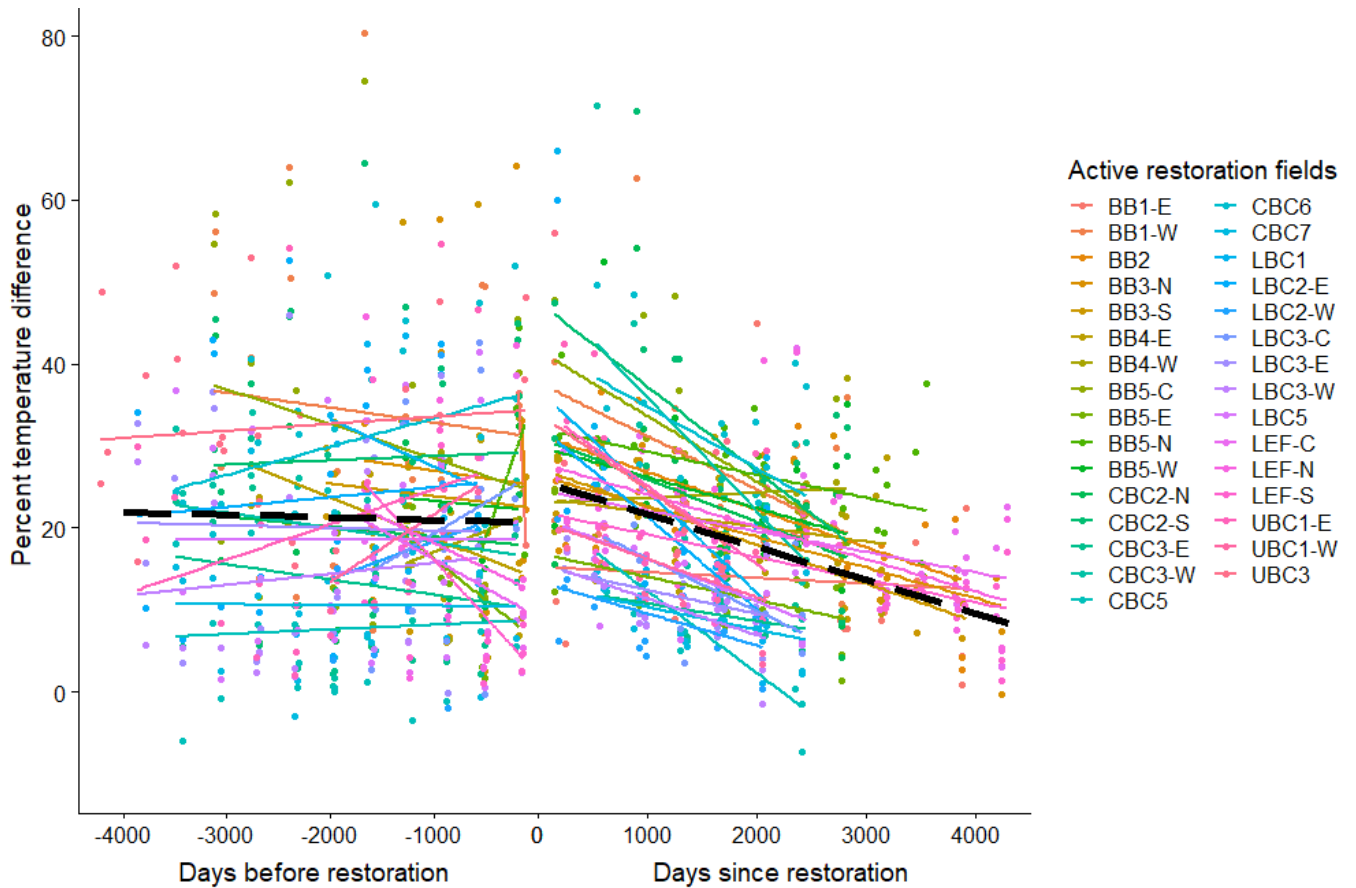
The linear-mixed effects model performed shows no significant change (95 % CI: -0.003 to 0.100 C°/year, N = 1148) of surface temperature in the mature forest normalization areas from 2002 to 2018. This indicates that the normalization areas are appropriate stable comparisons for the restoration areas in Experiment 1 and 3.

#### 2.3.1. Results of Experiment 1 - Temperature change over time

For the experiment comparing relative surface temperature of 31 restoration fields over time between 2002 and 2018, 1109 measurements of mean land surface temperature were included; 599 of those measurements were taken after restoration was initialized. On average, each field had 34 measurements of land surface temperature. All data are available in Hamberg (2019). The mean surface elevation of all selected areas was 201 m above sea level with a mean difference of 1.1 m (SD: 4.0 m, maximum: 9.9 m) between restoration fields and their paired normalization areas. All paired normalization areas shared major soil type with its nearby restoration area.

When comparing the relative surface temperature of the pre-restoration agricultural fields, we found no evidence of a significant and consistent change. The linear mixed effects model comparison of relative temperature over time grouped by field was -0.1 percentage point (p.p.) per year, with the possibility of no change being within the 95% confidence interval (-0.57 to 0.35). The repeated measures correlation was low ( $r = -0.017$ ) and not significant ( $p = 0.711$ ), meaning that there was no common pattern of change in temperature over time among the restoration fields (Figure 2.5). Pre-restoration agriculture fields showed a high variance in surface temperature between years, which we suspect was due to changes between years and time of imagery in which fields were fallow, planted, or already harvested. Temperature variation may also be due to what crop was grown, and in the amount of fertilizer and irrigation used.

After initial restoration, 30 of 31 restoration fields decreased in temperature relative to their paired normalization areas over time. BB4-W was the only field increasing in temperature over time (Figure 2.5). The mean decrease in temperature, as measured through the linear mixed-effects model, was 1.5 p.p. per year (95% CI: -1.73 to -1.26) and the repeated measures correlation of slopes in between the different fields was significant ( $p < 0.001$ ) and moderately strong ( $r = 0.36$ ).

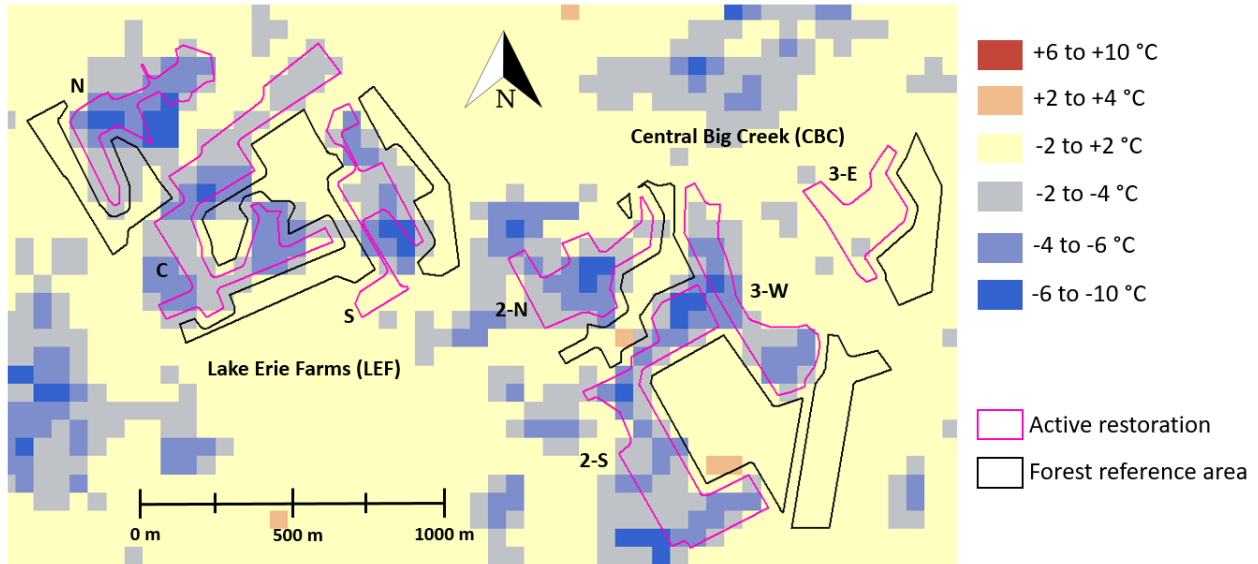


**Figure 2.5: Best fit linear regression models for each restoration field (N = 31) in our study for temperature difference against days before and days after restorations using Landsat 5, 7 and 8 and ECOSTRESS data from 2002 to 2018. The black dashed line shows the average slope and intercept from the linear-mixed effects model.**

The average restoration area was 25.6 % warmer than its normalization area when restoration began, falling to 7.7 % warmer 12 years later. To put this into context, if the surface of one of the mature forest normalization areas experienced a 25 °C day in 2007, and another one in 2018, we would expect its paired restoration area to have been 31.4 °C in 2007 and 26.9 °C in 2018, a 4.5 °C drop.

The apparent increase in temperature the first year after restoration can be explained by the considerably higher biomass of fertilized and irrigated agricultural fields, as compared to the restoration fields in the first year after restoration when plants have not yet had time to establish and spread. The difference in mean relative temperature between fields could be due to differences in field size, shape, elevation, slope and different surrounding land-use matrix. Differences in the angle of the slope in figure 2.5 between different fields can be due to differences in the ground water levels and weather conditions of the first few year of restoration, as restoration was initialized in stages between 2006 and 2013. Variation

in quality of the seeds, mainly harvested the year before from wild populations, may also have affected the speed of restoration.



**Figure 2.6: Map of mean temperature change between 2014 and 2018 imagery. All imagery was subtracted against Backus Woods mature forest normalization area (Figure 1). Active restoration areas are outlined in magenta and paired normalization areas in black, not including the 20 m internal buffer. Full map in Appendix 2B.**

The temperature difference map (Figure 2.6) shows that most restoration areas have decreased in temperature over time 2014-2018. Pixels near the edge of the restoration fields generally show a less steep decline than those in the center, as the edge pixels are more affected by the stable forests near them. Field CBC 3-E exhibit almost no decline and may be an ideal candidate for follow-up monitoring to find out why that is. Areas decreasing in temperature outside restoration areas may be agricultural fields that have changed crops, fertilizer or irrigation intensity, or have gone from being fallow to active.

### **2.3.2. Results of Experiment 2 – Diurnal temperature variation change with time**

Comparing diurnal temperature change in the space-for-time substitution experiment, we first analyzed temperature change for 2018 data separately for night and day. The three nighttime temperature measurements from ECOSTRESS all showed an increasing trend over time since restoration. The linear mixed effects model of the nighttime measurements showed a mean increase in nighttime temperature of 0.20 °C per year (95% CI: 0.07 to 0.32) and the repeated measures correlations for them showed a significant ( $p=0.004$ ), and moderately strong ( $r=0.38$ ), correlation between the measurements. Taken together this provides evidence that nighttime temperatures of restoration areas were increasing at a

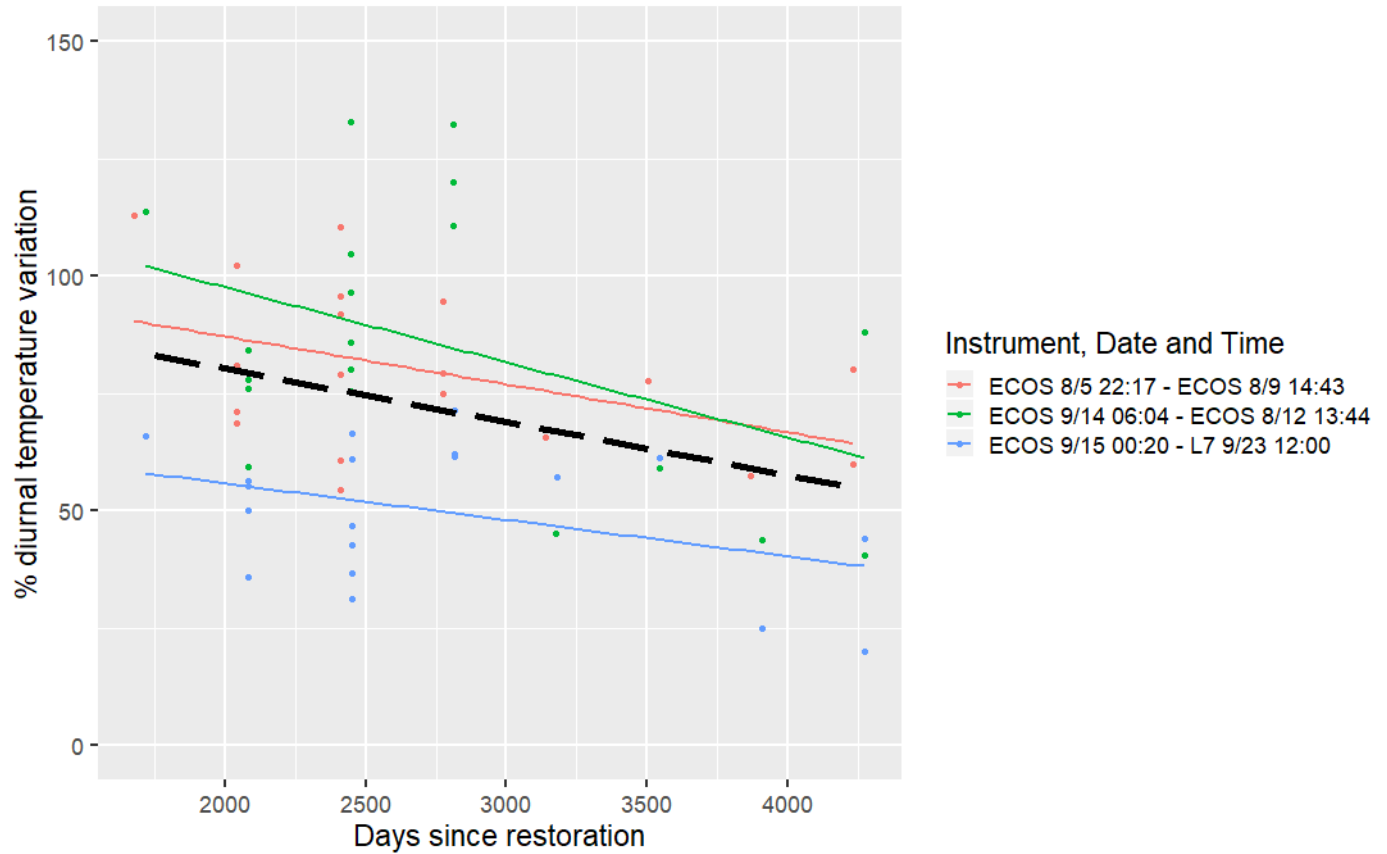
significant, and relatively consistent, rate with time since initial restoration, independent of which imagery the data was taken from.

While the linear mixed-effects model using the two ECOSTRESS and two Landsat daytime measurements showed a mean decrease of 0.42 °C per year, the decrease was not significant (95% CI: -0.85 to 0.01) and the repeated measures correlation was relatively weaker than the nighttime measurements ( $r=-0.22$ ) and not statistically significant at the 95% confidence level ( $p=0.058$ ).

To study change in diurnal temperature variation, we paired daytime and nighttime readings together, removing the Landsat 8 observation taken in June, as it was the furthest away in the season from the others. The linear mixed-effects model showed a decrease of 4.14 p.p. per year in diurnal temperature difference since restoration (95% CI: -6.68 to -1.59) (Figure 2.7). The repeated measures correlation showed that the slopes were significantly ( $p=0.002$ ), and moderately strongly correlated ( $r=0.40$ ). For the time studied, the mean diurnal difference fell from 86 % 4 years after restoration, to 50 %, 12 years after restoration. As an example, a field restored in 2006 with a 28 °C day and 15 °C night in 2010 would be expected, on a similar day in 2018, to have a 25 °C day and a 17 °C night.

Comparing these results with those of Experiment 1 shows that diurnal temperature change could decrease faster than just daytime temperature change. The variation that was not explained by time since restoration may be explained by remaining differences in size, shape, elevation and the seed-mix viability, weather and ground water levels in the first few years after initial restoration.





**Figure 2.7: Percent diurnal temperature difference for paired data over time since restoration. Separate linear regression lines are shown for each pair of measurements and the mean slope coefficient for the linear mixed-effects model is shown as a dashed black line. Legend shows instrument (ECOS – ECOSTRESS, L7 – Landsat 7) of paired day-night imagery.**

### 2.3.3. Results of Experiment 3 - Comparison of vegetation and temperature

All Lake Erie Farms restoration areas were found to have a mean altitude between 197 and 203 m and a mean slope between 1.78 and 2.81°. To compare thermal and ecological data over time for the Lake Erie Farms property we performed three separate multiple linear regression models based on theory, and two more based on highest statistical explanation power as per AICc. For the first model performed, for the period 2007-2017, using all in-situ measured independent variables (Table 2.2) there was a mean decrease of relative temperature of 5.06 p.p. for each one increase in the effective species number, when controlling for all other factors (Table 2.2A). There was also a 1.00 p.p. mean increase in relative temperature per year after restoration that was not explained by other factors and was significant at the 95% confidence level (p=0.03).

Our proxies for biomass and shade - ground cover, canopy cover and number of woody stems do not significantly affect temperature. This analysis provides evidence that the relative temperature of

restoration areas was mainly affected by species diversity. When translated on to a day with 24 °C surface temperature in the mature forest normalization area, an increase of one in effective species number for plant diversity, without an increase in ground cover, canopy cover or stem count would decrease temperatures in the average restoration field from 30 °C to 29.7 °C. The model with the lowest AICc value (Table 2.2B), highest statistical explanation power (lowest AICc) only include plant species diversity, % ground cover, and years since restoration as independent variables (Table 2.2B).

**Table 2.2: Two multiple linear regression models with relative land surface temperature as the dependent variable were performed for the period 2007-2017. Model A includes all in-situ measured independent variables. Model B includes the variables that has the lowest AICc of all combinations of in-situ measures and the relative NDVI.**

	Relative surface temperature – 2007-2017	
	(A) All variables	(B) Lowest AICc
Species diversity	-5.06, p < 0.001	-6.25, p < 0.001
% Ground cover	-0.09, p = 0.24	-0.12, p = 0.07
% Canopy cover	-0.13, p = 0.48	
# Woody stems	-0.25, p = 0.80	
Relative NDVI		
Years since restoration	1.00, p = 0.03	0.68, p = 0.09
Constant	34.11, p < 0.01	33.41, p < 0.01
Observations	60	60
R <sup>2</sup>	0.44	0.41
AICc delta	6.53	0
Adjusted R <sup>2</sup>	0.39	0.39
Residual Std. Error	7.82 (df = 54)	7.83 (df = 57)

Variance inflation factor (VIF) for canopy cover (3.90) and number of woody stems (5.67) was high in the 2007-2017 model, indicating a high amount of collinearity between the two factors. This makes sense as all canopies have one or more stems. However, removing either canopy cover or number of woody stems from the model does not affect the significance or coefficient of any of the remaining variables noticeably. No VIF for any other variable in any other model in Experiment 3 was > 3.

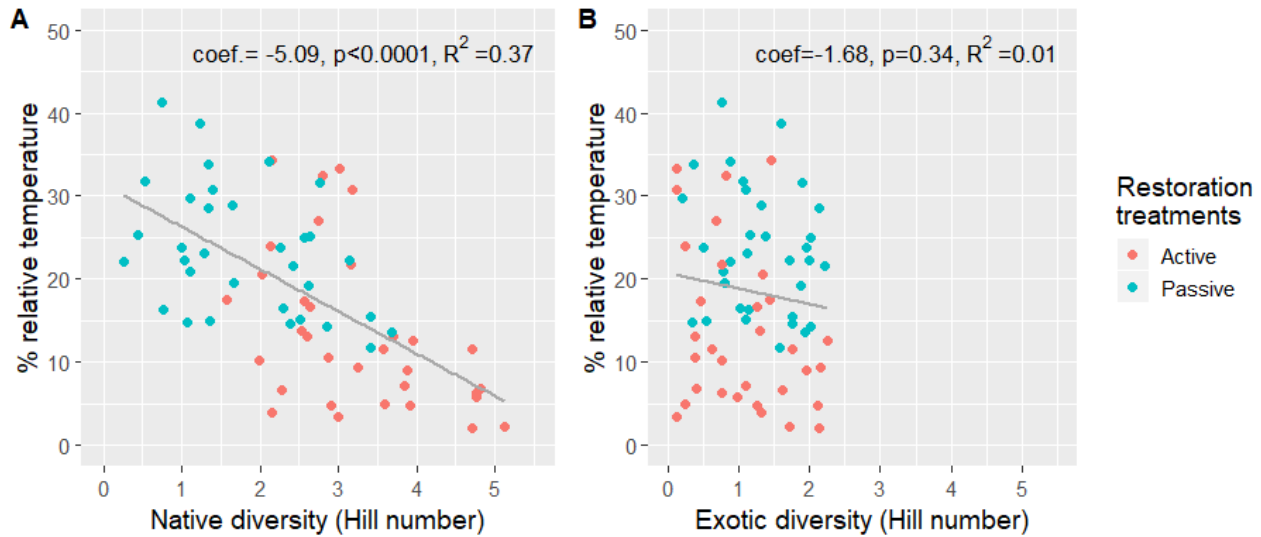
Two separate models were fitted for 2007-2018 (Table 2.3). These models excluded canopy cover and stem count as these variables were not properly recorded that year. The first model (Table 2.3A) includes ground cover as the only proxy for biomass, with similar results to the model in Table 2.2B. Using relative NDVI as the proxy for biomass instead of ground cover, showed similar results, with relative NDVI not significantly affecting relative temperature (Table 2.3B). The model using cover as

biomass proxy (Table 2.3A) was found to have the lowest AICc when testing all combinations of the independent variables presented in Table 2.3.

**Table 2.3: For 2007-2018 two regressions models are fitted against the dependent variable relative surface temperature : (A) uses percent ground cover as the independent variable proxy for biomass and (B) uses percent difference of NDVI between restoration area and normalization area as the proxy for biomass.**

	Relative temperature	
	(A) Cover as biomass proxy	(B) NDVI as biomass proxy
Species diversity	-5.08, p < 0.001	-4.96, p < 0.001
% Ground cover	-0.12, p = 0.07	
Relative NDVI		-0.18, p = 0.19
Years since restoration	0.66, p = 0.07	0.75, p = 0.08
Constant	36.47, p < 0.01	23.22, p = 0.01
Observations	66	66
R <sup>2</sup>	0.40	0.39
Adjusted R <sup>2</sup>	0.37	0.36
AICc delta	0	1.73
Residual Std. Error	7.82 (df = 62)	7.92 (df = 62)

Separating the diversity measurements of native and exotic plant species of both passive and active restoration treatments through linear regressions for 2007-2018 showed a significant ( $p < 0.0001$ ) decrease in temperature (-5.09) with an increase in native species diversity (Figure 2.8 A). Conversely, relative temperature was not significantly ( $p = 0.34$ ) affected by exotic species diversity and the slope coefficient (-1.68) was also smaller (Figure 2.8 B). This indicates that native species may have a stronger decreasing effect on temperature in restoration areas than exotic ones. Caution should however be exercised when comparing the relative effect of native and non-native plants on temperature for our plots, as the plots had a maximum of 2.3 in effective number of species of exotics, compared to 5.1 for natives.



**Figure 2.8: Linear regression of relative temperature with native and exotic species richness and diversity.** Each point represents an active or passive restoration plot and its native or exotic diversity as compared to its relative temperature as compared to its paired normalization area.

## 2.4. Discussion

We find compelling evidence that relative surface temperature decreased with an increase in plant species diversity, beyond what was explained by proxies for biomass or shade. We propose that it is the new functions, structures, phenology and trophic interactions that were introduced with new species that decreases temperature. This aligns with the idea that an increase in biodiversity, structural and functional complexity and ecosystem self-organization should increase use of available energy, also called exergy, and decrease reflected and reemitted daytime heat (Fraser and Kay, 2004; Jorgensen and Svirezhev, 2004; Kay, 2000; Maes et al., 2011; Schneider and Kay, 1994). More simply, transpiration is the main method of cooling for plants (Michaletz et al., 2015) and diversity of structure and function increases transpiration under more conditions (Baldocchi, 2005). Structures and functions that increases available energy use and potential transpiration, lowering vegetation surface temperature, include phototropism, stomatal arrangement, xylem structure, and optimized overlapping leaf arrangements (Baldocchi, 2005; Schneider and Kay, 1994).

Although our comparison of native and non-native species should be interpreted with caution, there is an argument for why native plants may cool more than non-natives could. Native species may be better adapted to the local site conditions, having structures and functions that are especially effective at using energy in the local environment. Native prairie-grasses have deep roots (Hayes and Seastedt, 1987; Nippert et al., 2012), allowing them to take up water in the local sandy soil and continue to transpire and photosynthesize, when non-native ruderals, such as the creeping *Poa annua*, would be unable to. Many of

the same native prairie grasses also employ the C4 photosynthesis pathway which allows for more efficient energy usage at high temperatures and low nitrogen levels, which was the case for our study sites (Howe, 1994; Nippert et al., 2012). Native tallgrass prairie plants can also have more developed relationships through mycorrhizae (Hartnett and Wilson, 1999) and may more effectively provide habitat and food to native fungi, insects and other organisms that they have co-evolved with; and this cycles, and makes available, nutrients needed for plants to build structure (Loreau, 1995). There is also evidence that leaves have evolved directly to regulate their temperature for climate conditions within its natural range (Michaletz et al., 2015). These examples provide a potential explanation for our findings that increases in native species increases the system's ability to use available energy, manifesting as lower land surface temperature. Conversely, many of the more common exotic species in the study plots are considered invasive in the area and are generally ruderals (e.g., *Meliolotus alba*, *Poa annua* and *Vicia cracca*). These exotic invasive ruderals rely on spreading quickly and modifying the environment (e.g. by shading) to their favor, while potentially being less efficient at using available energy than native and non-ruderal plants, especially in varying conditions and under moderate stress (Grime, 1977; Norris et al., 2012).

Overall, relative temperature has fallen over the 12 years since restoration. Due to cost and time restrictions, vegetation sampling has only been conducted on the Lake Erie Farms property. However, all 31 fields have similar soil, similar history of agricultural use, similar landscape context and all sown with a similar seed-mix. There has also been no indication of any large-scale disturbances or failures in the project (personal communication, Liv Monck-Whipp, 2019). For these reasons, it is likely that both biomass and plant species diversity would be increasing for most of the 31 fields as it did at the Lake Erie Farm fields; and, based on the results from the third experiment, the decrease in relative temperature should be an indicator of an increase in plant diversity in these fields. Diurnal temperature variation also decreased with time since restoration, with restoration areas becoming warmer at night. Ulanowicz & Hannon (1987) posited that this should happen as the ecosystem increases in structures which hold on to more of the heat for a longer time, even into the night, through physical heat storage and chemical bonds.

Kutsch et al. (2001) criticized attempts at measuring ecosystem development, organization or complexity through temperature, as they found that a monoculture cornfield was as cool as a mature beech forest. However, they did not consider the input of fertilizer to the cornfield or that the two systems were very different from each other. In our study, we do not assume that if we compared two ecosystems in different climate, soil or nutrient availability, the one with higher plant species diversity would necessarily be colder than the other, less diverse, ecosystem. Alvar ecosystems, for example, can have very high plant species diversity while being quite warm in the day, due to lack of soil and water availability (Tomlinson et al., 2008). We argue that, all other factors equal between two ecosystems, higher plant species diversity should lead to lower temperatures, which is still useful in relative

comparisons of multiple fields in a restoration project. Terrestrial ecosystems are open and complex thermodynamic systems and even with the controls and assumptions given, we expected and detected variation that could be accounted to landscape context, ground water change and precipitation before the imagery was acquired. Acknowledging the caveats and limitations above, the evidence from the current study lend support to relative land surface temperature readings as a potential indicator of change in plant species diversity in restoration projects, at least for the first decade of restoration.

Many earlier studies of thermal energy budgets in ecosystem focus on measures of entropy, exergy (e.g., Kutsch et al., 2001; Lin et al., 2018) or measures of temperature buffer capacity (Aerts et al., 2004; Luvall et al., 1990). Energy budget methods may be less affected by other energy flows, such as wind, but require expensive and location-specific equipment such as flux-towers, multiple aircraft flyovers or an arrangement of thermometers and net-radiometers. Due to its free and public availability, large extent and coverage, and spatial resolution that allows for imaging restoration sites down to one hectare, Landsat and ECOSTRESS thermal instruments could also be used to identify problem areas and emerging issues in large scale, remote or fragmented restoration projects. Algorithms could potentially be developed to compare imagery over time, to find spots where temperature is not decreasing as fast as their surroundings – thus, guiding professionals on the ground to the problem spot for more in-depth diagnosis and adaptive management. To operationalize surface temperature as a useful indicator of plant species diversity and restoration progress more experiments in other locations and ecosystems are needed to establish when and how plant species diversity affects ecosystem surface temperature. If the linear trend continues, then restoration areas will be as cold as the forest in another decade, which is improbable. Continued longitudinal research can give us insight into the curve of temperature decrease, allowing for stronger predictions of restoration progress.

## **Chapter 3. Topsoil transfer to a reforestation site and gravel pit increases native forest plant diversity**

### **3.1. Introduction**

Deforestation and forest degradation due to construction projects and extraction of resources lead to decreases in biodiversity and ecosystem services including climate mitigation, biodiversity, soil retention, pollination, and water filtration (Chazdon, 2008; Costanza et al., 2014; Thompson et al., 2012). To mitigate the loss of forests and their services, projects can be encouraged or mandated to reforest, afforest or restore nearby areas (Aerts and Honnay, 2011; Holl and Howarth, 2000; Poulton, 2014).

In this study, the terms ‘reforestation’ (of areas that have historically held forests), and ‘afforestation’ (of areas that have not held forests), are used for projects where only trees are planted, which is the method most commonly employed by commercial and industry-led projects (Cunningham et al., 2015; Mansourian et al., 2005; Stanturf et al., 2014). In contrast, the term ‘restoration’ is used for methods that aim to re-establish more of the full biodiversity, function, structure, and integrity of the forest ecosystem (Aerts and Honnay, 2011; DeLuca et al., 2010; Dudley et al., 2005). Methods of forest restoration include ‘directed succession’ seeding of mid and late succession herbaceous species and trees, as seen with the tallgrass prairie and oak woodland seeding in Chapter 2 (e.g. Henry et al., 2019; Mansourian et al., 2005). The drawback when only planting trees, and even in part when seeding mixed woody and herbaceous plant species, is that locally adapted understory species and soil biota, needed for a functioning forest ecosystem, would still have to colonize from outside the project area, which can take over a century (Harmer et al., 2001; Vellend, 2003).

A method of forest restoration that can increase both locally adapted forest understory vegetation and soil conditions rapidly is direct transfer of topsoil (e.g. Buckley et al., 2017; Ferreira et al., 2015; Hall et al., 2010; Hietalahti et al., 2005; Koch, 2007). The topsoil of forest ecosystems contains many of the components needed to recreate a forest, including plant propagules (Fowler et al., 2015), mycorrhizae (Harley, 1989), fungi, soil microorganisms, nutrients, and the organic detritus that accumulates and decomposes on its surface (Raven et al., 2003 chap. 30). While there are examples of forest topsoil transfer in academic literature, it is still an uncommon enough method that it was not mentioned in a large review of reforestation and forest restoration methods (see Stanturf et al., 2014).

For restoration by topsoil transfer, topsoil is excavated from the donor forest where the construction or extraction (quarrying or mining) project will take place. It is then moved to a nearby recipient area suitable for forest restoration. The main limitation of this method is that transfer of forest

topsoil is generally only considered when a nearby forest is to be removed (e.g. Buckley et al., 2017; Fowler et al., 2015; Koch, 2007), presumably because the excavation of the topsoil will by necessity cause as much or more destruction in the donor forest as it will help to restore in the recipient area.

Quarrying and mining projects often transfer topsoil and deposited it on bare rock or crushed mine tailings to restore or create different types of ecosystems (e.g. Ferreira et al., 2015; Hall et al., 2010; Rodgers et al., 2011; Sheoran et al., 2010; Waterhouse et al., 2014). Although, if there are no parts of the quarry or mine that is already decommissioned, where the topsoil can be transferred to, it means that the topsoil has to be stored in berms. Storage is not ideal, as many of the components of the topsoil are alive, and can die or go dormant (e.g. Abdul-Kareem and McRae, 1984; Boyer et al., 2011; Golos et al., 2016; Rivera et al., 2012). Depending on soil depth, temperature, and humidity, one study showed that up to 90 % of plant propagules can become unviable in a year (Golos et al., 2016).

An alternative to moving the topsoil to the quarry or mine-floor, or store it, is to move it to nearby farmland in or near forests. In many parts of the world farmland, especially small and marginal plots of farmland, were being abandoned at an increasing rate throughout the last century, reaching over 200 million ha by 1992 (Cramer et al., 2008; Ramankutty and Foley, 1999). This abandonment opens up possibilities of restoration if the land is not developed in other ways (Hobbs and Cramer, 2007). While there are large possibilities, there are so far few examples of forest topsoil transfer to farmland in scientific literature.

In one of the few well-documented cases of forest topsoil transfer to farmland, the topsoil of the recipient site – a former pasture in southern England - was first removed before the transferred forest topsoil was deposited and young trees were planted (Buckley et al., 2017; Helliwell et al., 1996). The pasture topsoil was removed to make the soil's chemical profile more like the forest topsoil in terms of soil nutrients and to remove the seedbank of ruderal field species. Even so, with the open conditions and young planted trees after topsoil transfer, ruderal species did re-emerge and out-compete much of the transferred forest understory vegetation in the British study. Only when the tree canopy closed, 20 years later, did the ruderal field species diminished, and remaining forest understory vegetation return (Buckley et al., 2017). To overcome the issue of lack of shade, which can aid ruderal field species (e.g. Valladares et al., 2016), Zhao et al. (2020) added artificial shade, through shade cloth, as well as weeding of ruderal species (herbaceous and liana species in their case) to their forest topsoil transfer experiment. The shade did improve the species richness and growth of forest plants in their experiment.

Another option for overcoming the issue of lack of shade leading to ruderal species takeover is to deposit the forest topsoil on farmland where reforestation through tree-planting has already been done. This method would preclude topsoil removal, as it would damage tree-roots, but may still lower the



competition from ruderal species by providing faster canopy closure and shade. This option is what will be tested in this study.

Not only could the shade of the already planted trees potentially aid the competitive ability of the understory species in the forest topsoil, but the topsoil addition may potentially increase tree growth and survival. The planted trees may benefit from lowered competition from ruderal field species as the added topsoil may lower germination from the original topsoil of the recipient site (Bliss and Smith, 1985; Fowler et al., 2015). Trees may also benefit from increasing soil-water retention (e.g. Douterlungne et al., 2018), changed soil nutrient conditions, mycorrhizae, and microbes in the topsoil (Hietalahti et al., 2005, p. e.g.; Morris et al., 2014). However, adding topsoil on top of already planted trees could also potentially decrease growth and survival: Increasing soil depth decreases root respiration (Makita et al., 2011; Pregitzer et al., 1998), new soil may contain tree pathogens (e.g. Reinhart et al., 2005) and machines may damage trees directly at the time of soil transfer.

Along with topsoil transfer, other parts of the forest structure can also be moved, including coarse woody debris. Woody debris can increase habitat heterogeneity, habitat for microbes, insects, and other small animals, nutrient availability, and water retention (McIntyre et al., 2015, p. e.g.; Shoo et al., 2014). A study tested the addition of woody debris together with forest topsoil transfer in a boreal forest with large amounts of peat and found that the combination of these two treatments increased both native and non-native plant species diversity and cover (Brown and Naeth, 2014).

The goal of this study was to test if direct transfer of forest topsoil, along with additional shade and/or woody debris treatments, could speed up and alter the successional trajectory of a reforestation site, towards a native and diverse forest ecosystem. The relative improvement of the reforestation sites on former farmland was compared against controls with no added topsoil and with topsoil transfer to an abandoned gravel pit. In line with common usage in restoration ecology literature, the relative success of restoration and its trajectory is measured through plant growth, species diversity, and community composition (Ruiz-Jaén and Aide, 2005; Ruiz-Jaen and Aide, 2005; Wortley et al., 2013). In this case, a successfully improved successional trajectory is specifically measured as (1) an increase in native forest plant species diversity in the recipient sites towards that of the donor site, (2) the plant community in the recipient sites becoming more like that of the donor forest, and (3) an increase, or at least lack of decrease, in the growth of planted trees.

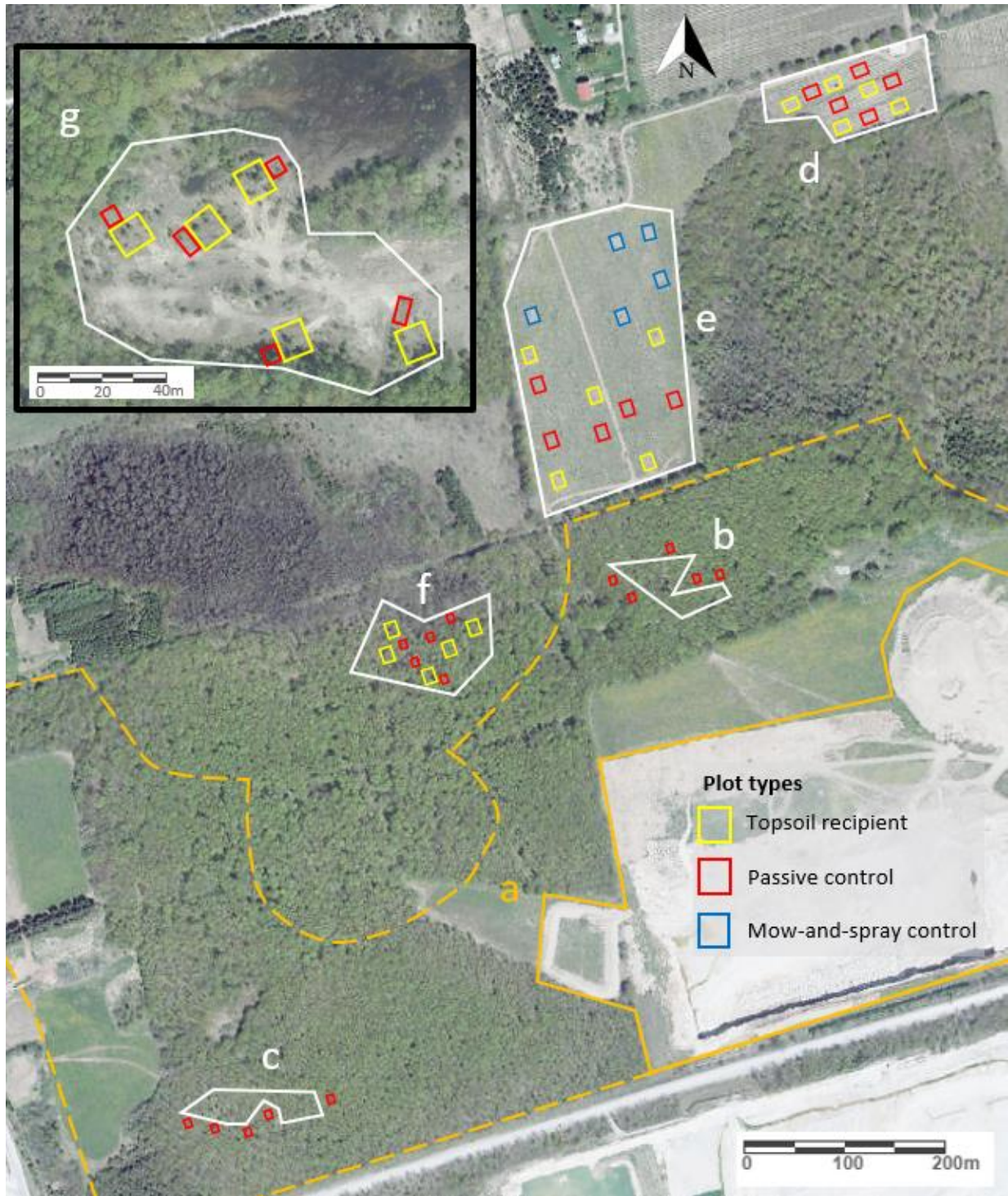
## 3.2. Methods

### 3.2.1. Study Area

All sites in this study were located within Clearview Township, Simcoe County, Ontario. Three sites were selected to have experimental plots which received topsoil – two of these were former farmland that had been planted with trees in 2015 (Figure 3.1e) and 2016 (Figure 3.1d), and the third was an abandoned gravel pit site (Figure 1g). Topsoil was excavated from a northern (Figure 1b) and a southern (Figure 3.1c) forest donor area within the boundaries of a planned quarry expansion (Figure 3.1a). To control for the effect of topsoil excavation and mixing, without the change in ecosystem, ‘lift-and-drop’ control plots were also prepared in a protected forest site (Figure 3.1f).

Except for the gravel pit, all sites were located within the quarry company properties, within 1.3 km of each other (44.39° N, 80.25° W), on the Niagara Escarpment at 510-535 m asl. The 2015 reforestation site was 5.1 ha in size, while the 2016 reforestation site was 1.0 ha. The donor forests were both hardwood forest dominated by sugar maple (*Acer saccharum*) with a closed canopy since at least 1954 (Hunting Survey Corporation Limited, 1954), both donor sites were within the planned extent of a quarry expansion. The protected forest site was located approximately 50 m outside of the planned quarry expansion and was shielded from future quarrying as it was within the buffer zone for the provincially protected *Asplenium scolopendrium* (Hart’s tongue fern) and *Juglans cinerea* (butternut tree) species.

The gravel pit was located 6.5 km away to the north-east (44.43° N, 80.18° W), below the Niagara Escarpment at 235 m asl. The full gravel pit is 14.3 ha in size, but the section of the gravel pit site used for the experiment was 0.5 ha in size. The section of the gravel pit site used for the experiment had been inactive for at least 28 years (1989) before the study, according to the previous landowner (Stephen Sage, Chief Administrative Officer, Clearview Township – Personal Communication, July 14-16 2020). From aerial imagery, the gravel pit was established sometime between 1954 and 1984 (Hunting Survey Corporation Limited, 1954; USGS, 2019). Most of the site was devoid of vegetation at the beginning of this study, presumably due to lack of topsoil after gravel excavation ended. Some smaller patches (5 - 20 m<sup>2</sup>) within the site did have ground vegetation and trees, including *Pinus sylvestris*, *Juniperus communis*, and *Salix sp.* The gravel pit site was selected based on distance to the donor site, accessibility, and landowner permission.



**Figure 3.1: Main map (a-g) of topsoil donor and recipient sites near or within the planned expansion (a – orange, dashed) of the current (as of 2018, a – orange, solid) Duntroon quarry located on the Niagara Escarpment, Clearview Township, Ontario. Topsoil was excavated from northern (b) and southern (c) donor forest. Topsoil was transferred to plots within the 2016 (d) and 2015 (e) post-agricultural reforestation sites. Topsoil was also excavated and deposited in plots within the protected forest site (f). Insert map shows the gravel pit topsoil recipient site (g) 6.5 km away from the other sites. All plots are to scale within each of the maps.**

The reforestation sites are referred to as such (as opposed to afforestation) in this study as their surroundings are mainly mixed deciduous and coniferous forest, and that is the type of ecosystem the area would over time return to without human intervention. The reforestation sites were on former farmland, last in production in 2014, and planted with young trees in spring 2015 and 2016, as part of a larger (51.5 ha) industrial reforestation project. The reforestation project was undertaken to mitigate impacts on ecosystem services and to satisfy legal requirements in order to expand quarry operations nearby (Charlton and Hims, 2013).

Before tree planting, the reforestation sites were plowed and seeded with a non-native commercial ground cover seed-mix (*Lolium multiflorum*, *Secale cereale*, *Avena sativa*, and *Trifolium repens*). Every 10<sup>th</sup> row was a windrow created from soil, stumps, boulders and brush-piles from nearby cleared forest areas; no trees were planted in the windrows. A mix of 20 deciduous and coniferous tree species of different size-categories were planted in the reforestation sites (Table 3.1), based on inventories of nearby forests (Charlton and Hims, 2013). Tree tubes and support stakes were installed after planting and the reforestation sites were managed through herbicide spraying, mowing, irrigation, tree tubes, and rodent trapping for two years.

The two reforestation sites used in this study differed not only in size but also in that the 2015 site had a year of no mowing, spraying, or irrigation before topsoil transfer and one extra year of tree-growth, as compared to the 2016 site. The two reforestation sites were selected among all reforestation areas based on accessibility from donor to recipient sites.

**Table 3.1: Mean number of trees per ha planted in the two reforestation sites in 2015 and 2016 by species and size-category, as well as totals planted per species per ha for both 2015 and 2016.**

	2015 plantings by height (cm)			2016 plantings by height (cm)			Totals per species per ha
	10 - 60	60-150	150-250	10 - 60	60-150	150-250	
<i>Abies balsamea</i>	122			76			99
<i>Acer rubrum</i>		46	51	60		19	88
<i>Acer saccharum</i>		46	131		64	383	312
<i>Betula alleghaniensis</i>		28		64			46
<i>Betula papyrifera</i>		46	64		60	74	122
<i>Carya cordiformis</i>		46		86			66
<i>Fagus grandifolia</i>				13			6.5
<i>Juglans nigra</i>		46		71			58.5
<i>Larix laricina</i>	122			64			93
<i>Picea glauca</i>	103			218			160.5
<i>Pinus resinosa</i>	128						64
<i>Pinus strobus</i>	128			218			173
<i>Populus tremuloides</i>		46	86		26	77	117.5
<i>Prunus serotina</i>		46	36	12	38		66
<i>Quercus macrocarpa</i>		46		90			68
<i>Quercus rubra</i>		46	71	64		32	106.5
<i>Thuja occidentalis</i>	100			77			88.5
<i>Tilia americana</i>		42			58	85	92.5
<i>Tsuga canadensis</i>	128			90			109
<i>Ulmus americana</i>		46		64			55

### 3.2.2. Experimental setup

Approximately 525 tonnes of topsoil (A-horizon) was excavated from the southern (Figure 3.1b) and northern (Figure 3.1c) donor forests using mini-excavators (Takeuchi TB-138 and TB285, Takeuchi Mfg. Co., Ltd.) in October and November 2017. All matter on top of the topsoil was transferred along with the soil; this included leaves, woody debris, small shrubs and trees (approximately below 2 m in height), rocks, moss, and fungi. The gravel pit (Figure 3.1g) received topsoil exclusively from the southern donor forest, while the reforestation sites (Figure 3.1d-e) received a majority of its topsoil from the northern donor forest.

Topsoil was excavated and transferred after senescence of most plants in late fall, as one study (Craig et al., 2015) has shown that this timing minimizes plant mortality, and increases the speed of recovery of vegetation cover, compared to excavation and transfer during spring. Topsoil was excavated at a minimum distance of 20 m from the edge of the donor forest. This distance was selected as a compromise between reducing forest edge effects while allowing for machinery access. For all plots in

the reforestation sites and the protected forest site, the topsoil was excavated, transferred, and spread on the same day. Due to weather conditions limiting equipment access, topsoil for the gravel pit plots had to be stockpiled for 2-3 days before being spread. Further details on the methods of topsoil excavation and transfer are available in section 1 of Appendix 3A.

#### 3.2.2.1. Plot setup

Forest topsoil was transferred to, and loose-tipped inside 5 ‘topsoil recipient plots’ measuring 12.5 x 10 m in each for the three recipient sites (Figure 3.1 & Table 3.2). Topsoil was added and spread until it reached a depth of approximately 20 cm, similar to the depth of the A-horizon in the donor sites. The plot-size was a compromise between minimizing edge effects, by having larger plots, while using the resources available to have 5 plot replicates at each site. Loose-tipping, as opposed to moving soil in intact turfs, has been recommended for forest soil transfer as tree roots make intact turf moving technically difficult (e.g. Buckley et al., 2017; Hietalahti et al., 2005) and the difference in impact on vegetation sprouting of the two methods has been shown to be small (e.g. Craig et al., 2015).

Before adding the forest topsoil to plots in the two reforestation sites the plots were prepared by mowing between tree-rows and then spraying with a glyphosate herbicide. Mowing and spraying were done 1-2 weeks before topsoil transfer took place, to suppress ruderal field species. Due to the mostly bare ground in the gravel pit, no mowing or spraying was done for any plots in that site.

Along with the topsoil recipient plots, each site also had 5 passive control plots where no treatment was done (Table 3.2). These were quarter-plot sized (6.25 x 5 m) in terms of vegetation sampling, but 12.5 x 10 m for planted tree sampling in the reforestation sites. Additionally, 5 more 12.5 x 10 m plots in the 2015 reforestation site were mowed and sprayed but received no topsoil. These 5 ‘mow-and-spray control plots’ were used as controls for the effect of mowing and spraying.

**Table 3.2: Site, plot, and sub-plot setup. For passive control plots in the reforestation sites, only the northeast corner was sampled for vegetation diversity, but the full plot was used for tree growth measurements. TRo – Topsoil recipient only, MSO – Mow-and-spray only, WD – additional woody debris, DS – additional dogwood shrubs + woody debris, SS – additional shade shelter + woody debris.**

Site	Number of topsoil recipient plots (sub-plot treatments)	Number of mow-and-spray control plots (sub-plot treatments)	Number and size of passive control plots
Reforestation 2015	5 ( TRO, WD, DS, SS)	5 ( MSO, WD, DS, SS)	5 (12.5x10 m)
Reforestation 2016	5 ( TRO, WD, DS, SS)	-	5 (12.5x10 m)
Gravel Pit	5 (TRO, WD, DS, SS)	-	5 (6.25x5 m)
Southern Donor Forest	-	-	5 (6.25x5 m)
Northern Donor Forest	-	-	5 (6.25x5 m)
Protected Forest	5 ( TRO, WD, DS)	-	5 (6.25x5 m)

The selection of plot locations in the two reforestation sites was based on accessibility for equipment, a minimum of 10 m to the forest edge, and avoidance of unplanted windrows. Plot locations in the gravel pit were selected based on having a minimum amount of spontaneous succession, defined as at least 3 woody stems at > 1 m height and similarities in terms of soil moisture. The passive control plots in the gravel pit site were shaped to create as much similarity as possible with the treatment plot in terms of previous vegetation cover and tree cover. Shape and placement also attempted to avoid past recreational vehicles tracks and, as much as could be predicted, potential future routes for recreational vehicles that regularly trespassed on the property. Further details on plot selection methods and criteria for all sites are available in section 2 of Appendix 3A.

The protected forest site was selected to be as similar to the donor forest sites as possible while being outside the planned quarry expansion. Similarities were qualitative and based on visual inspection of aboveground vegetation species community and structure, species make-up, and amount of tree canopy, soil moisture, and litter layer. In the protected forest site, a mini-excavator (TB285, Takeuchi Mfg. Co., Ltd.) was used to excavate topsoil and loose-tip it in a 12.5 x 10 m plot area within 10 m of the point of excavation, to make 5 ‘lift-and-drop control plots’. These lift-and-drop control plots were created to test and quantify the impact of excavation and mixing of the topsoil, without a change of the surrounding ecosystem. 5 (6.25 x 5 m) passive control plots were also set up at a 10-30 m distance for comparisons (Figure 3.1f).

In each of the two donor forest sites, 5 (6.25x5 m) passive control plots were set up randomly at a distance of approximately 10 m from where topsoil had been excavated, with an understanding at the time that these plots may become cut and quarried before the end of this experiment.

### 3.2.2.2. Sub-plot treatment area set-up

After mowing, spraying and topsoil transfer, each topsoil recipient plot was split into 4 sub-plot treatment units of 6.25x5 m (Figure 3.2). Different but overlapping treatments were then added to each of the 4 sub-plot treatment units:

1. Topsoil recipient only (TRo)
2. Topsoil recipient and woody debris addition (WD)
3. Topsoil recipient, woody debris, and dogwood (*C. alternifolia*) shrubs (DS)
4. Topsoil recipient, woody debris, and a shade shelter (SS)

For the mow-and-spray control plots in the 2015 reforestation site, the same sub-plot treatments (woody debris, shrubs, and shade shelter) were applied, but without receiving topsoil. The protected forest ‘lift-and-drop’ topsoil treatment plots were also divided in 4, with woody debris added to three quarters and dogwood shrubs added to one quarter, but no shade shelter was added as these plots were already under full tree canopy.

Woody debris was gathered from the forest using mini-excavators, focusing on pieces showing signs of decay. Most woody debris material was gathered and stockpiled in the forest sites in November 2017 and spread in recipient plots in May 2018 as winter weather conditions limited access to sites. *C. alternifolia* shrubs were planted in fall 2017 and spring 2018 to add understory shade. The species was chosen as it was a common understory species in the donor forests and is relatively tolerant to full sun. Six shrubs from one-gallon pots (approximately 0.5-1 m in height) were planted in a 2-3 m diameter ring to add understory shade.

Square shade shelters with a width of 3.7 m were constructed and installed in spring 2018 out of sturdy wooden poles with a shade-cloth rated to filter 75 % of sunlight, attached at 1.8 m height. The shade cloth had a 0.3 m overlap over the sides to filter diagonal sunlight. The shade cloths were attached and removed at the approximate time of leaf-out and leaf drop, to simulate the predominantly deciduous canopy in surrounding forests.



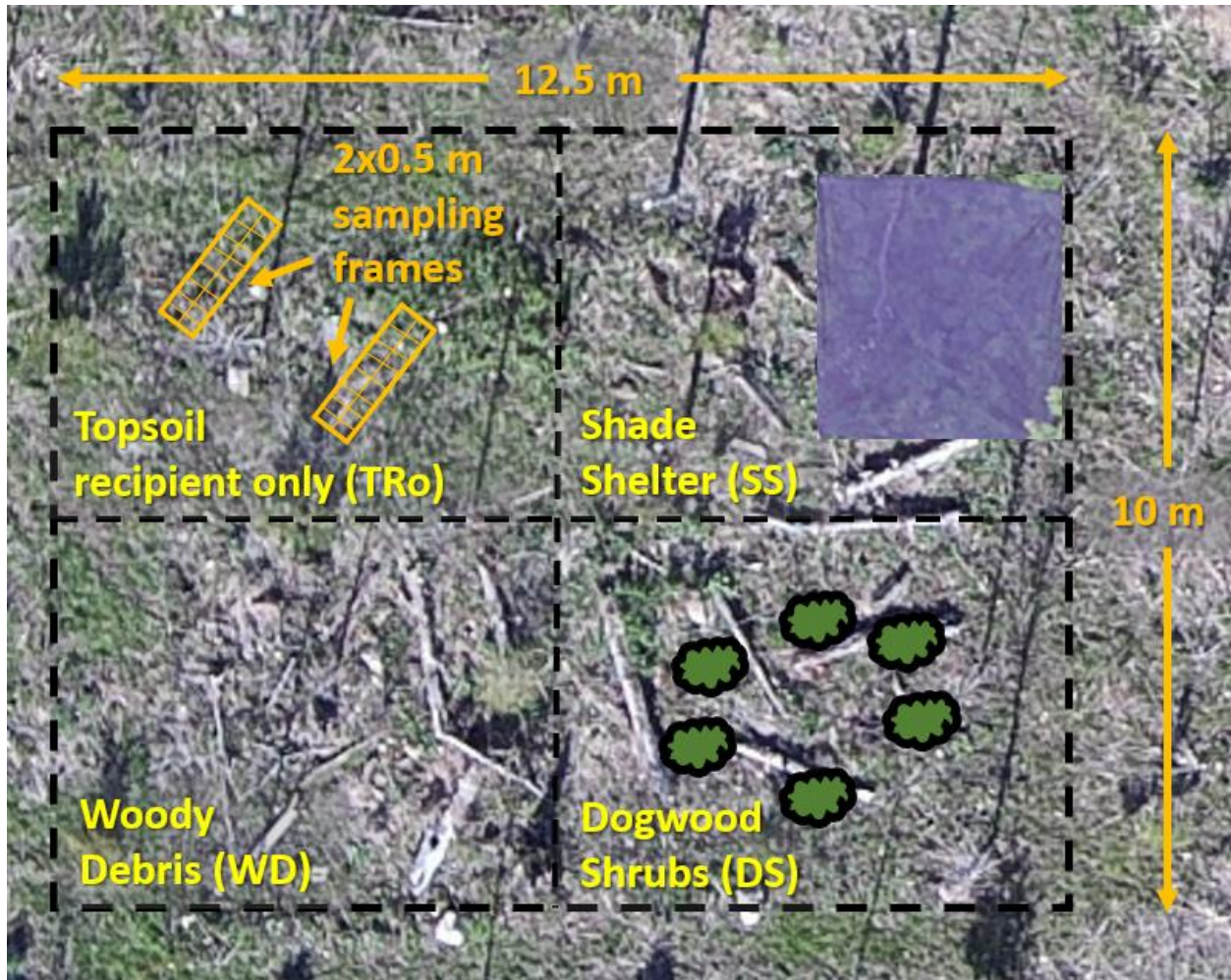


Figure 3.2: Diagram of topsoil recipient plot layout and sub-plot treatments. Image was taken of plot #09 in the 2015 reforestation site, in mid-May 2019. Shade shelter cloth is superimposed on the image from summer imagery as it would only be attached at the time of canopy leaf-out, 1-2 weeks later. Dogwood shrubs are symbolized by the 6 green circles. Inner and outer sampling frames are indicated by the orange rectangles, to scale.

### 3.2.3. Sampling and measurements

#### 3.2.3.1. Sampling of soil characteristics

Measurements of the depth of excavation and the depth of the soil A-horizon (topsoil) were taken in the northern donor forest during the topsoil excavation in November 2017 in order to establish if it was mainly the targeted A-horizon that was excavated. No excavation depth measurements were taken in the southern donor forest, but the author of this study had supervised all excavation at this site and had adjusted the depth of excavation when needed.

Soil moisture was measured on site for all plots in each recipient site. Soil moisture is important as it sets the limit of potential evapotranspiration, and thereby photosynthesis. Soil moisture was measured at a depth of 6.5 cm using a W.E.T. sensor (Delta-T Devices) in September 2019. Three measurements of both soil moisture were taken in each sub-plot treatment of each plot in each site within a span of 2 hours in the afternoon, on a day with at least two rainless days preceding it. The same measurements of the two reforestation sites were taken on the same day, with the gravel pit measured on a separate day.

#### 3.2.3.2. Census of planted trees

Height and diameter of all row-planted trees in all plots in the two reforestation fields were measured on May 12-13 2018 before leaf-out; and then again, two growing seasons later, in November 16-18 2019, following leaf senescence. Height was measured to the nearest cm using a pole and measuring-tape from the base of the tree to the highest live growth point vertically.

Diameter was measured by caliper at ground level for topsoil recipient plots and at 20 cm above the ground for trees in passive and mow-and-spray control plots, in order to equal the new ground level of plots with topsoil added. Only the thickest stem was measured in the case of multiple tree-stems but measurement was taken below stem branching if within 10 cm of the original measurement point. Any visible signs of disease or damage to the trees were noted. Trees were recorded as dead if there were no observed living buds or signs of resprouting.

#### 3.2.3.3. Plant diversity sampling

Vegetation sampling was conducted in 2019, in the second growing season after topsoil transfer. Sampling took place in May, mid-June to mid-July, and mid to late September, to coincide with the flowering of early, mid- and late-flowering species. All vascular plants were identified to species level if possible. Plants belonging to Poaceae and Cyperaceae were identified to section, genus, or in some cases, to family. Species were categorized as 'native' to Ontario following the VASCAN database (Brouillet et al., 2010). Species were considered 'forest species' if their main habitat was described in the literature as being forest (e.g. Voss and Reznicek, 2012), and if they were not considered a weed species as per the province of Ontario weed-database (OMAFRA, 2020).

The first vegetation sampling was done in mid-May (spring), mainly to identify spring-ephemerals. Presence of all vascular plant species in all sub-plot treatment areas of all plots in the reforestation, gravel pit, and protected forest sites was recorded, except for passive control plots in the gravel pit site. Plots in the donor forest sites were not sampled in spring. Due to low overall plant cover in

all plots (estimated to < 10 %) for the early season, all plant species present were very likely found and recorded.

The second vegetation sampling took place mid-June to mid-July (summer) and measured both species richness and the abundance of species occurrence within each sub-plot in all sites. For sampling two 2 x 0.5 m metal-wire sampling frames were positioned at a diagonal 2.7 m from the edge and center of each sub-plot (as illustrated in Figure 3.2). Each of the two metal-wire sampling frames were subdivided into 64 12 cm<sup>2</sup> squares and each plant species was counted as present or absent from each of the 64 squares. The presence count, between 0 and 64, was translated to percent to obtain each species' abundance (methods comparable to: Schaefer and Larson, 1997; Tomlinson et al., 2008).

Abundance measurements from sampling frames were used to capture the differences in species evenness between common and rare plants for the summer sampling when live vegetation dominated the plots. Due to the low overall plant cover in spring (< 10 %) no sampling frames were used for the May sampling, as relative abundance measurements from sampling frames would have provided very little additional information. A follow-up sampling in September was conducted to identify late-flowering species. The same 2 x 0.5 m sampling frames were used in the same locations as the summer and all plants not identified to species in summer, according to the summer sampling-sheets, were located and re-identification was attempted, but no changes to abundance were made. Further details on summer and fall follow-up sampling in section 3 of Appendix 3A.

### **3.2.4. Statistical analysis**

#### **3.2.4.1. Soil characteristics**

The depth of excavation and depth of A-horizon in the northern donor forest were compared using a non-parametric paired Wilcoxon signed rank test due to non-normality of paired differences (Shapiro-Wilks test  $p = 0.017$ ). The differences in soil moisture between plot-types and sub-plot treatments within each site were tested through a three-way ANOVA with interactions on a linear mixed-effects models. Plot-type and sub-plot treatments, and their interactions were considered fixed effects while each plot was considered as a random intercept effect. The formula used was: Soil moisture ~ Plot-type \* Sub-plot treatment + (1 | Plot). Tukey HSD post-hoc tests were performed on all significant linear models throughout this study.

The W.E.T. sensor had a minimum level of detection, causing an inflation of zeros, especially in the relatively dry gravel pit site, which in turn caused the variance of residuals to become unequal (Appendix 3B). To adjust, 0.5 was added to each observation of the soil moisture in the gravel pit site and

the values were then log-transformed. 0.5 was selected as a reasonable value of minimum detection in the context of other measures within the site.

#### 3.2.4.2. Planted trees

Both reforestation sites were grouped together when comparing tree survival and damage to trees between plot-types, as the same methods of tree planting and topsoil transfer were used at both sites. Sub-plot treatments were not taken into account as the study was only concerned with the potential damage from the transfer, tipping, and spreading of the topsoil. The data on trees in the shade shelter sub-plot treatment area was removed before analysis, as many of these were damaged or had stunted growth due to physical resistance from the shade shelter cloth, and because this method was not considered for application in large-scale restoration projects.

The differences in plot-wise ratio of survival were tested through the non-parametric Kruskal-Wallis test as assumptions of normality were not met in the Shapiro-Wilks test (Appendix 3B).

Differences in plot-wise ratio of tree damage were tested through a one-way ANOVA on a simple linear model.

All trees that were dead, missing, or had a lost its main stem were omitted from the analysis of height and diameter change. Our rationale was that the focus of the test of difference in tree growth due to plot-type was on the effect of the added forest topsoil, and the loss of the main stem could cause very large changes that were not attributable to the topsoil itself. Loss of main stem was either recorded directly or indicated by a negative change in height beyond what could be accounted for by seasonal differences ( $> 5$  cm in height or  $> 5$  mm in width) (Pastur et al., 2007). Out of a total of 467 trees, 34 (6.9 %) were dead and 33 (6.8 %) experienced a loss of main stem. The effect of full plot treatments on tree height (cm) and diameter (mm) in both reforestation fields between May 2018 and November 2019 were compared by applying a three-way ANOVA without interactions to a linear mixed-effects model.

The formula for the linear mixed effect model was: height change ~ Plot type + (1|genus and planting\_size) + (1|Site/Plot). This formula considered each combination of genus and planting size separately and handled it as a random effect in terms of intercept. It also used the plots, nested within their respective sites, as a random intercept effect. The same formula was used with diameter change as the fixed effect dependent variable as well. To create a more balanced model, trees from specific genera were only included if at least 20 trees of that genus were alive by November 2019 and the genus was represented in at least 3 topsoil recipient plots and 3 of either control plot. Genera fulfilling these conditions were *Acer*, *Tilia*, *Quercus*, *Betula*, *Pinus* and *Picea*, which were represented by a total of 267 trees throughout all reforestation plots.

### 3.2.4.3. Plant species richness and community composition

This part of the study tested both the differences in native forest plant species richness and the differences of plant species community composition. The first test was needed to make sure that the plant community composition (tested second) was not changed solely by transfer of non-native or non-forest species from the donor forest, which was a possibility as the donor forest did contain large populations of non-native species including *Alliaria petiolata* and *Tussilago farfara*. Tests on spring and the summer sampled plant data were performed separately as sampling strategies differed between them.

#### 3.2.4.3.1. Spring plant sampling

The spring native forest species presence/absence was compared for both plot-type and sub-plot treatment, and their interactions, by fitting a generalized linear-mixed effects model with a Poisson distribution and testing it through an Analysis of Deviance type III Wald chi-square ( $\chi^2$ ) test. A Poisson distribution was used as the species richness data took the form of non-negative integer counts. Poisson generalized linear mixed-effects model assumes equal variance and no significant outliers. Plots were included as random intercept effect as per the formula: Native forest species richness ~ Site and plot-type \* Sub-plot treatment + (1|Plot). Tukey HSD test were performed for site and plot-type only on a model where sub-plot treatments were considered as random effects: Native forest species richness ~ Site and plot-type (1|Sub-plot treatment) + (1|Plot). To test the potential ability of topsoil transfer to decrease or delay field ruderal species germination, the same Wald  $\chi^2$  test on the same model was utilized, but with non-native/non-forest species richness as independent factor. This test was only performed on plots within each of the two reforestation fields.

Spring plant communities composition, based on sub-plot presence absence data was compared through a Non-Metric Multidimensional Scaling (NMDS) using the Jaccard index method (Baselga and Leprieur, 2015; Koch, 1957). Sub-plots with additional treatments (woody debris, shrubs and, shade shelter) were excluded in the NMDS analysis of difference of plant communities between full plot-type treatments and sites. These additional sub-plot treatments were instead compared separately to each other.

The NMDS method is non-metric as it uses ranking and aims to collapse the many dimensions of the plant community differences (as many as there were plant species in this case) into fewer dimensions. It aims to do so with as little ‘stress’ as possible, shifting the positions of each plot as little as possible while trying to visualize the relative difference between plots as well as possible.

The goodness-of-ordination fit, a measure of the ‘stress’ of the NMDS plot was assessed using the non-metric  $R^2$  fit in a stress-plot. Higher stress levels and lower  $R^2$  indicate that the NMDS ordination may be more arbitrary. Two dimensions for NMDS were used if stress-values were  $< 0.2$ , as this is considered a usable fit (Clarke, 1993). The NMDS was performed using the ‘metaMDS’ package in R

(Kenkel and Orlóci, 1986; Kruskal, 1964; Oksanen et al., 2019) with the ‘monoMDS’ engine set to 1000 random starts to find a stable solution. The NMDS was visualized through an NMDS scatter plot with hierarchical clustering of plots.

The significance of the NMDS results, i.e. the difference in plant communities, was tested using a permutational multivariate analysis of variance (PERMANOVA) on a distance matrix (Anderson, 2017). The PERMANOVA tests whether the centroids of the plant community of each plot-type and their spread were significantly dissimilar from other centroids and spread. The PERMANOVA was conducted using the ‘adonis’ function in the ‘vegan’ package in R (Oksanen et al., 2019). 100,000 permutations were used to calculate an approximate p-value from F-value obtained from the non-metric distance matrix (Anderson, 2001).

PERMANOVA assumes exchangeability of the sample units between groups, which was fulfilled by testing sub-treatments separately. It also assumes similar multivariate dispersion between groups; this was tested using an ANOVA on the ‘multivariate homogeneity of groups dispersions’ through the ‘betadisper’ function in the ‘vegan’ package (Oksanen et al., 2019) of R. The violation of similar multivariate dispersion does not invalidate the test, but results need to be considered in connection with the NMDS visualization (Anderson, 2017). Significant differences of groups that have high overlap in location but differ in dispersion should be evaluated with caution (Anderson, 2017).

After significant PERMANOVA results, plot-types at different sites were compared by post-hoc pairwise PERMANOVA tests. To avoid overtesting, to reduce the risk of type-1 errors in unadjusted p-values and type-2 errors in adjusted p-values, and to increase visual clarity, plot-types were grouped together further (see section 3.3.1.2). P-values were adjusted for multiple tests using the Benjamini-Hochberg procedure for false discovery (Benjamini and Hochberg, 1995). P-values from full and pairwise PERMANOVA are approximate as they depend on the random start and permutations.

An NMDS was also performed for sub-plot treatments in topsoil recipient plots within the gravel pit and reforestation sites. No distinction was made between the two reforestation sites for this analysis. The statistical significance of the difference between sub-plot treatments for the combined reforestation sites and the gravel pit site was tested separately through PERMANOVA to fulfill the assumption of exchangeability of units.

#### *3.2.4.3.2. Summer plant sampling*

To analyze the results from the 2019 summer sampling and fall follow-up, observations of row-planted trees in the reforestation sites were omitted, as the focus of the analysis was on the potential of the 2017 topsoil transfer to move plants. The difference in species richness of native forest species between plot-types and sub-plot treatments was compared using the same models and post-hoc tests as in spring. The

Poisson linear mixed-model equation was: Native forest species richness ~ Site and plot-type \* Sub-plot treatment + (1|Plot). The difference in non-native/non-forest species were also tested in the same way, for the same reason, and for the same sites, as it was for the spring sampling.

Plant community composition from the summer sampling were compared between sites, plot treatments and sub-plot treatments through NMDS using the Bray-Curtis method (Beals, 1984; Clarke et al., 2006). NMDS and PERMANOVA tests were performed in the same way as for the spring sampling but with values based on the Bray-Curtis method. The Bray-Curtis method takes into account not just the presence of species, but also their relative abundance. Sub-plots with additional treatments beyond receiving topsoil were again excluded when comparing plot-types and sites. As in the case of the spring sampling, sub-plot treatments were only compared within topsoil recipient plots in the gravel pit site, as well as in the two reforestation sites combined together.

#### 3.2.4.4. General statistical details, and assumption testing

All statistical analysis in this study were performed in R (R foundation) and all code for the analysis as well as all diagnostics plot and tests of assumptions are available as a Rmarkdown document in Appendix 3B organized by the sub-headings within the results section. All tests applied a 95 % confidence limit for significance as an appropriate limit for the number of plot and sub-plot replicates, to balance the risk of type-1 and type-2 error (Lakens et al., 2018).

The ‘lme4’ and ‘lmerTest’ packages in R were used for all linear mixed-effect models (Bates et al., 2015; Kuznetsova et al., 2017). For Gaussian distributed linear-mixed effect models, the Kenward-Roger (K-R) degrees of freedom method was used to determine the p-values of type-III ANOVAs. The K-R method resolves variability issues in the default “Satterwith method” (Chawla et al., 2014; Keselman et al., 1999). Variance explained in mixed models were calculated as the conditional  $R^2$  (c- $R^2$ ) for the full model, and marginal  $R^2$  (m- $R^2$ ) for fixed effects only, using the ‘MuMin’ R package (Barton, 2020; Nakagawa and Schielzeth, 2013). Tukey HSD test was performed using the ‘emmeans’ R package (Lenth et al., 2018; Searle et al., 1980).

Levene’s test for equal variance, Bonferroni outlier tests, and Shapiro-Wilks normality test was used to check assumptions of simple linear models. However, for linear mixed models these tests are considered too conservative; visual inspection of diagnostic plots was used instead (Gelman and Hill, 2006; Winter, 2013; Zuur et al., 2009)

### 3.3. Results

#### 3.3.1. Soil depth and moisture

The mean depth of excavation in the northern donor forest was 19.6 cm (SD = 8.8), and the A-horizon had a mean depth of 15.5 cm (SD = 6.2). These measures were not statistically significantly different from each other (Wilcoxon signed rank test  $p = 0.177$ ).

In the gravel pit site, the difference in soil moisture between sub-plot treatments was significant ( $p < 0.001$ ) in a K-R method three-way ANOVA on the log-transformed mixed linear model. The full model explained 33 % of variance ( $c\text{-}R^2$ ) and the fixed effects explain 14 % ( $m\text{-}R^2$ ). The mean soil moisture of passive control treatments was 2.73 %, and this treatment was significantly ( $p < 0.05$ ) drier than all soil recipient sub-plot treatments in a Tukey HSD post-hoc test. There was no significant ( $p < 0.05$ ) differences between the different sub-plot treatments in the topsoil recipient plots. There were no statistically significant differences in soil moisture between plot-types or sub-plot treatments in either of the two reforestation sites.

#### 3.3.2. Tree survival, damage, and growth results

##### 3.3.2.1. Tree survival and damage

Out of 467 planted trees measured in the two reforestation sites in spring 2018, 34 (7.3 %) were dead or could no be found (due ~30 cm snow-cover during November 2019 sampling), and 75 were damaged (16.1 %). Mean tree survival did not differ significantly between plot treatment types (Kruskal-Wallis test  $H = 3.5$ ,  $p = 0.172$ ), and neither did tree damage (95 % CI,  $p = 0.282$ ). Most signs of damage were from rot or loss of bark where tree-tubes were used or breakage at the point where the trees were tied to supporting bamboo poles. There was no indication of tree damage specifically related to the transfer, tipping or spreading of the topsoil.

##### 3.3.2.2. Change in tree height and diameter by plot-type

The change in height between May 2018 and November 2019 was not significantly different between tree-types (three-way ANOVA  $F = 2.0$ ,  $p = 0.163$ ).

The stem tree diameter change for the same period, was significantly different between plot-types (three-way ANOVA  $F = 3.8$ ,  $p = 0.036$ ). The full linear mixed-model  $c\text{-}R^2$  was 45 %, but the  $m\text{-}R^2$  was only 4 %, indicating a relatively low explanatory power of the fixed effects. The estimated mean marginal diameter increase was 16.8 mm for both topsoil recipient and mow-and-spray control plots, but 13.6 mm (23 % less) for passive control plots (Table 3.3).



In the post-hoc Tukey HSD test, the difference between passive plots and topsoil recipient plots was found to be statistically significant (95 % CI,  $p = 0.049$ ), while the mow-and-spray control plots were not significantly different from any other plot treatments (Table 3.3). The marginally statistical significance and low explanatory power of the fixed effect means we should be cautious about interpreting the difference in diameter growth.

**Table 3.3: Post-hoc Tukey HSD test between plot-types, based on the mixed effects model controlling for tree genera, planting size, field and plot. MSC = mow-and-spray control, PC = passive control plots, TR = topsoil recipient plots.**

Contrast	Difference	SE	df	t-ratio	p-value
MSC - PC	3.22	1.79	25.6	1.803	0.1887
MSC – TR	0.06	1.8	26.5	0.031	0.9995
PC - TR	-3.17	1.25	20	-2.536	0.0495

### 3.3.3. Plant richness and diversity results

#### 3.3.3.1. Spring vegetation sampling

##### 3.3.3.1.1. Comparing native forest species richness between plot and sub-plot treatments

A total of 65 species were identified in the spring sampling, 34 (52 %) of those were categorized as native forest species (Appendix 3C). 1029 out of 1129 observations were done to species level, with the remainder to genus level because the specimens were too immature. All observation of plants that could only be identified to genus-level were still possible to be categorized as being either native forest plants or not. The species occurring in the highest number of sub-plots was non-native *Taraxacum officinale* (84/106 sub-plots). Native forest species that were most abundant in sub-plots which received forest topsoil at the reforestation and gravel pit sites included *Geranium robertanum* (59/60 sub-plots), *Erythronium americanum* (56/60), *Viola sororia* (47/60), *Caulophyllum giganteum* (29/60) and *Trillium grandiflora* (28/60).

The difference of native forest plant species richness between plot-types was statistically significant (Wald type III  $\chi^2$ : 57.2,  $p < 0.001$ ) between full plot-types, but not significantly different ( $\chi^2 = 0.3$ ,  $p = 0.960$ ) between sub-plot treatments or from the interaction effect between plot-type and sub-plot treatment ( $\chi^2 = 2.2$ ,  $p = 0.998$ ). The explanatory power of the fixed effects was high ( $m-R^2 = 90\%$ ).

Plots in the protected forest, as well as topsoil recipient plots in all recipient sites had significantly higher (95 % CI) native forest species richness than control plots in the reforestation sites as per the post-hoc Tukey HSD test (Table 3.4).

The number of non-native/non-forest species were significantly ( $\chi^2 = 7.2$ ,  $p = 0.007$ ,  $m-R^2 = 19\%$ ) lower in topsoil recipient plots than in passive control plots in the 2016 reforestation site by an estimated marginal mean of 1.6 species, indicating that the topsoil transfer may have delayed or decreased non-native and non-forest species sprouting in the spring at this site. No such difference was found between plot-types in the 2015 reforestation plot.

**Table 3.4: Estimated marginal mean of total species richness and native forest species richness for spring sampling 2019 by field and plot-type, sorted in falling order of native forest species richness. All data to the right of the dotted line refer to native forest species richness, including standard error (SE), asymptotic (z-distributed) 95 % confidence intervals, and post-hoc Tukey HSD test of native forest species richness, based on a generalized linear mixed-effects model with Poisson distribution– plot-types with the same letter were found to be not significantly different from each other. P. Forest = Protected Forest, RF = Reforestation.**

Site	Plot-type	# of plots (sub-plots)	Sp. richness	Native forest sp. rich.	SE	asympt. 95 % CL	Post-hoc grouping
P. Forest	Passive control	5 (1)	12.2	9.8	1.2	7.4 - 13	A
P. Forest	Lift-and-drop	5 (3)	10.27	8.6	1.1	7.2 - 10.2	A
Gravel pit	Topsoil recipient	5 (4)	13.7	8.0	1.1	6.8 - 9.3	A
RF 2016	Topsoil recipient	5 (4)	12.75	7.1	1.1	6.1 - 8.4	A, B
RF 2015	Topsoil recipient	5 (4)	10.9	5.2	1.1	4.3 - 6.3	B
RF 2015	Passive control	5 (1)	7.4	1.2	1.5	0.5 - 2.7	C
RF 2016	Passive control	5 (1)	9.4	0.4	2.0	0.1 - 1.6	C
RF 2015	Mow-and-spray	5 (4)	5.19	0.4	1.5	0.2 - 0.8	C

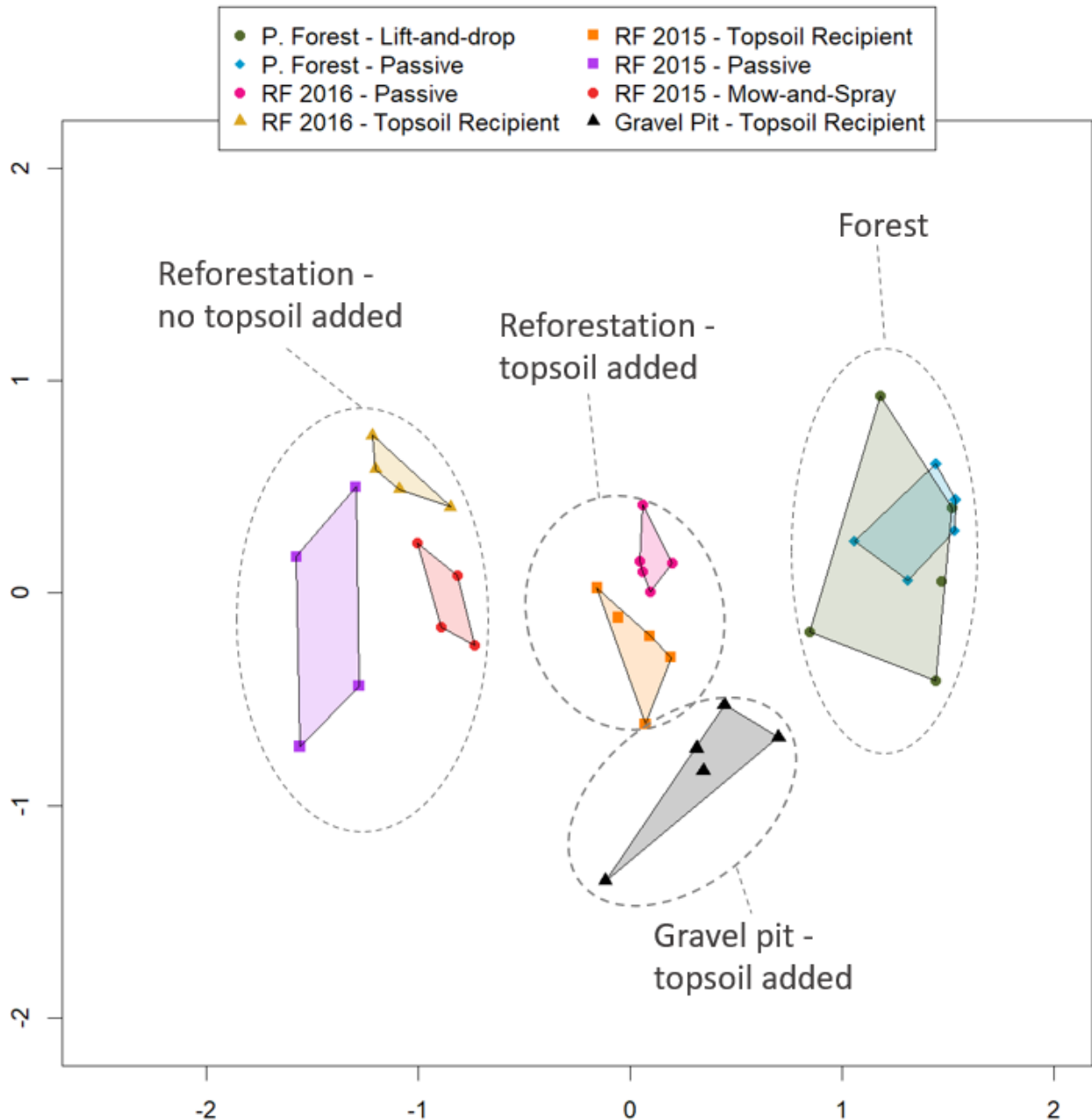
*Fragaria vesca* was the most common native forest species identified in control plots in the reforestation site; making up 11 out of 15 of total native forest species presences noted during the sampling. *Fragaria vesca* is a common species to both fields, edges and forests in this region and its occurrence throughout the post-agricultural reforestation fields, even without forest topsoil added is not out of the ordinary.

### 3.3.3.1.2. Comparing plant community composition between plot-types and sub-plot treatments

The overall visual pattern of the Jaccard index NMDS on full plot treatments (Figure 3.3) was that the plant community composition of topsoil recipient plots in the two reforestation sites were separate from control plots in the same sites. Topsoil recipient plots in reforestation sites were positioned in between the protected forest and the control plots in the reforestation sites as they encompass species from both the donor and recipient communities.

While separate from all other sites, the plant community composition of the two plot-types in the protected forest site were similar to each other, indicating that there were no clear shift in species composition from lifting and dropping the forest topsoil within the same site. The stress-plot goodness of

ordination fit was high, with a non-metric  $R^2 = 0.989$  and a stress value of 0.10, which is generally considered good (e.g. Dexter et al., 2018).



**Figure 3.3: NMS of plant communities in all plot-types and sites in the spring 2019 sampling, using the Jaccard index method for presence/absence data. Axes represent rank-ordered (non-metric) dissimilarity of plant communities. No data from sub-plots with additional treatments beyond topsoil addition were used. Colored hulls outline extent in non-metric ranked space of plot plant community. Gray dashed ellipsis outline larger patterns of reforestation control plots, plots with added topsoil in reforestation sites and in gravel pit, and forest site plots. RF – Reforestation, P. Forest – Protected Forest.**

Differences between sites and plot-types were statistically significant (PERMANOVA:  $F=10.1$ ,  $R^2 = 0.69$ ,  $p < 0.001$ ), with no problematic significant differences in multivariate dispersion between

group (dispersion ANOVA  $p = 0.057$ ). The post-hoc pairwise PERMANOVA was performed on sites and plot-types grouped into one of 4 groups; (1) all control plots in reforestation sites, (2) all topsoil recipient plots in reforestation sites, (3) all topsoil recipient plots in the gravel pit site, and (4) all plots in the protected forest site (Appendix 3D - Table I). The procedure found that the plant community of each grouping of plot-types and fields were significantly dissimilar from others ( $p$ -adjusted  $< 0.001$ ) when the  $p$ -value was adjusted using the Benjamini-Hochberg procedure. In terms of spring plant communities, plots that had received forest topsoil were significantly different from both controls in recipient sites and from the protected forest site, set close to where the topsoil was excavated from. All pairwise PERMANOVAs of the 4 groups fulfilled the assumption of similar multivariate dispersion.

There were no statistically significant differences found between sub-plot treatments within either an NMDS or a PERMANOVA. Results for these are available in section 2.1 of Appendix 3D.

### 3.3.3.2. Summer and fall vegetation sampling results

The summer vegetation sampling, and fall follow-up, included 246 2 x 0.5 frames sampled over 123 sub-plot treatment areas. A total of 162 vegetation species were identified, of which 50 were considered native forest species. A total of 44569 observations of vascular plants were made in the 12.5 x 12.5 cm sub-quadrats in the sampling frames. Of these, 39539 (89 %) were made to species level and 3077 to genus level (6.9 %) with the remaining observations identified to family level.

Poaceae made up 99.9 % of the observations which could only be made to family level. No native forest species in Poaceae were observed in this study. *Carex arctata*, *Carex gracilima* and *Carex debilis* are difficult to distinguish from each other and were grouped by their common section, Hymenochlaenae (Voss and Reznicek, 2012). In the summer, *Viola sp.* could be identified to either a grouping of all native forest *Viola* species, or the non-native and non-forest species *Viola arvensis*.

The most abundant species was *Daucus carota* with 3435 observations (7.7 % of total). *Impatiens pallida* was the second most frequent species overall, and the most frequent native forest species with 3092 observations (6.9 %). *I. pallida* was also the most frequent native forest species when only counting recipient (reforestation and gravel pit) sites. *Circea lutetiana* and *Geranium robertianum* was the second and third most frequent native forest species in recipient sites. The native forest species *Caulophyllum giganteum* was the most frequent species in donor and protected forest plots with 891 out of 4694 observations total in these areas. However, *C. giganteum* had low abundance in recipient sites with 94 observations, pointing to low viability of transfer of this species.

### 3.3.3.2.1. Comparing summer species richness and diversity of plot-type and sites and sub-plot treatments

Similar to the spring-sampling results, there was a statistically significant (Wald type-III test  $\chi^2 = 100.1$ ,  $p < 0.001$ ) difference in summer native forest species richness between plot-types and sites for the summer sampling but not between sub-plot treatments ( $\chi^2 = 0.8$ ,  $p = 0.835$ ) or the interaction between sub-plots ( $\chi^2 = 17.7$ ,  $p = 0.088$ ). The explanatory power of the fixed effects was high ( $m-R^2 = 81\%$ ).

Native forest species richness was significantly higher (95% CI) in topsoil recipient plots in each of the recipient sites, compared to the control plots in their respective sites in pairwise tests (Table 3.5). The topsoil recipient plots in each of the recipient sites did not have significantly different native forest species richness as compared to the donor forest site from which they had received a majority of their topsoil. Considered together, this indicates that native forest species were introduced through the topsoil transfer, and that most of these species survived the transfer after two years.

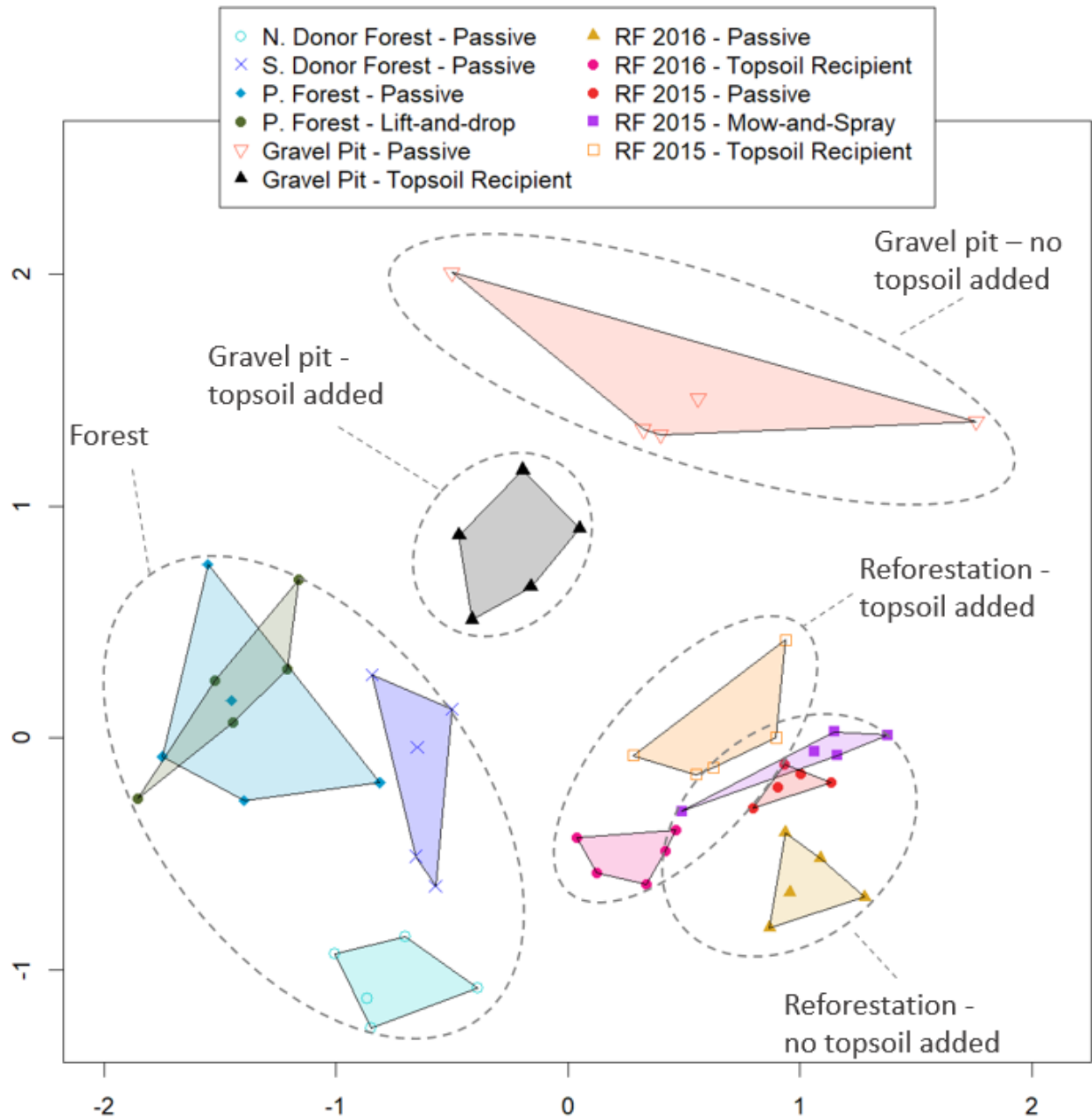
In the gravel pit site, the topsoil recipient plots had a significantly ( $\chi^2 = 15.3$ ,  $p < 0.001$ ,  $m-R^2 = 50\%$ ) higher richness of non-native/non-forest species, with an estimated marginal mean difference of 1.9 additional species. This result indicates that the topsoil transfer had either brought in new non-native/non-forest species, or had improved conditions for spread of those types of species already within the gravel pit site. No such increase of non-native/non-forest species were found for the two reforestation sites.

**Table 3.5: Estimated marginal mean of total species richness and native forest species richness for summer sampling 2019 by field and plot-type, sorted in falling order of native forest species richness. All data to the right of the dotted line refer to native forest species richness, including standard error (SE), asymptotic (z-distributed) 95 % confidence intervals, and post-hoc Tukey HSD of native forest species richness, based on a generalized linear mixed-effects model with Poisson distribution– plot-types with the same letter were found to be not significantly different from each other. P. Forest – Protected Forest; RF – Reforestation; D. Forest – Donor Forest.**

Site	Plot-type	# plots/sub-plots/frames	Species richness	Native forest sp. rich.	SE	asympt. 95 % CI	Post-hoc grouping
S. D. Forest	Passive Control	5/1/2	17.6	13.8	1.1	10.9 - 17.5	A
Gravel pit	Topsoil Recipient	5/4/2	31.8	11.2	1.1	9.8 - 12.8	A, B
P. Forest	Lift-and-drop	5/3/2	11.5	9.6	1.1	8.2 - 11.4	A, B, C
P. Forest	Passive control	5/1/2	11.1	8.8	1.2	6.5 - 11.8	A, B, C, D
N. D. Forest	Passive Control	5/1/2	10.6	8.6	1.2	6.4 - 11.6	A, B, C, D
RF 2016	Topsoil Recipient	5/4/2	25.3	8.1	1.1	6.9 - 9.4	B, C, D
RF 2015	Topsoil Recipient	5/4/2	25.8	6.7	1.1	5.7 - 8	C, D
Gravel pit	Passive Control	5/1/2	14.7	3.8	1.3	2.4 - 5.9	D, E
RF 2015	Passive control	5/1/2	21.3	1.9	1.2	1.4 - 2.6	E
RF 2016	Passive control	5/1/2	18.4	1.8	1.4	0.9 - 3.5	E
RF 2015	Mow-and-spray	5/4/2	18.7	1.8	1.4	0.9 - 3.5	E

### 3.3.3.2.2. Comparing plant community composition of sites, plot-types and sub-plot treatments

Most site and plot-type combinations of plots were visually grouped separately by the scaling function in the NMDS of summer plant species community composition, based on Bray-Curtis dissimilarity of relative species abundance (Figure 3.4). The only exception being the passive and ‘lift-and-drop’ plot-types within the protected forest site, which, similar to the spring NMDS, indicates that the mixing from excavating and dropping the topsoil does not affect the plant species community significantly. There was an overall visual grouping of all forest sites and plots together and all plots in the reforestation sites together. All plots that received topsoil in the reforestation and gravel pit were closer in community similarity to forest sites than control plots in the same sites. The goodness of ordination test fit for the NMDS had a high non-metric  $R^2$  of 0.977 and the stress value was 0.15, which is considered fair.



**Figure 3.4: NMDS of summer 2019 plant species community in all sub-plots with no additional treatments beyond receiving topsoil in all sites . Calculated using the Bray-Curtis method on the abundance of plant species averaged over two sampling frames (2 x 0.5 m each). Axes represent rank-ordered (non-metric) dissimilarity of plant communities. RF – Reforestation, P. Forest – Protected Forest,**

There was a statistically significant differences between sites and plot-types as per a PERMANOVA on the NMDS data ( $F = 6.57$ ,  $p = 0.001$ ,  $R^2 = 0.60$ ). However, due to a significant difference in multivariate dispersion in the data used for the PERMANOVA ( $p = 0.0489$ ), care should be taken to evaluate PERMANOVA results against the NMDS visualization. Differences in dispersion,



rather than difference in location, may account for any significant dissimilarity for overlapping groups. The difference in dispersion between the two plot-types in the protected forest site; and possibly the passive and mow-and-spray controls in the 2015 reforestation area may be affected by dissimilar dispersion effects (Figure 3.4). We do not compare these pairs of plot-types further.

Pairwise PERMANOVA tests were conducted on the Bray-Curtis dissimilarity between sites which had donated or received topsoil from each other, or were controls for the other (Appendix D, Table II). The Benjamini-Hochberg adjusted pairwise tests showed that all topsoil recipient plots in recipient sites were statistically dissimilar (adjusted  $p > 0.05$ ) from both the control plots at their respective sites, and from the donor forest where the majority of the topsoil was taken for that recipient site. Taken together with the native forest species richness and diversity results, this indicates that while success was high in transferring native species, there was also a strong presence of the plant community of the recipient site within topsoil recipient plots.

The post-hoc pairwise comparison between topsoil recipient and passive control plots in the 2016 reforestation site showed significantly dissimilar multivariate dispersion ( $p = 0.023$ ), which is problematic. However, considering the lack of overlap in the NMDS and the highly significant pairwise PERMANOVA ( $p = 0.008$ ) it is likely that the difference of location of centroids is still significantly dissimilar.

There was no visual or significant difference of plant species community composition between sub-plot treatments within topsoil recipient sites when tested through the Bray-Curtis method NMDS and PERMANOVA (Appendix 3D, section 2.2).

### 3.3.4. Summary of results

1. *An increase in native forest species diversity in the recipient sites –*
  - **The spring and summer native forest species richness of topsoil recipient in recipient sites was similar to that of donor forest sites, and dissimilar to that of control plots in the respective recipient sites. This demonstrated that the native forest species came with the forest topsoil and survived at a high rate two years after transfer.**
  - **There were no significant effects on plant species richness due to additional woody debris, *C. alternifolia* plantings, or shade shelters.**
2. *The plant community in the recipient sites becoming more like that of the donor forest –*
  - **The spring and summer plant community composition of topsoil recipient plots was in between control plots in their respective recipient site and the donor forests, and significantly dissimilar from both. This demonstrated that the plant community of**

**the topsoil recipient plots had become more like the donor forests, but also retained part of the recipient site community.**

- **There were no significant effects on plant community composition due to additional woody debris, *C. alternifolia* plantings, or shade shelters.**

3. *An increase, or at least lack of decrease, in the growth of planted trees -*

- **Topsoil recipient plots did not have significantly more damaged or dead trees than controls plots. This suggests that the topsoil can be transferred into a site where trees are already planted without excess damage to trees.**
- **Trees in topsoil recipient plots grew on average 23 % more in diameter than passive control plots over two years. This difference was however only marginally statistically significant.**
- **There was no significant difference in diameter growth between topsoil recipient plots and mow-and-spray control plots. The difference in growth of trees may partly be due to the weed-suppressing effects of mowing and spray.**
- **There was no difference in tree height growth between plot treatments.**

### **3.4. Discussion**

The two first attributes of a restored ecosystem, according to the Society for Ecological Restoration primer on restoration is for it to contain similar species diversity and community composition as a reference ecosystem, and that these species are mainly indigenous (native) to the area (Clewell et al., 2004; Ruiz-Jaen and Aide, 2005). Results showed that native forest plant species were 3 to 5 times higher in topsoil recipient plots, as compared to recipient site controls. Native forest plant species were not significantly lower in topsoil recipient plots as compared to the reference sites – the donor forest control plots. Compared to unassisted recovery of forests and reforestation by tree-planting, where it can take many decades, even up to a century, for a native herbaceous understory to colonize (Harmer et al., 2001; Vellend, 2003), the topsoil transfer caused a rapid increase in these important ecosystem attributes.

That the plant community composition of the topsoil recipient plots became a mixture of that of the donor and recipient site in this study is similar to the results of the majority of forest topsoil transfer projects (e.g. Buckley et al., 2017; Ferreira et al., 2015; Hall et al., 2010; Ryan, 2013). However, those projects differed by removing (or not having) topsoil at the recipient site and either planting trees after topsoil transfer (Helliwell et al., 1996) or not planting trees at all (Ferreira et al., 2015; Hall et al., 2010; Koch, 2007; Ryan, 2013; Zhao et al., 2020). The risk of transferring topsoil after tree-planting is that the machinery required to do so could damage the planted trees and removing the previous topsoil is not possible without damaging the roots. However, there were no significant increases in damaged or dead

trees in topsoil recipient plots. The results from our study was also in agreement with most of the literature on forest topsoil transfer in that the stripping of topsoil with excavators and loose-tipping it at recipient sites cause no significant decrease of plant species emerging from the transferred soil (e.g. Buckley et al., 2017; Craig et al., 2015).

A benefit of transferring topsoil to a site after tree planting is that there would be less time before canopy closure (e.g. Buckley et al., 2017; Zhao et al., 2020). Improving the chances of shade-tolerant understory plants to survive long enough to outcompete ruderal field species (Valladares et al., 2016). For sites with a less pressure from ruderal plants, such as the gravel pit site, early canopy closure is less important for help in outcompeting ruderals, but it is still important for avoiding drought stress and direct sun-damage to shade-tolerant understory species (Valladares et al., 2016)(e.g. Hall et al., 2010). Topsoil recipient plots in the gravel pit site were mostly located near the edge of surrounding forests, which may have aided with further shade. If possible, depositing topsoil in circles from edge to center in over multiple decades could help the forest encroach into these more bare-soil areas.

While waiting to deposit topsoil in a reforestation site until canopy is nearly closed may be preferred, the decision of when to transfer the topsoil will depend on the schedule of the project requiring its removal. Depending on how close to each other trees are planted, and the growth form and spread of different tree species, the reforestation site may not be accessible by the time the quarry is ready to expand and move the topsoil. For example, in this study, the trees in the reforestation sites were planted 2.5 m apart, allowing for access for small machinery (e.g. skid-steers, mini-excavators and tractors). However, there were already re-sprouting *Populus tremuloides* trees in between planting rows, and *Picea glauca* canopies which were spreading out from near ground level. This type of spread would make access, without tree damage, impossible even with small vehicles in these types of sites.

The time-line and speed of quarry expansion depends on market demand and decisions by the industry. Because of this, it may be that the donor forest in this project will be cut at any time between year after this study to a decade later, and quick decisions need to be made about where the forest topsoil should be moved to when that happens. For this reason a two-year study like this one still carries importance as it informs the time-sensitive decision of where the resource – the forest topsoil – should be deposited to mitigate the loss in forest biodiversity.

Each excavation or development project where topsoil transfer is possibly would be different in terms of what kind of topsoil recipient sites are available and accessible nearby (e.g. brown-field areas, former roads, quarries, agricultural fields). Having short-term but site-specific experimental data, even if incomplete, can help improve decision-making about how, and at what site, the topsoil will do the most good in accelerating succession towards a native and diverse forest ecosystem. Short term data of both tree growth and plant species composition, with and without addition of topsoil, can be used in meta

analyses of restoration success (e.g. Crouzeilles et al., 2016), and be applied through plant community modeling in a similar manner to earlier modelling studies of plant traits and competition in forests (e.g. Berger et al., 2008; Warren and Topping, 2004; Wramneby et al., 2008). The data collected can be useful in predicting different scenarios and potential successional trajectories (e.g. Rastetter et al., 2003; Young et al., 2001) based on when canopy closure can be expected. It may also in turn be used to predict the value of the ecosystem services improved upon by the topsoil transfer over time (Turner et al., 2016).

The economic cost and benefits of topsoil transfer should be weighted against each other by the quarry operator. Assuming the same costs and methods used for the experimental topsoil transfer in this study, it would cost 720,000 CAD (2017 value) to cover the full 5.1 ha 2015 reforestation site to 20 cm depth (10,000 m<sup>3</sup> of topsoil or 10,000 – 15,000 loads using the bucket of a tractor or skid-steer). The benefits, on the other hand, is avoiding delays in achieving goals of tree growth and spread of a ‘tolerant hardwood understory’ (Charlton and Hims, 2013), which were required to be met to be permitted to expand the quarry further. Delays to achieving these goals and thereby delays to quarry expansion could be very costly.

Considering that it can take over a century for forest restored through tree planting to achieve an understory similar to that of older intact forests (Harmer et al., 2001; Vellend, 2003). The opportunity cost of not doing the topsoil transfer, and thereby slowing understory creation and tree growth, may outweigh the cost of going through with it, even in terms of just the economics of the project operator. And that is not even to speak of the ecological opportunity cost of slower recovery of the ecosystem services the site could be providing to the local community.

While this study found no effect of the woody debris, shrubs or shade shelter treatments on plant species diversity, they may have had effects on other aspects of the ecosystem, the treatments may have been too small, and/or 2 years may have been too short to notice any effect. Woody debris did not affect plant communities or tree growth in this experiment, but it may still have affected soil microbial and insect communities. While not monitored systematically in this study, there seemed to be a low rate of survival of *C. alternifolia* shrubs. Better planting and protection methods of these shrubs may have yielded more of an effect on understory plant species diversity. Shade shelter construction caused a large amount of damage and were limited in size by tree-plantings. A future experiment, perhaps done jointly with a larger topsoil transfer, would be to hang larger shade cloth between tree-stems under their canopy, this would avoid the damage and cost of installing poles to hang the shade cloth on, simplify installation and allow for larger areas to be covered.

### **3.5. Recommendations**

Based on the results of this study, it is recommended that forest topsoil be loose-tipped directly to old field tree planting areas or pit or quarry sites. For sites with trees already planted, soil should be added using small equipment, such as tractors, taking care to not harm any existing trees or overly compacting the new topsoil once it has been spread. To increase chances of canopy closure aiding shade-tolerant forest understory species, and to improve tree growth, the priority should be to add topsoil to tree-planting areas. However the decision of using the soil on pit and quarry sites or in reforestation areas would depend on accessibility, pressure from ruderal species, and projections of where canopy can be expected to close earliest.

# **Chapter 4. UAV thermal imaging detects lower temperatures after topsoil transfer restoration increased native forest plant species richness**

## **4.1. Introduction**

Vegetated terrestrial ecosystems can decrease local daytime surface temperatures during the growing season (e.g. Aerts et al., 2004; Lin et al., 2020; Luvall et al., 1990). Short-wave radiation energy from the sun is absorbed, transformed, and stored by plants through transpiration, respiration, and creation of new biomass (Gosz et al., 1978; Raven et al., 2003 chap. 5). Transformation and storage of part of the incoming energy reduces the portion of energy that is re-emitted directly as long-wave thermal radiation, i.e. sensible heat (Fraser and Kay, 2004; Luvall et al., 1990; Schneider and Kay, 1994). Sensible heat is measured as surface temperature, therefore, the more transformation and storage of energy by an ecosystem, the lower the surface temperature.

If there is enough soil-water available, and photosynthetic and respiration capacity is not yet at maximum, the relative capacity of ecosystems to transform and store energy increases with increasing solar energy radiation (Bonan, 2008 chap. 17; Raven et al., 2003 chap. 5; Wilhelm and Selmar, 2011). This increased transformation and storage means vegetated ecosystems tend to have more of a cooling effect at daily and seasonal peaks of radiation compared to non-peak times (e.g. Aerts et al., 2004; Frey et al., 2016; Lin et al., 2020; Wagendorp et al., 2003). This mitigating of peak temperatures, and storage of heat until later in the day, is referred to as the ecosystem thermal buffering capacity (Aerts et al., 2004; Lin et al., 2020; Wagendorp et al., 2003). The cooling and buffering of summer daytime temperatures has been shown to be higher in forest ecosystems than croplands, grasslands, and savannah in similar climate conditions (e.g. Lin et al., 2020; Luvall et al., 1990). Buffering and cooling has also been shown to increase as forest ecosystems mature (e.g. Aerts et al., 2004; Lin et al., 2017b) and decrease with forest degradation (e.g. Lin et al., 2017a; Luvall and Holbo, 1989).

The ability to buffer daytime surface temperatures during the growing season increases with photosynthetically active biomass (PAS) (e.g. Lin et al., 2020, 2017b). Increased photosynthetically active biomass (PAS) increases total physical heat capacity (Jayalakshmy and Philip, 2010) and gross primary productivity (GPP) (Williams et al., 1997). Increased GPP in turn increases potential transpiration and respiration (e.g. Baldocchi, 2005; Lin et al., 2018). Increased transpiration, respiration and heat capacity thereby transform and store more of the incoming energy, releasing less as sensible heat.

Thermal buffering has also been shown to increase with increasing species diversity (Hamberg et al., 2020). Plant diversity can affect temperature indirectly through an increase in the variation and effectiveness of different water, energy, and nutrient cycling related functions and structures (Baldocchi, 2005; Hooper et al., 2005; Loreau, 1995). These functions and structures include, among others, variations in root depth (Guderle et al., 2018; O’Keefe et al., 2019), leaf shape, timing of leaf-out, and phototropism; all of which can aid in increasing and stabilizing respiration, gross primary productivity (GPP) and transpiration (Baldocchi, 2005; Hooper et al., 2005; Loreau, 1995; Schneider and Kay, 1994).

We found that surface temperature decreased faster year-to-year in sites with higher plant diversity, controlling for understory cover, tree cover, and number of tree stems (Hamberg et al., 2020). However, satellite thermal imagery has a relatively low spatial resolution. Thermal image resolution varies between 60 to 120 m<sup>2</sup> for Landsat and ECOSTRESS instruments and is 1 km<sup>2</sup> for instruments on MODIS and Sentinel satellites (Barnes et al., 1998; Guzinski and Nieto, 2019; Hook et al., 2019; Zanter, 2018). These coarse resolution satellite images are not suitable for comparing vegetation differences in small plot-level experiments, or in sites smaller, or narrower, than the satellite’s pixel size. This matters as many restoration sites are at or smaller than these pixel-sizes. For example, out of the 100 pits and quarries that are planned to be restored, or are already being restored, closest to our research site, 17 sites are  $\leq 1$  ha (100 m<sup>2</sup>) (TOARC, 2020). If this size pattern is true for all of Ontario’s 6527 surrendered pits and quarries, then there would be 1110 potential project sites that are  $<1$  ha, which could not be imaged by satellite.

Satellite thermal images also have limited temporal resolution – the coarse (1 km<sup>2</sup>) resolution data from MODIS and Sentinel satellites can be acquired approximately once per day, while the moderate resolution data from Landsat can be acquired every 16 days (Barnes et al., 1998; Guzinski and Nieto, 2019; Zanter, 2018). The satellite data acquired is also only useable if acquired under cloud-free conditions.

In contrast to thermal imagers on satellites, thermal imagers on UAVs can have spatial resolutions down to single-digit centimeters, depending on the instrument and the distance between the UAV and surface (Kelly et al., 2019; Santesteban et al., 2017). Also, in contrast to satellites, UAVs can be used on-demand to image a site multiple times per day, over multiple set times each day, allowing for comparison of thermal buffering at different times of the day. Beyond just temperature measurements, this potentially consistent and high temporal resolution of UAV’s can also be important for studying many other diurnal ecosystem processes, including transpiration and CO<sub>2</sub> exchange.

Research on vegetated areas using remote sensing thermal cameras attached to UAV has mostly focused on identifying water-stressed areas in managed production landscapes, especially in agriculture (Berni et al., 2009b; Maes and Steppe, 2019), but also in orchards and monoculture tree-plantations (e.g.

Iizuka et al., 2018; Ludovisi et al., 2017; Smigaj et al., 2019; Zarco-Tejada et al., 2012). Using UAV-mounted thermal cameras, significant temperature differences have been detected between diseased and non-diseased pine (*Pinus* sp.) in a semi-naturalized plantation setting (Smigaj et al., 2019). While tree plantations have some of the structure and function of more diverse forests, plantations are generally simpler, often with only one even-aged tree species (Aerts and Honnay, 2011; Brockerhoff et al., 2008; Chazdon et al., 2016). In comparison to plantations, forests have a larger variation in species, height, age, structure, and overall complexity (Brockerhoff et al., 2008; Chazdon et al., 2016). There is so far a lack of published studies on UAV thermal imaging of diverse, non-commercial ecosystems. There are also no studies comparing thermal images from UAV with in situ measured plant species diversity, biomass, cover, or other ecosystem attributes, which would be needed in order to operationalize UAV thermal imaging for detecting ecosystem change.

Measuring ecosystem change, and change in plant diversity through thermal imaging, could be useful for conservation and restoration evaluation and monitoring. While there has been interest in using UAVs in conservation and restoration initiatives, there has so far been very little knowledge of, or interest displayed in, thermal imagers specifically (e.g. Buters et al., 2019; Cordell et al., 2017; Nagendra et al., 2013; Reif and Theel, 2017). Thermal imaging of restoration projects can potentially be relevant to restoration in multiple interconnected ways. It can be used both to quantify the amount of thermal buffering ability added by restoration, and it can be used as a holistic metric of ecosystem change to monitor and evaluate the trajectory of the restoration, and thereby the treatments and methods used.

Thermal image data was collected using a UAV, and ecological data was collected in-situ, to determine if restoration of post-agricultural reforestation sites, and gravel pit sites could be accelerated through the transfer of topsoil. Using a series of control and topsoil transfer restoration plots, we sought to determine (1) if plots with forest topsoil, added two growing seasons prior, cool and buffer surface temperatures more than non-topsoil recipient control plots; and if so by how many degrees at different times of the day and season? And, (2) are there significant correlations between summer daytime surface temperature and ground-measured ecological attributes, including plant cover, plant species richness and diversity of all plants, native forest plant species richness, and tree size; and if so, what are the patterns of correlations within and between sites?

## **4.2. Methods**

### **4.2.1. Study site**

This study was conducted at two reforestation sites on former agricultural land, and one inactive gravel pit. All sites are located in Clearview Township, southern Ontario, Canada.



The two reforestation sites are both located on the Niagara Escarpment, 510-530 m asl, and between 30 to 500 m from a planned quarry expansion (Figure 3.1). The reforestation sites were part of a larger effort aimed at mitigating the negative effects of a quarry expansion (Charlton and Hims, 2013). The planned quarry expansion encompasses the forest from which the topsoil was excavated and donated for all topsoil recipient plots in the experiment. The two reforestation sites are referred to in this study by the year they were planted with trees – 2015 and 2016. Both reforestation sites were planted in spring with a mix of native trees in rows (Table 3.1). After every 5<sup>th</sup> row of planted trees, a windrow was created by heaping up woody debris, rocks, and soil (Charlton and Hims, 2013). No trees were planted in the windrow. The 2015 site was 5.1 ha in size, with a mean slope of 6.4° at a mean aspect of 142° and a maximum difference in elevation of 17.2 m. The 2016 reforestation site was 1.0 ha, with a mean vertical slope of 5.7° at a mean aspect of 185° and a maximum difference in elevation of 4.5 m. Methods for acquiring the digital elevation model used to acquire slope, aspect and elevation differences are available in Appendix 4A.

The gravel pit site is located 6.5 km north-east from the reforestation sites, below the escarpment at 235 m asl. The full gravel pit is 14.3 ha in size, but the section of the gravel pit site used for the experiment is 0.5 ha in size. No elevation model was created for the gravel pit site, but topographic data shows a maximum of 5 m elevation difference within the site (OMNRF, 2020), however, due to the gravel extraction the section of the site used for plots appeared nearly flat. While gravel pits resemble quarries, the pits are generally shallower, with only excavation of surface gravel and without the blasting done to break rocks in quarries. Prior to the experiment, in summer 2017, the site mostly had exposed sand and gravel, with 16 % overall vegetation cover.

More details on the study sites and the excavation of the forest topsoil are described in section 2.1 of Chapter 3.

#### **4.2.2. Topsoil transfer and plot setup**

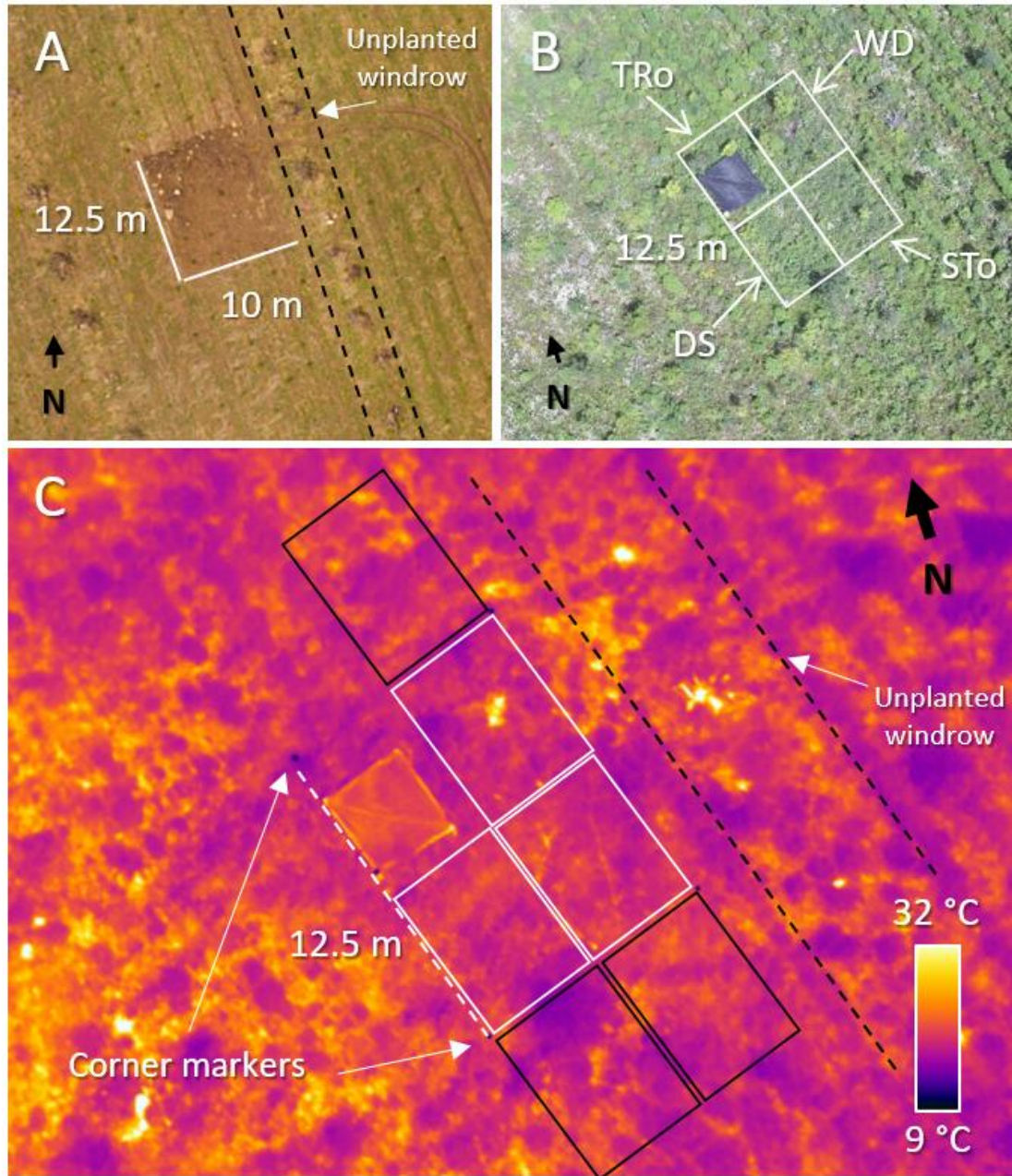
Topsoil was moved from two deciduous temperate forest donor sites to plots within a post-agricultural reforestation site as well as to a post-extraction gravel pit. Topsoil transfer moves not just the soil itself but also the propagules, mycorrhizae, nutrients, and other valuable ecosystem components in and on top of the soil (Fowler et al., 2015; Raven et al., 2003 chap. 30). The intent of restoration by topsoil transfer is to improve the restoration trajectory, by increasing the diversity and biomass of native forest plants, including trees (e.g. Douterlungne et al., 2018; Koch, 2007). Depending on the condition of the recipient site, an added reason for topsoil transfer is to suppress ruderal field species of plants, even to the point of causing a net-loss in total species diversity (e.g. Buckley et al., 2017; Fowler et al., 2015).

The inclusion of reforestation sites with an already high richness of ruderal and non-native field species allows for a comparison of the effect on the surface temperature of all plant species as compared to the effect of only native forest species, most of which were introduced with the forest topsoil. In each of the three sites studied (gravel pit site, 2016 reforestation site, and 2015 reforestation site), 5 plots received forest topsoil in October and November 2017. The topsoil recipient plots were 12.5 x 10 m in size (Figure 4.1A), with topsoil added to a depth of approximately 20 cm. For the two reforestation sites, the topsoil recipient plots were mowed and then sprayed with glyphosate herbicide one to two weeks before receiving topsoil, to limit the re-emergence of ruderal field species.

In the 2015 reforestation site, 5 plots of the same size were also mowed and sprayed but received no topsoil. These mow-and-spray control plots were used as control for the effect of the mowing and spraying done before the topsoil transfer. No mow-and-spray control plots were established in the 2016 reforestation field. Each topsoil recipient treatment plot and mow-and-spray control plot was divided into 4 sub-plot treatment areas: One sub-plot area had no further treatment (TRo), one was provided additional woody debris (WD), one had woody debris and *C. alternifolia* (alternate-leaved dogwood) shrubs added (DS), and one had woody debris and a 3.7m<sup>2</sup> shade shelter added (SS) (Figure 4.1B). The sub-plots containing the shade shelter were not included in this study as the shade-cloth material affected the surface temperature registered by the UAV.

Five passive control plots, where no experimental treatment was applied, were established at all three sites. The passive control plots were 12.5 x 10 m in the reforestation sites and 6.25 x 5 m at the gravel pit site. In the gravel pit site, each of the passive control plots was paired with, and established adjacent to, each of the topsoil-recipient only (TRo) sub-plot of the topsoil recipient plots (Figure 3.1g)

The set-up of the experimental plots is described in more detail in section 2.2 of Chapter 3.



**Figure 4.1: Example of full plot set-up and thermal imagery.** All images are of the same topsoil recipient plot within the 2015 reforestation site. A – Mosaic of optical images taken from UAV in May 2018, 6 months after soil transfer, but before the addition of woody debris and shade shelter. Dashed black lines indicate approximate edges of the nearest unplanted windrow (also in Fig. 1C). B - Single optical image taken from UAV at 12 pm, July 9<sup>th</sup>, 2019 with sub-plot treatments noted: TRo – Topsoil recipient only, WD – Soil transfer and woody debris, DS – Soil, woody debris, and dogwood shrubs, SS – Soil, woody debris, and shade shelter. C - Single false-color thermal image taken of the same plot and simultaneously with image B. Rectangular sub-plot areas of interest are marked by white rectangles. For the 2015 Reforestation area only, temperatures were normalized by subtracting the mean temperature of the sub-plot area (white rectangle) by that of an area adjacent to it, outside the plot, and parallel to the tree rows, avoiding unplanted windrows (black rectangle) (see section 2.5.1). Reflective corner markers noted were used to identify the plot corners and guide manual extraction of mean temperature in FLIR Thermal Studio software (FLIR Systems Inc.). Length of side of plot as dashed white line.

#### **4.2.3. Selection and field methods of measuring and sampling ecological attributes**

The ecological attributes (Schick et al., 2019) used in this study were selected based on whether they were hypothesized to directly or indirectly affect surface temperatures. The variables included were % plant cover, species richness of all vascular plants, native forest species richness, species inverse Simpson diversity of all vascular plants, tree diameter, tree height, and tree genera richness.

Plant sampling was conducted in June and July 2019, with a follow up for further species identification in September 2019. Two 2 x 0.5 m sampling frames, subdivided into 64 12.5 cm<sup>2</sup> squares, were used to quantify the abundance of vascular plant species occurrence in each sub-plot. The same sampling frames were used to estimate % of living plant cover inside the frame. The areas not covered by living plants were observed to be covered with bare soil, woody debris, rocks, or dead biomass. Plants were included in measurements of cover and diversity if they were present below chest-height of sampling personnel (110-140 cm). Row-planted trees in the two reforestation sites were excluded from this sampling. Measurements from the two sampling frames were averaged by sub-plot area for statistical analyses to avoid pseudoreplication (Hurlbert, 1984).

All sub-plots within topsoil recipient plots and mow-and-spray control plots were sampled for plant species and ground cover. For passive control plots in the two reforestation sites, only the north-east quarter sub-plot was sampled. For the gravel pit site, the passive control plot only consisted of one sub-plot, which was sampled.

The plant species diversity was calculated using the inverse Simpson diversity index, which weighs the plant species richness (i.e. number of species) against the overall evenness of plant species distribution (Chao et al., 2014). Native forest species richness was a subset of the total plant species richness that only includes vascular plant species considered native to Ontario and generally found in a forest habitat (Chapter 3, section 2.3.3.).

Tree height and diameter at ground-level of planted trees in the two reforestation sites were measured in November 2019. The same planted trees were identified to genera. Tree diameter, tree height and tree genera richness were measured as the cumulative total of all living planted trees within the sub-plot. Tree height and diameter were included in this study as an increase in height and diameter should, on average, correlate with higher total leaf area, more root-mass and higher potential xylem flow (e.g. Vertessy et al., 1995; Wullschleger et al., 2001), in turn correlating with higher potential transpiration and lower temperatures.

The tree genera richness of planted trees was also included as a separate variable for similar reasons as the measures of plant species richness. Tree genera richness was not manipulated through the study; however, tree planting was done semi-randomly and depended on the availability of stock of

seedlings and saplings of different species. Because of potential patterns in tree species distribution, and because of potentially higher tree mortality of certain genera, it was still possible that there may have been enough variation in tree genera richness to influence surface temperature.

The vegetation and tree sampling methods are described in further detail in Chapter 3, sections 2.3.2 and 2.3.3. Data on the number of vegetation samples taken and trees measured and their growth and survival are available in section 3.2 and 3.3 of Chapter 3.

#### **4.2.4. Thermal imagery acquisition and processing**

Thermal imagery was acquired using a Zenmuse XT2 (SZ DJI Technology Co., Ltd.) gimbal and camera-housing system, which contains a FLIR Tau 2 radiometrically calibrated uncooled microbolometer (FLIR Systems, Inc.) and an optical camera in the visual spectrum, mounted to a DJI Matrice 200 (SZ DJI Technology Co., Ltd.) quadcopter. The Tau 2 thermal imager had a 13 mm lens, a 640 x 512 resolution, and a field-of-view of 45 x 37°. The UAV was flown at a height of 60 meters above its take-off location, providing a pixel size of approximately 7-9 cm<sup>2</sup>. Due to the sloping terrain of the 2015 reforestation site, the height of the UAV above ground level varied by  $\pm 5$  m between plots at this site.

GPS points of each plot were taken to allow for automatic waypoint flying and imaging. To avoid trampling inside the plots, GPS points were acquired at the north-east corner of each plot. GPS data was recorded using a Garmin Montana 650 GPS unit (Garmin Ltd.) with an accuracy of 1.98-4.3 m in open conditions (USDA Forestry Service, 2020). Waypoints were input and flown using the DJI Pilot App (SZ DJI Technology Co., Ltd.), on a DJI CrystalSky tablet. Wooden corner-stakes with 12.5 cm<sup>2</sup> plywood squares, with reflective low-emissivity tape on top, were added to each plot corner to locate plot-corners in thermal imagery (Figure 4.1C).

To calibrate the full field of the microbolometer, the instrument was turned on 5 minutes before beginning the flight take-off, and flat-field correction was performed at the beginning of each flight (Kelly et al., 2019). Some of the images taken still showed pronounced patterns of lowered temperatures towards the edges of each image, a common issue with these types of uncooled microbolometers, even when radiometrically calibrated, due to the temperature of the camera-housing (Kelly et al., 2019). The area of each image was about 40 x 50 m in size, with the 10 x 12.5 m plot near the center, and therefore the edge cooling error effect was limited within the area of interest. With all plots near the middle of the image, and relative temperature comparisons made within the same site and flight, the relative effect of the edge cooling error effect should be further reduced.

UAV thermal imagery was acquired over 5 days in July, and 4 days in September 2019 (Table 4.1). July and September were selected as months representative of the middle and late growing season, respectively. Each day imaged either the two reforestation sites together, due to their close proximity to

each other, or the gravel pit site. In July all plots were imaged twice per hour at 12, 2, 4, and 8 pm, except for on July 7<sup>th</sup>, due battery charging issues, and on July 14<sup>th</sup> at 12 pm due to mixed cloud conditions. In September all plots were imaged twice per hour at 12, 2, and 4 pm. Flights began within half an hour of the beginning of the flight hour, ending at the latest 35 minutes after the beginning of the flight hour. All flight times and temperature data are available in Appendix 4B. The duration of one flight, from first to the last image, was 191 seconds (15 images) for the 2015 reforestation site, 82 seconds (10 images) for the 2016 reforestation site, and 57 seconds for the gravel pit site (5 images).

**Table 4.1: Number of thermal imaging flights flown by site, month, date, and hour of the day, local time (EDT) in 24-hour format.**

Site	Hour	July - dates					September - dates			
		7	8	9	14	15	7	8	9	17
Gravel Pit	12					2				2
	14				2	2				2
	16				2	2				2
	20				2	2				
RF-2016	12		2	2			2	2	2	
	14	1	2	2			2	2	2	
	16	1	2	2			2	2	2	
	20	1	2	2						
RF-2015	12	2	2	2			2	2	2	
	14	1	2	2			2	2	2	
	16	1	2	2			2	2	2	
	20	2	2	2						

In total 98 flights were conducted (Table 4.1), acquiring a total of 1080 thermal images of plots. Each plot in the gravel pit site was imaged 20 times each, while each plot in the 2016 and 2015 reforestation site was imaged 38 and 40 times respectively. The difference in the number of days a site was imaged was due to appropriate weather conditions and site access. The two reforestation sites were prioritized over the gravel pit site. The most comparable studies are those using UAV thermal imaging of tree plantations and low-diversity tree stands. These studies include Izuka et al. (2018), which flew a total of 4 times over 2 days, Smigaj et al. (2019), which flew 6 times during 1 day, and Ludovisi et al. (2017) which flew 1 time.

Flights were flown on days with uniformly clear, hazy, or overcast skies to keep incoming solar radiation equal throughout each flight. Skies were nearly clear or had light haze for 7 out of 9 flight-days (excepting July 14<sup>th</sup> at 12 pm, which had partially cloudy conditions). The two other days - September 7<sup>th</sup> and 8<sup>th</sup> - were uniformly overcast.

The relative humidity had to be below 75 % and wind-speeds below 5 m/s for imaging to take place. Low wind-speeds minimize unpredictable cooling and movement of heat from wind, and the humidity threshold allows for active cooling by transpiration (Bonan, 2008 chap. 14 and 17; Grace, 1988). Air temperature, relative humidity, and wind-speed were recorded with a Kestrel Weather Meter (ITM Instruments Inc.) at each flight-hour. For the days with acceptable sky-conditions, no flight had to be canceled due to humidity or wind-speeds above acceptable levels. Relative humidity had a mean of 42 % (SD = 8.4) for afternoon flights (12 – 4 pm), with humidity rising at 8 pm to a mean of 64.1 % (SD = 8.1). Wind speeds were overall low, with a mean of 0.6 m/s (SD = 0.7) and a maximum of 2.6 m/s. The

mean air temperature for July flights was 28.3 °C (SD = 2.4) during the day and 20.2 °C (SD = 2.4) in the evening; and for September flight-days it was 24.9 °C (SD = 4.8) in the day and 14.8 °C (SD = 4.5) in the evening.

Differences in atmospheric conditions (humidity and atmospheric transmissivity) can affect the accuracy of absolute temperatures derived from thermal data acquired from UAV (Berni et al., 2009a; Hammerle et al., 2017; Kelly et al., 2019). However, with the low flight altitude (60 m) and with flights conducted at times of relatively low humidity, errors due to atmospheric conditions should be minimal (see Fig. 7a in Berni et al. (2009a)). FLIR Thermal Studio software (FLIR Systems, Inc.) was used to process all images to an emissivity of 0.95 for all images. Emissivity is the ratio of emittance of radiation as compared to a perfectly emitting blackbody. The emissivity of bare soil and vegetation can vary between approximately 0.90 and 0.99, depending on the type of soil and vegetation, and the part of the thermal spectrum used for imaging. An emissivity of 0.95 was selected as it is a measure between bare soil and vegetation for the spectral band of the camera (7.5 - 13.5  $\mu\text{m}$ ) (for several different emissivity estimations, see: Meerdink et al., 2019; Rubio et al., 1997; Sobrino et al., 2008). Changes in emissivity and albedo were not measured separately as they were considered a secondary effect of plant cover, tree size, and species diversity. Mean, maximum and minimum temperatures for each image were extracted manually using the polygon measurement tool in the FLIR Thermal Studio software (FLIR Systems, Inc.). While the camera was always directly downward vertically, the horizontal angle of the camera had slight variations. Due to these variations, the imagery could not easily be batch-processed, and manual extraction of data was found to be faster than geo-referencing each image.

#### **4.2.5. Statistical analyses**

##### **4.2.5.1. Ecological attributes and thermal buffering over time in topsoil and control plots**

To characterize the ecological attributes of topsoil transfer and control sub-plots at each site, their means are presented along with thermal data for each site. Separate one-way ANOVAs were performed comparing each ecological attribute (section 2.3.) in each site by plot-type. More in-depth comparisons of ecological differences between plot-types are available in Chapter 3.

To test the effect of topsoil transfer on surface temperature at different times of the day and in different seasons, sub-plots that received topsoil were compared to controls within the same site and at the same flight-hour and month. Sub-plots with additional treatments (woody debris and dogwood shrubs) were not used in these analyses to avoid the confounding effects of these treatments. For the two reforestation sites, only the north-east quarter of passive control plots were used, in order to balance the statistical models in terms of having the same number and size of samples per plot-type. The north-east



quarter was selected as this was the only quarter of passive control plots that were sampled for ecological attributes. For the gravel pit site, the full plot was used as it was already quarter-sized (5 x 6.25 m) and sampled as a sub-plot (section 2.3.).

To compare surface temperature over time for topsoil recipient and control sub-plots, at all sites, the primary response variable used was the mean surface temperature of the sub-plot.

For the 2015 reforestation site, a normalized mean ‘net-temperature’ was tested as well. The net-temperature normalization was calculated by subtracting the mean temperature of the sub-plot by that of a same-sized area adjacent (0-30 cm distance) to the sub-plot. The adjacent area was always in the direction of the planted tree rows, to avoid un-planted windrows (Figure 4.1C). The normalized net-temperature was used to control for the possible effect of higher variation in terrain in the 2015 reforestation site (up to 17.2 m difference in elevation), as compared to the other two sites (up to 5 m difference in elevation). The more varied terrain in the 2015 site may have affected the plots beyond the effect of the experimental treatment, through differences in soil-moisture availability, incoming sunlight, and differing wind-conditions (Bennie et al., 2008; Cantlon, 1953; Iizuka et al., 2018). This was especially relevant as 4 out of 5 of the mow-and-spray control plots were all located within one corner of the 2015 site (Figure 3.1e). In contrast, the other sites were smaller in size, with less difference in elevation. Net-temperature normalization would not be appropriate for the gravel pit site as the passive control plots were already established next to the topsoil transfer sub-plot. It would also not be helpful for the 2016 reforestation site as passive control plots were already spaced out closely (15-40 m) in between each topsoil transfer plot.

Measurements of the same sub-plot, taken on the same flight-hour, within the same day, were averaged before analysis to avoid temporal pseudoreplication issues (Hurlbert, 1984). The estimated marginal mean surface temperature of sub-plots within a site was then compared by plot-type within the same combination of flight hour (12, 2, 4, and 8 pm) and month (July and September). For the gravel pit site during all of September, and July at 12 pm, temperature data was only acquired on one day (Table 4.1). For these single-day data, a one-way ANOVA was performed on a linear model, with plot-type as predictor and surface temperature as the response variable. For all other sites and flight hours, measurements were collected on more than one day. With multiple day measurements, two-way ANOVAs with interactions were performed on a linear mixed effect model. The linear mixed effect model included the date of the flight and its interaction with plot-type, to control for the influence of different weather at different flight-days (Zuur et al., 2009). The linear mixed effect models also included the plot number, or ID, as a random intercept effect to control for temporal autocorrelation between images of the same plots on different days (Zuur et al., 2009). Since the 2015 reforestation site had three plot-types (topsoil recipient, passive control, and mow-and-spray control) a pairwise Tukey HSD post-hoc

test was performed if significant differences were observed in the ANOVA. No post-hoc test was needed for the two other sites as they only contained two plot-types.

All statistical analyses in this study were performed in the R programming language code (Core Team, 2013). Estimated marginal means of surface temperature and its standard error were calculated using the ‘emmeans’ package (Lenth et al., 2018). The linear mixed-effects models were fitted using the ‘lme4’ package (Bates et al., 2015) and the ANOVAs on linear mixed-effects models were performed using the ‘lmerTest’ package (Kuznetsova et al., 2017). The ANOVAs on the mixed-effects models used the Kenward-Roger degrees of freedom method (Chawla et al., 2014; Keselman et al., 1999). For mixed-effects models where p-values were found to be statistically significant, the variance explained was calculated as the marginal  $R^2$  ( $mR^2$ ) for fixed effects (plot-type, date, and their interaction), using the ‘MuMin’ R package (Barton, 2020; Nakagawa and Schielzeth, 2013). The code and output (models, visualization, diagnostic graphs, and assumption tests) of all statistical analyses used in this study are available in Appendix 4C, organized by the results sub-sections.

For all statistical analyses in this study, a two-sided 95 % confidence interval was used ( $\alpha = 0.05$ ) to determine if results were statistically significant. This confidence interval and p-value  $\alpha$ -level was selected based on the number of plots and measurements available in the experiment, and to balance the risk of false positive (type-1) and false negative (type-2) error (Lakens et al., 2018).

Bonferroni outlier tests, from the ‘car’ package (Fox et al., 2012), were used to identify significant outliers for all ANOVAs. One data-point was removed at 12 pm, September 9<sup>th</sup> as it was a significant outlier (Bonferroni outlier test  $p = 0.045$ ). The reason for the outlier value was due to a corner shading of the sub-plot area by the nearby forest. ANOVAs on simple linear models were tested for assumptions of normality of residuals using the Shapiro-Wilks test in the core R stats package (R. Core Team, 2013), and of equal variance using Levene’s test in the ‘car’ package (Fox et al., 2012). For 12 pm September flights at the gravel pit site, the assumption of normality was not met, and a non-parametric Kruskal-Wallis test was therefore performed instead of the ANOVA. Normality and equal variance assumptions of ANOVA on linear mixed-effects models were checked visually through diagnostics plots (Appendix 4C), as the Shapiro-Wilks and Levene’s tests are considered too conservative for these models (Gelman and Hill, 2006; Winter, 2013; Zuur et al., 2009). No diagnostic plots of mixed-effects linear models appeared to indicate heteroscedasticity or non-normality issues.

#### 4.2.5.2. Comparing temperature response and ecological attributes

To test if there was a potential statistical relationship between surface temperature and the selected ecological attributes, correlation analyses were calculated for all sub-plots within each of the three study sites separately. Correlations of the different ecological attributes with each other were also performed to

ascertain potential interactions and underlying factors that may affect multiple attributes. Spearman's ranked correlation coefficient was used to detect monotonic relationships as not all variables were normally distributed (Appendix 4C). The 'Hmisc' package in R was used to perform the correlation analyses (Harrell Jr and Dupont, 2017)

Because ground vegetation data (richness, diversity, and cover) was collected during mid-June to mid-July it was correlated to July surface temperatures only, excluding temperature data from September. Only the surface temperature data from afternoon flights (12 to 4 pm) were used, excluding temperature from 8 pm flights as vegetation was expected to have a stronger effect on temperature during active photosynthesis and transpiration. Photosynthesis peaks during mid-afternoon and is mostly inactive by 8 pm in July at the latitude of the experiment (Raven et al., 2003 chap. 31). To control for the difference in solar radiation and weather at different days and times of the afternoon, the mean surface temperature of each sub-plot measurement was normalized and then averaged together over all July afternoon flights. For the gravel pit site and the 2016 reforestation site, temperature was normalized by dividing the mean temperature of each sub-plot with the combined mean temperatures of all sub-plots within that site during the same flight. For the 2015 reforestation site the already normalized net-temperatures, described in section 2.5.1, were used and no further normalization was needed.

Detecting correlation between ecological attributes and temperature requires that there be enough variation in the ecological attribute to overcome noise and uncertainty in measurements (Aggarwal and Ranganathan, 2016). To investigate the variation of ecological attributes between sites, we compared their standard deviation and examined whether they correlate significantly with temperature or not. Standard deviation was used as the measure of variation, rather than variance, to retain the unit of measure in comparisons. There is no specific level of variation that could be said to be 'enough', but by comparing between sites, we could potentially rule out lack of variation as a reason for non-correlation in a site. If sites with lower variation in an ecological attribute showed a significant correlation with temperature, and those with higher variation did not, then a lack of variation could be ruled out as a reason for non-significant correlation. Only attributes that were significant in one, but not all sites were included, as it was only with these variables that comparisons of variation and significant correlation between fields could be made.

### 4.3. Results

#### 4.3.1. Ecological attributes and surface temperature change over time by plot-type

##### 4.3.1.1. Gravel pit site in July and September

Species richness and inverse Simpson diversity of all plant species, and species richness of native forest species only, were significantly ( $p < 0.05$ ) higher in topsoil recipient plots compared to passive controls the gravel pit site in one-way ANOVAs (Table 4.2). There was no significant difference in % plant ground cover between the two plot-types in a one-way ANOVA.

**Table 4.2: Ecological attributes of topsoil recipient (TR) and passive control (PC) plots in the 2016 reforestation site sampled in June-July 2019. Mean values and mean difference between the two plot-types. Bolded mean differences were found to be significant ( $p < 0.05$ ) in one-way ANOVA. \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . NF sp. richness – Native forest plant species richness; all sp. richness – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plant**

	N	% ground cover	NF sp. richness	All sp. richness	All sp. diversity
TR mean	5	70	11.0	29.2	11.0
PC mean	5	43	3.8	14.8	5.2
TR-PC diff.		27	<b>7.2**</b>	<b>14.4***</b>	<b>5.8**</b>

Surface temperatures were significantly lower in the topsoil recipient plots, compared to the passive control plots during the afternoon (12- 4 pm) in July at the gravel pit site (Figure 4.2). The afternoon temperatures were buffered the least at 12 pm and 4 pm, with an estimated marginal mean difference of 6.2 °C (SE = 1.4), and the most at 2 pm, by 8.7 °C (SE = 1.12). The mean difference in surface temperature for the full afternoon (12-4 pm) was 7.1 °C.

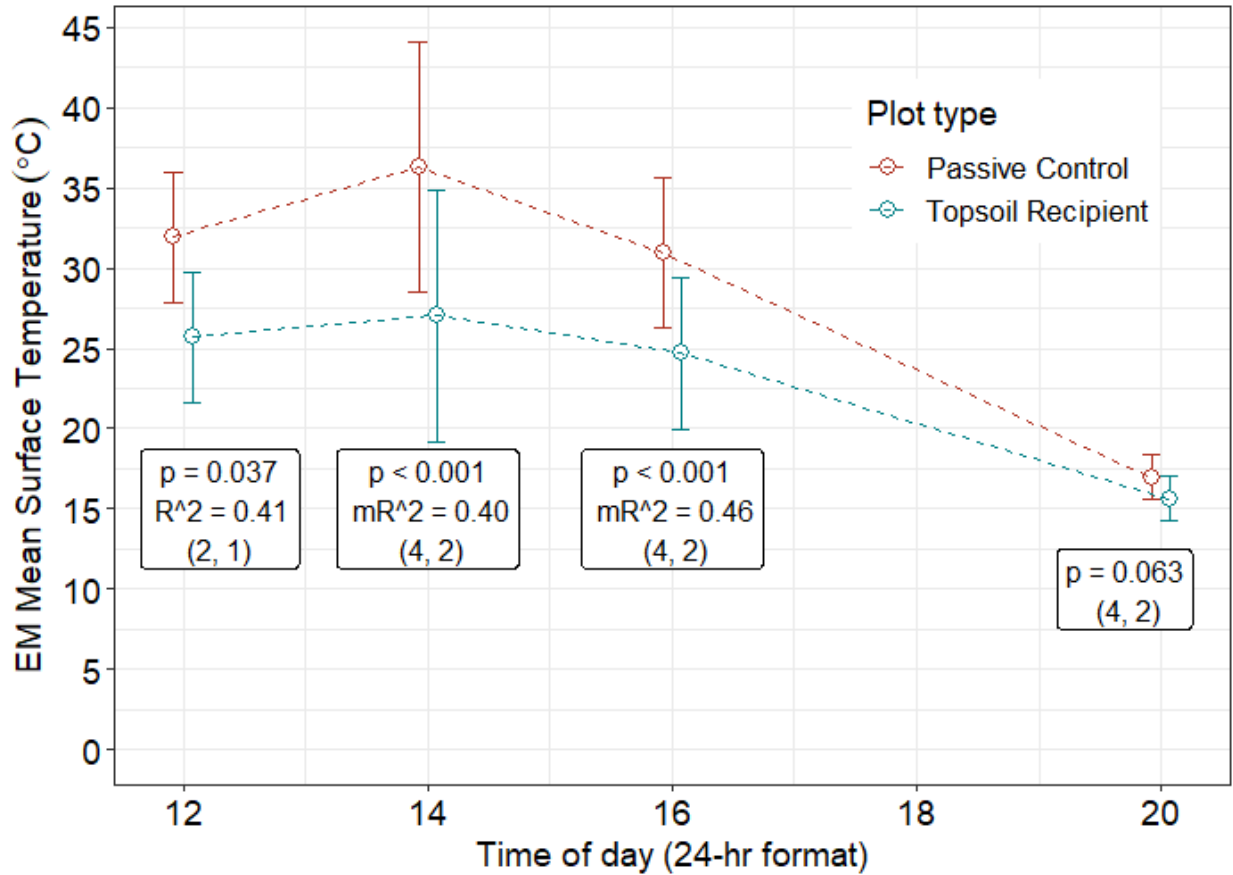


Figure 4.2: Estimated marginal (EM) mean surface temperature of topsoil recipient and passive control plots in the gravel pit site July 14<sup>th</sup>-15<sup>th</sup> 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships of temperature between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot ID. P-values from ANOVAs are presented in the boxes below each flight-hour, along with total number of flights and number of flight-days in parenthesis. For significant ( $p < 0.05$ ) differences  $R^2$  or marginal- $R^2$  ( $mR^2$ ) are also presented in the same box.

There were no significant ( $p < 0.05$ ) differences in surface temperature at any of the flight hours on September 17<sup>th</sup> for the gravel pit site (Figure 4.3).

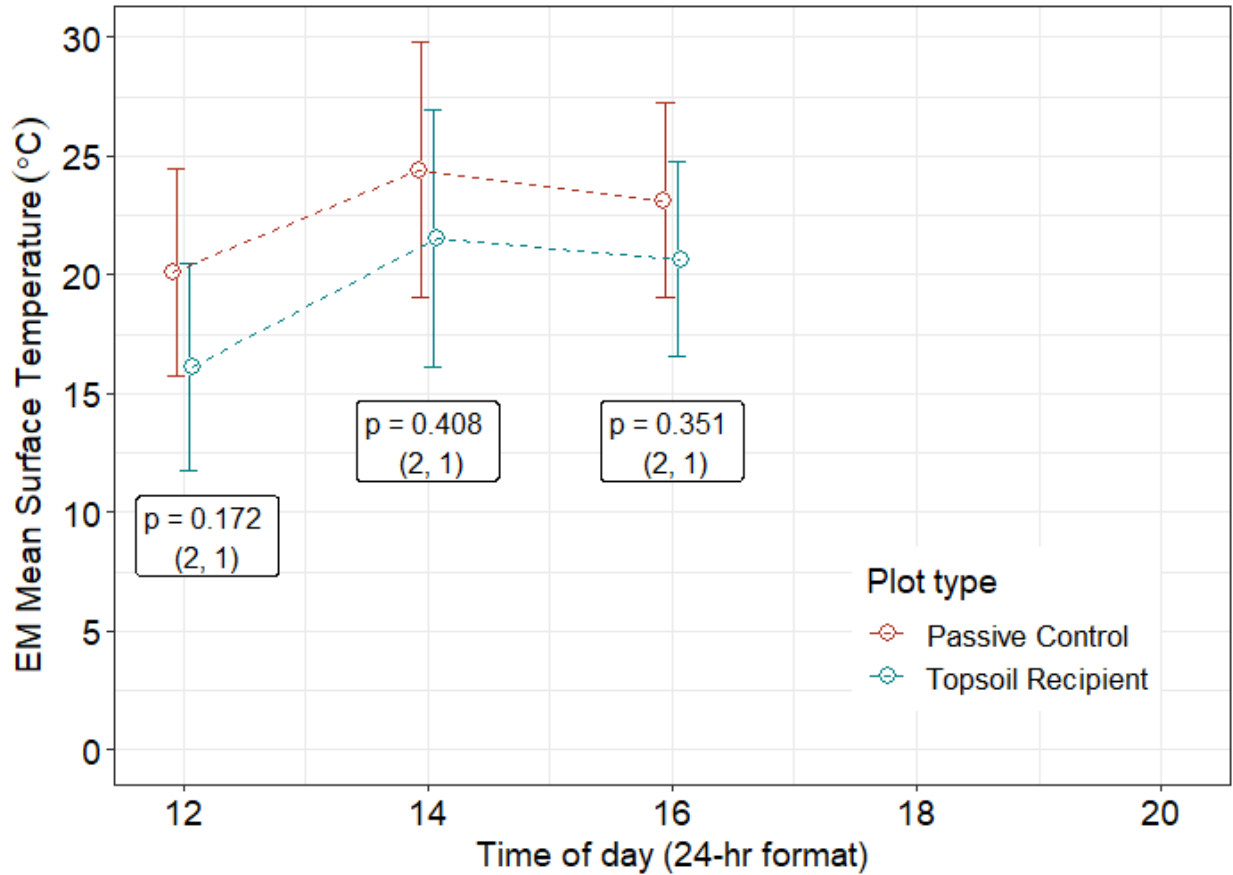


Figure 4.3: Estimated marginal (EM) mean surface temperature of topsoil recipient and passive control plots in the gravel pit site September 17<sup>th</sup>, 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships of temperature between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot. P-values from the ANOVAs, and marginal- $R^2$  ( $mR^2$ , only if p-value was significant), are presented in the boxes below each flight-hour, with total number of flights and number of flight-days in parenthesis. Kruskal-Wallis test used for 12 pm due to non-normality issues.

#### 4.3.1.2. 2016 reforestation site in July and September

The topsoil recipient plots in the 2016 reforestation site had significantly higher species richness (all plant species and native forest species only) and % ground cover than passive controls in the same site in one-way ANOVAs (Table 4.2). There was no significant difference in inverse Simpson species diversity of all plants or any of the tree measures between the two plot-types in a one-way ANOVA.

**Table 4.3: Ecological attributes of topsoil recipient (TR) and passive control (PC) plots in the 2016 reforestation site sampled in June-July 2019. Mean values and mean difference between the two plot- types. Bolded mean differences were found to be significant ( $p < 0.05$ ) in one-way ANOVA. \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . † Kruskal-Wallis test used as ANOVA did not fulfill assumptions. NF sp. richness – Native forest plant species richness; all sp. richness – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plant; Tree gen. rich. – Planted tree genera richness; Cum. tree height – Total cumulative tree height of planted tree; Cum. tree diameter – Total cumulative tree diameter of planted trees.**

	% ground cover	NF sp. richness	All sp. richness	All sp. diversity	Tree gen. richness	Cum. Tree height (cm)	Cum. tree diam. (mm)
TR mean	79	9.0	27.6	9.4	4.0	1482	236
PC mean	50	1.8	18.4	8.4	3.0	1592	219
TR-PC diff.	<b>29*</b>	<b>7.2***</b>	<b>9.2**</b>	1	1.0†	110	17

The topsoil recipient plots had significantly ( $p < 0.05$ ) lower surface temperatures than passive control plots at 12, 2 and 4 pm, but not at 8 pm, in the 2016 reforestation site in July (Figure 4.4). The difference in estimated marginal mean afternoon temperature was highest at 2 pm by 2.1 °C (SE = 0.4), with a mean difference of 1.9 °C at 12 pm (SE = 0.4) and 1.9 °C at 4 pm (SE = 0.5). The overall mean afternoon difference in temperature was 2.0 °C.

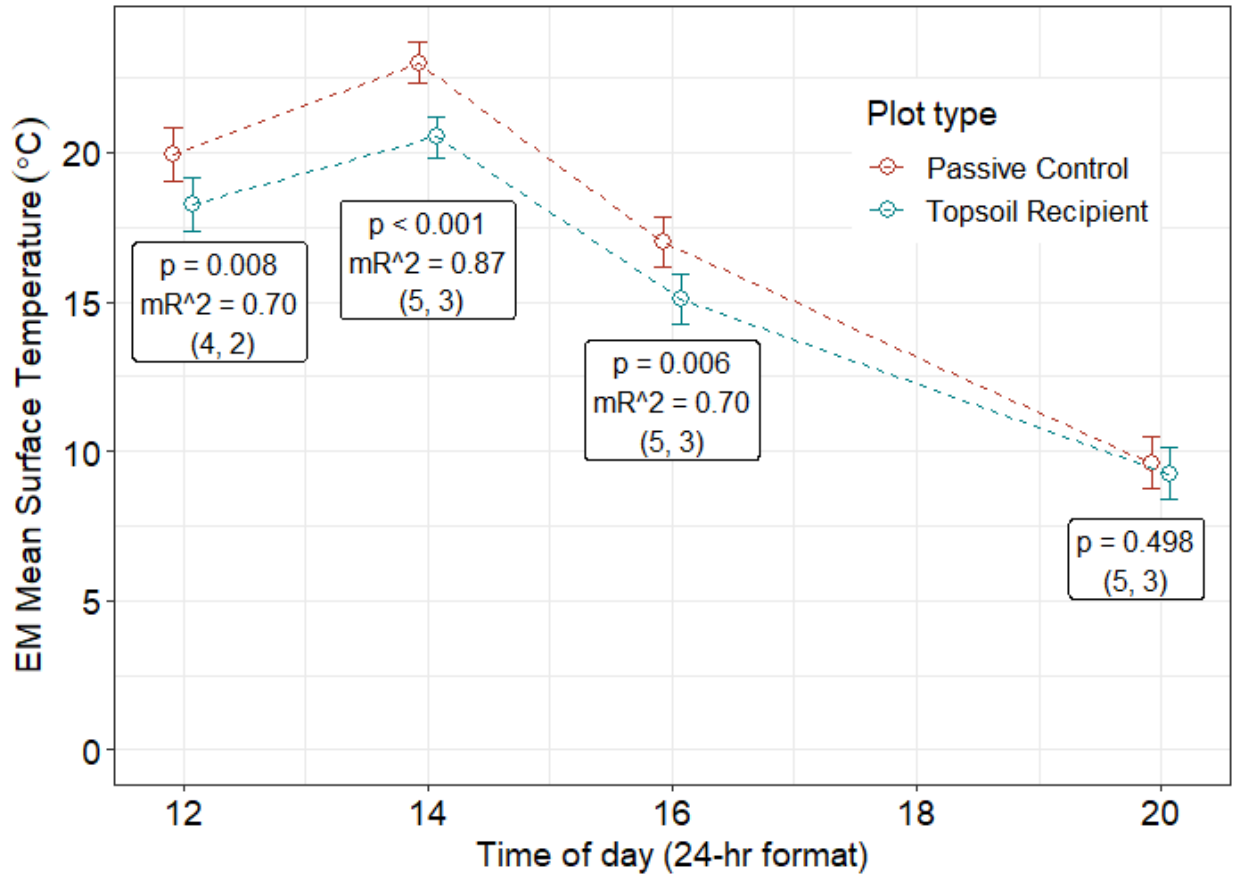


Figure 4.4: Estimated marginal (EM) mean surface temperature of topsoil recipient and passive control plots in the 2016 reforestation site July 7<sup>th</sup>-9<sup>th</sup> 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships of temperature between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot. P-values from the ANOVAs are presented in the boxes below each flight-hour, with total number of flights and number of flight-days in parenthesis. Marginal-R<sup>2</sup> (mR<sup>2</sup>) are presented in the same box if differences were significant.

At the 2016 reforestation site in September, topsoil recipient plots had significantly ( $p < 0.05$ ) lower temperatures than passive control plots at 2 and 4 pm, but not at 12 pm (Figure 4.5). The estimated marginal mean difference between plot-types was 1.8 °C (SE = 0.5) at 2 pm and 1.7 °C (SE = 0.4) at 4 pm.



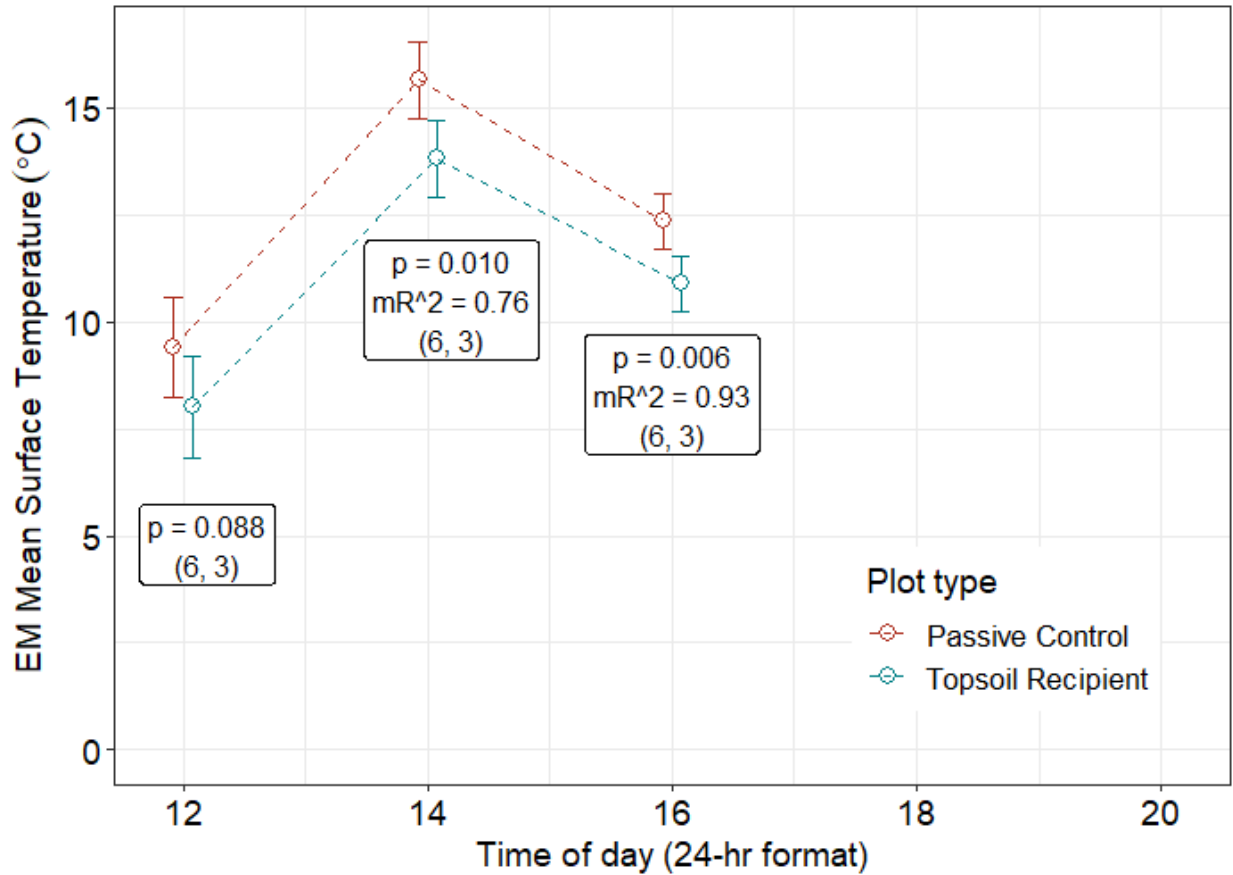


Figure 4.5: Estimated marginal (EM) mean surface temperature of topsoil recipient and passive control plots in the 2016 reforestation site September 7<sup>th</sup>-9<sup>th</sup> 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships of temperature between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot. P-values from the ANOVAs, and marginal-R<sup>2</sup> ( $mR^2$ , only if p-value was significant), are presented in the boxes below each flight-hour, with total number of flights and number of flight-days in parenthesis.

#### 4.3.1.3. 2015 reforestation site in July and September

There were no significant differences in any ecological attributes between topsoil recipient plots and either the mow-and-spray control plots or passive control plots in the 2015 reforestation site (Table 4.4).

**Table 4.4: Ecological attributes of topsoil recipient (TR), mow-and-spray control (MSC), and passive control (PC) plots in the 2015 reforestation site sampled in June-July 2019. Mean values and mean difference between the three plot-types. Bolded mean differences were found to be significant ( $p < 0.05$ ) in one-way ANOVA. \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . † Kruskal-Wallis test used as ANOVA did not fulfill assumptions. NF sp. richness – Native forest plant species richness; all sp. richness – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plant; Tree gen. rich. – Planted tree genera richness; Cum. tree height – Total cumulative tree height of planted tree; Cum. tree diameter – Total cumulative tree diameter of planted trees.**

	N	% ground cover	NF sp. richness	All sp. richness	All sp. diversity	Tree gen. richness	Cum. tree height (cm)	Cum. tree diam. (mm)
TR mean	5	94	5.8	27.6	11.0	2.8	1295	227
MSC mean	5	96	2.6	20.2	8.2	3.4	1384	180
PC mean	5	87	1.8	21.4	10.2	2.6	945	151
TR-MSC diff.		-2†	3.2	7.4	2.8	-0.6	-89	47
TR-PC diff.		7†	3.0	6.2	0.8	0.2	250	76

During the July flights, mean temperature of mow-and-spray control plots in the 2015 reforestation site was significantly ( $p = 0.046$ ) lower than passive control plots at 4 pm by a mean of 2.4 °C (SE = 0.9) (Figure 4.6A). There were no significant differences in mean surface temperature between any plot-types at any other hours.

When normalizing surface temperatures of each sub-plot by their adjacent surroundings as ‘net-temperatures’ the pattern changed. Net-temperature of topsoil recipient plots were significantly lower than passive controls ( $p = 0.025$ ) and mow-and-spray control plots ( $p = 0.028$ ), but only at 8 pm (Figure 4.6B) as per a post-hoc Tukey HSD test. The estimated marginal mean difference of net-temperature at 8 pm between the topsoil recipient plots, and both the mow-and-spray plots and passive control plots, was 0.7 °C (SE = 0.2).

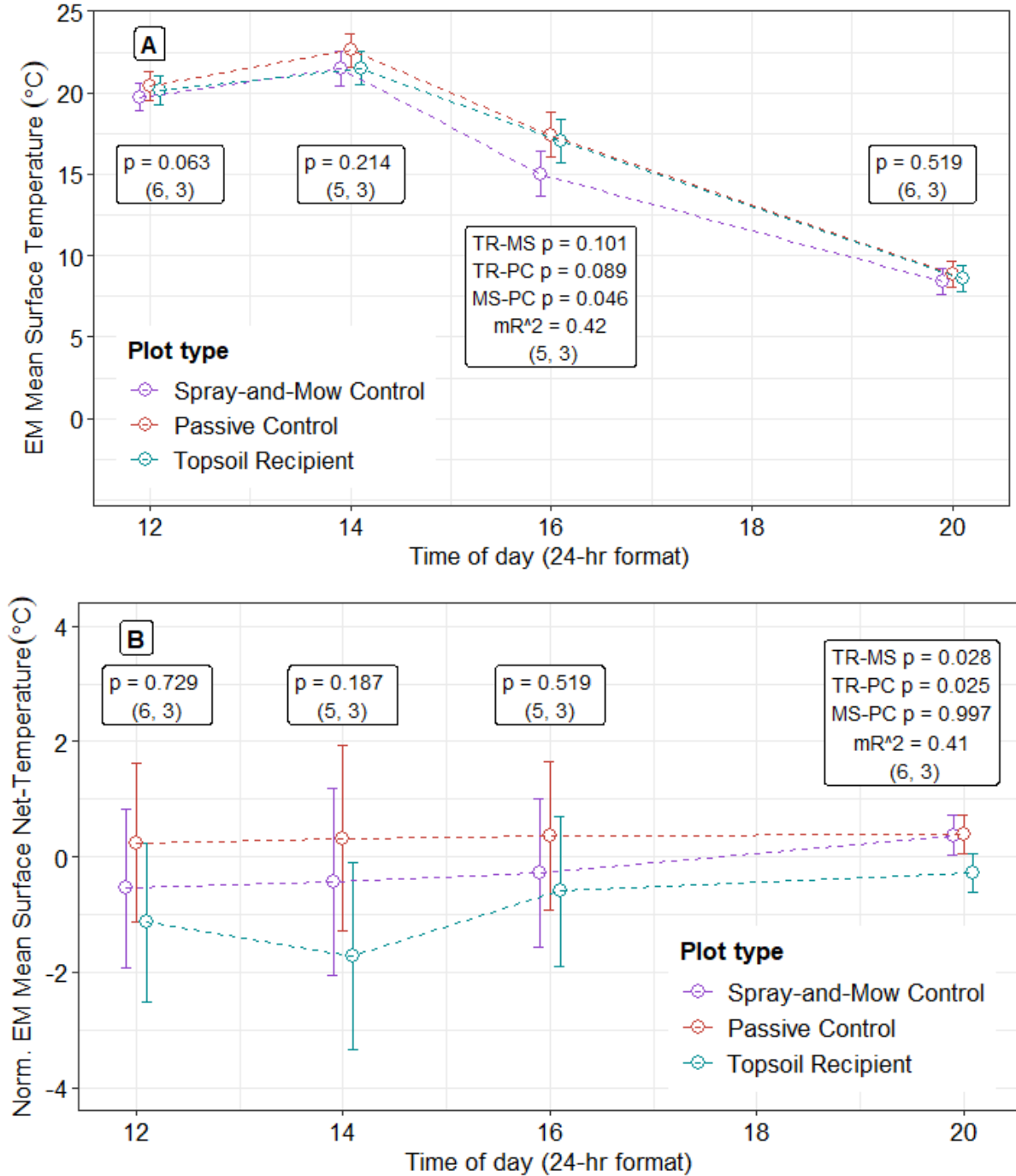
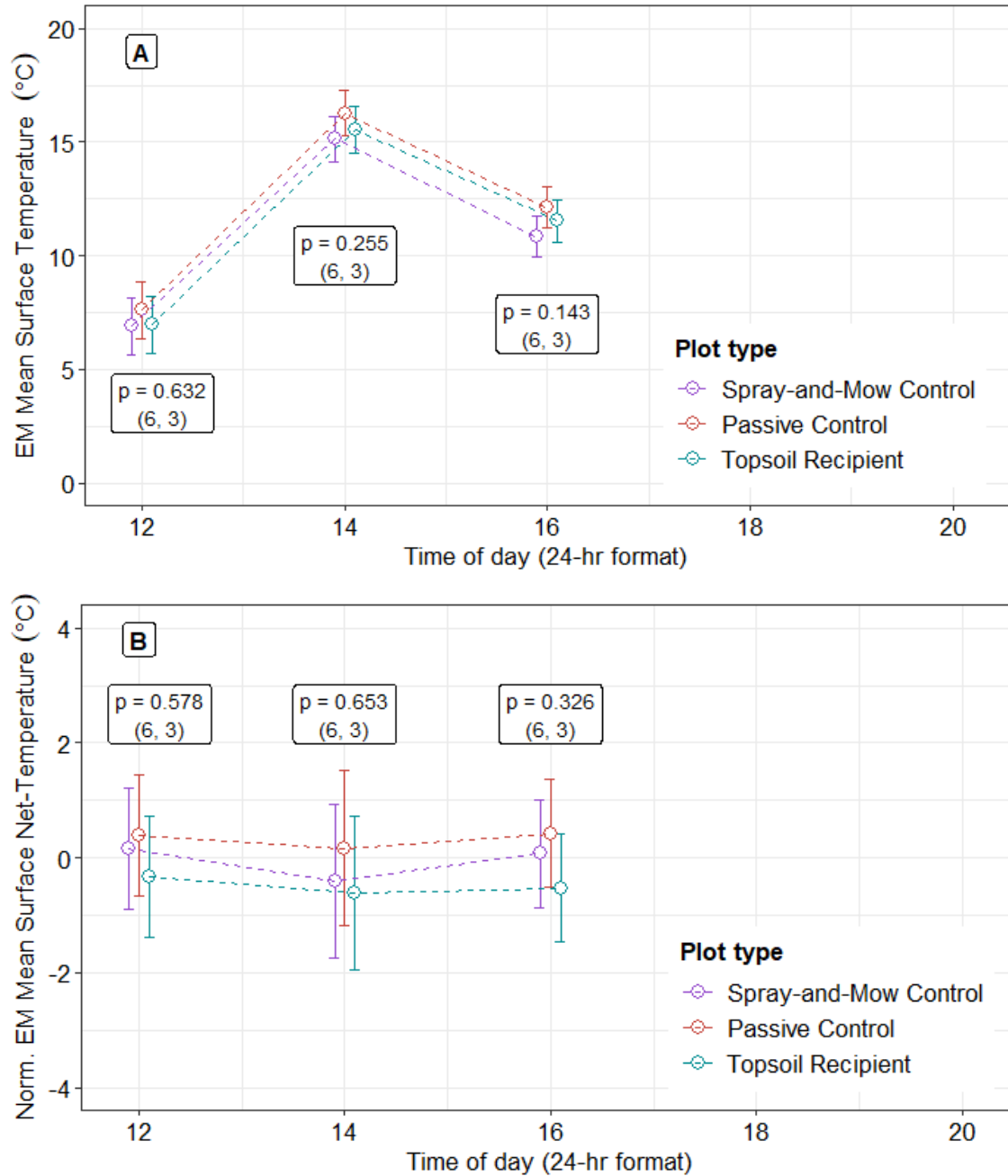


Figure 4.6: (A) Estimated marginal (EM) mean surface temperature and (B) normalized EM mean surface net-temperature of topsoil recipient, mow-and-spray control, and passive control plots in the 2015 reforestation site, July 7<sup>th</sup>-9<sup>th</sup> 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot. P-values from ANOVAs are presented in the boxes above each flight-hour. For the significant ANOVAs, post-hoc Tukey HSD pairwise p-value and marginal  $R^2$  of fixed effects ( $mR^2$ ) are presented in the box. Total number of flights and number of flight-days are presented in the boxes in parenthesis.

There were no significant ( $p < 0.05$ ) differences in either mean temperature or normalized net-temperature between plot-types for September at any flight-hour (Figure 4.7).



**Figure 4.7:** (A) Estimated marginal (EM) mean surface temperature and (B) normalized EM mean surface net-temperature of topsoil recipient, mow-and-spray control, and passive control plots in the 2015 reforestation site September 7<sup>th</sup>-9<sup>th</sup> 2019 by time of day. Results from multiple flights on multiple days are grouped by flight-hour, presented on the x-axis. Dashed lines are only visual aids for grouping by plot-type and do not pre-suppose linear relationships between flight-hours. Error bars at each flight-hour represent the 95 % confidence interval ( $p < 0.05$ ) of temperatures by plot-type when taking into account the effect of date and plot. P-values from the ANOVAs are presented in the boxes below each flight-hour, with total number of flights and number of flight-days in parenthesis.

#### 4.3.1.4. Contrasting the effect of topsoil transfer on ecology and temperature between sites

Topsoil recipient plots in both the 2016 reforestation site and the gravel pit site both had significantly higher plant species richness (all and native forest species only) and significantly lower July afternoon temperatures compared to passive control plots. Topsoil recipient plots also had higher % plant cover in the 2016 reforestation site and higher inverse Simpson diversity of all plants in the gravel pit site. In contrast, the 2015 reforestation site, with no statistically significant differences in ecological attributes, only showed a significant net-temperature difference between topsoil recipient plots and both control plot-types at 8 pm in July.

### 4.3.2. Surface temperature response and ecological attributes

#### 4.3.2.1. Correlations of temperature and ecological attributes within fields

At the gravel pit site, there was a significant ( $p < 0.05$ ) monotonic correlation between lower normalized surface temperature ratio and higher native forest plant species richness and % plant ground cover (Figure 4.8). Native forest species richness and % plant ground cover also correlated significantly and positively (0.79) with each other. Species richness and inverse Simpson species diversity of all plant species were not significantly correlated with temperature.

	Temp. norm.	Sp. diversity	All. sp. rich.	NF. sp. rich
% ground cover	<b>-0.6**</b>	0.28	0.37	<b>0.79***</b>
NF sp. rich.	<b>-0.47*</b>	<b>0.57**</b>	<b>0.70***</b>	
Sp. rich.	-0.29	<b>0.92***</b>		
Sp. diversity	-0.28			

**Figure 4.8: Matrix of Spearman rank correlation coefficients for correlations of temperature (bolded outline) and ecological variables within the gravel pit site. Numbers explain the direction and strength of the Spearman correlation coefficient. Significant ( $p < 0.05$ ) correlations bolded and: \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . NF sp. rich. – Native forest plant species richness; all sp. rich. – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plants; Temp. norm. – Normalized surface temperature.**

At the 2016 reforestation site, a lower normalized surface temperature correlates significantly ( $p < 0.05$ ) with a higher plant species richness of all species, native forest plant species richness, and % plant ground cover (Figure 4.9). There were no significant correlations between temperature and cumulative tree diameter, cumulative tree genus richness, or the inverse Simpson diversity of all plant species.

	Temp. norm.	Tree diameter	Tree height	Tree gen. rich.	All sp. diversity	All sp. rich.	NF. sp. rich
% ground cover	<b>-0.63**</b>	-0.22	-0.27	0.23	0.27	0.3	<b>0.46*</b>
NF sp. rich.	<b>-0.64**</b>	-0.13	-0.16	0.39	0.34	<b>0.76**</b>	
All sp. rich.	<b>-0.58**</b>	-0.21	-0.20	0.23	<b>0.47*</b>		
Sp. diversity	-0.39	-0.22	-0.19	-0.01			
Tree gen. rich.	-0.31	0.27	0.06				
Tree height	0.19	<b>0.88***</b>					
Tree diameter	0.02						

**Figure 4.9: Matrix of Spearman rank correlation coefficients for correlations of temperature (bolded outline) and ecological variables within the 2016 reforestation site. Numbers explain the direction and strength of the Spearman correlation coefficient. Significant ( $p < 0.05$ ) correlations bolded and: \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . NF sp. rich. – Native forest plant species richness; all sp. rich. – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plants; Tree gen. rich. – Planted tree genera richness; Tree height – Total cumulative tree height of planted tree; Tree diameter – Total cumulative tree diameter of planted trees; Temp. norm. – Normalized surface temperature.**

At the 2015 reforestation site, there was a significant ( $p < 0.05$ ) negative correlation between normalized surface temperature and native forest understory species, tree diameter, and tree height (Figure 4.10). There was no significant correlation at this site between temperature and % ground cover, species richness of all plants, species diversity of all plants, or tree genera richness.

	Temp. norm.	Tree diameter	Tree height	Tree gen. rich.	Sp. diversity	All Sp. rich.	NF. sp. rich
% ground cover	-0.02	0.05	-0.08	-0.05	0.23	0.31	<b>0.34*</b>
NF sp. rich.	<b>-0.37*</b>	0.21	0.01	0.04	<b>0.81***</b>	<b>0.84***</b>	
All sp. rich.	-0.22	0.17	-0.02	0.09	<b>0.92***</b>		
Sp. diversity	-0.15	0.03	-0.16	-0.06			
Tree gen. rich.	-0.14	<b>0.48**</b>	<b>0.52**</b>				
Tree height	<b>-0.57***</b>	<b>-0.93***</b>					
Tree diameter	<b>-0.57***</b>						

**Figure 4.10: Matrix of Spearman rank correlation coefficients for correlations of temperature (bolded outline) and ecological variables within the 2016 reforestation site. Numbers explain the direction and strength of the Spearman correlation coefficient. Significant ( $p < 0.05$ ) correlations bolded and: \*  $p < 0.05$ , \*\*  $p < 0.01$ , \*\*\*  $p < 0.001$ . NF sp. rich. – Native forest plant species richness; all sp. rich. – Species richness of all plants; all sp. diversity – Species inverse Simpson diversity of all plants; Tree gen. rich. – Planted tree genera richness; Tree height – Total cumulative tree height of planted tree; Tree diameter – Total cumulative tree diameter of planted trees; Temp. norm. – Normalized surface temperature.**

Native forest plant species richness was the only variable that correlated significantly with normalized surface temperature in all sites, with a Spearman correlation between -0.37 and -0.64. Native forest plant species richness in turn positively and significantly correlated with % plant ground cover at all sites; likely due to both these factors increasing with forest topsoil transfer.

#### 4.3.2.2. Variation in ecological attributes and their correlation with temperature contrasted

Species richness of all plants, % plant ground cover, and cumulative tree height and tree diameter had significant correlations with surface temperature in some, but not all sites. For % plant ground cover, cumulative tree height, and tree diameter, the lowest standard deviation was found at sites where no significant correlation with temperature was found (Table 4.1). The lack of variation in these attributes could therefore be a reason for the lack of correlation with temperature in the respective sites. For species richness of all plants, the relationship was the opposite. The two sites with a significant correlation between surface temperature and species richness of all plants had the lowest and highest variation in terms of standard deviation. The lack of correlation between species richness of all plants with

temperature in the 2015 reforestation area cannot, therefore, be ascribed to a lack of variation in the mean number of plant species between sub-plots.

**Table 4.5: Comparison of variation (standard deviation - SD) of ecological variables by sub-plots in the three sites and whether the variable correlated significantly (sig. corr.) with surface temperature. Only variables that were significantly related to temperatures in some, but not all, sites are included.**

	Sp. richness of all plants		Plant ground cover (%)		Tree height (cm - sum)		Tree diameter (mm - sum)	
	Sig. corr. (p)	SD	Sig. corr. (p)	SD	Sig. corr. (p)	SD	Sig. corr. (p)	SD
			Yes					
Gravel pit	Yes (0.022)	9.5	(<0.001)	18.6	Not included		Not included	
RF 2016	Yes (<0.001)	4.7	Yes (0.003)	19.5	No (0.442)	344	No (0.946)	50
RF 2015	No (0.212)	7.1	No (0.898)	10.0	Yes	465	Yes	66

### 4.3.3. Summary of results

Two questions were asked in this study, here we summarize the answer to each:

- (1) Do plots with forest topsoil, added two growing seasons prior, cool and buffer surface temperatures more than non-topsoil recipient control plots; and if so by how many degrees at different times of the day and season?
  - **Topsoil recipient plots had significantly lower surface temperature than controls in the gravel pit site (7.0 °C), and in the 2016 reforestation site (2.0 °C) during July afternoons (12 – 4 pm). At both these sites, the difference in temperature buffering effect was most pronounced at 2 pm.**
  - **Topsoil recipient plots in the 2015 reforestation site had significantly lower normalized surface net-temperatures than both passive and mow-and-spray control plots at 8 pm in July only. At all other flight-hours, in September, and using non-normalized temperatures there were no significant differences between topsoil recipient plots and control plots for this site.**
  - **For September flights, only the 2016 reforestation site showed a significant difference, where surface temperature of topsoil recipient plots were 1.7 °C lower than controls at 2 pm and 4 pm.**
- (2) Are there significant correlations between summer daytime surface temperature and ground-measured ecological attributes, including plant cover, plant species richness and diversity of all plants, native forest plant species richness, and tree size; and if so, what are the patterns of correlations within and between sites?



- **In all three sites, lower temperatures were significantly correlated with higher native forest plant species diversity (Spearman  $r = -0.37$  to  $-0.64$ ).**
- **Higher total plant species diversity was only significantly correlated with lower temperatures in the 2016 reforestation site ( $r = -0.58$ ).**
- **In the gravel pit and 2016 reforestation site a higher % of ground covered by living plants was significantly correlated with lower temperatures ( $r = -0.60$  and  $r = -0.63$ ).**
- **In the 2015 reforestation site higher cumulative tree size (diameter and height) was correlated with lower temperatures ( $r = -0.57$  and  $-0.57$  respectively).**

## **4.4. Discussion**

### **4.4.1. Thermal buffering and its applications in evaluating restoration and its benefits**

Previous studies on surface or air temperature of more diverse forest ecosystems have mostly compared these to other types of ecosystems; for example, comparing mature forests to grasslands, burns, pastures (e.g. Luvall et al., 1990; Luvall and Holbo, 1989; Wagendorp et al., 2003), croplands (e.g. Kutsch et al., 2001; Maes et al., 2011) or monoculture tree plantations (e.g. Lin et al., 2020, 2017a; Norris et al., 2012). In all cases, these were comparisons of very dissimilar ecosystems with large differences in potential transpiration, structure, and function of plants. In contrast, the addition of forest topsoil in this study was created a relatively subtle difference, at least for the two reforestation sites. At these sites, the forest topsoil was added under and around already planted trees and on top of already existing agricultural soil. In the 2016 reforestation site, the topsoil recipient plots did not differ significantly in tree size, only in plant species richness, by a mean of 9.2 additional species and % plant ground cover, by a mean difference of 29 % (Table 4.3), but that was enough for afternoon surface temperature in July to differ significantly, by 2.0 °C (Figure 4.4). Finding a significant summer daytime cooling effect at topsoil recipient plots that are otherwise ecologically similar to its control, two years after the transfer took place, shows that even a relatively small difference ecosystem state can be detected through thermal imaging from UAV.

For the 2016 reforestation site and the gravel pit site, the largest surface temperature difference between plot-types occurred during 2 pm flights in July. In comparison, an earlier study also using thermal imaging from UAV found the largest surface temperature difference between diseased and non-diseased trees in a semi-naturalized tree plantation between 12 and 1 pm in August (Smigaj et al., 2019). That study did however not fly at 2 pm and had increased cloud cover as the one flight-day progressed (Smigaj et al., 2019). It is probable that the timing of maximum difference will vary by latitude, time-zones, and season. Finding generalizable patterns in the timing of the largest temperature difference can

aid in creating best-practices of when to fly, for when the objective is to search for ecosystem state change through thermal imaging.

It is not surprising that the surface temperature difference between topsoil recipient and control plots was found to be highest in the afternoon of July; photosynthetic activity and transpiration would be nearer their seasonal and circadian peak at these times compared to evenings and to September (Bonan, 2008 chap. 27 and 30; Raven et al., 2003 chap. 29). More surprising was the result of cooling at soil transfer plots in the 2015 reforestation site only being significant at 8 pm in July. Due to a lack of incoming sunlight and circadian rhythm of plant activity (Raven et al., 2003 chap. 29), changes in surface temperature at this time of day may have more to do with the physical heat capacity and respiration of plants than with transpiration. These evening cooling effects and the mechanisms behind it warrant further study.

As discussed in Chapter 3, the forest topsoil available for transfer is in limited supply, and when considering where to transfer it, one site-selection criterion is where the soil can do the most in terms of improving ecosystem services (Box, 2003; Holl and Aide, 2011). In the case of the service of cooling and thermal buffering, the gravel pit site was cooled more than the two reforestation sites by the topsoil, with a mean July afternoon cooling effect of 7.1 °C.

The ability to buffer temperatures provided by forests in both urban and rural areas has been considered an ecosystem service under the umbrella of climate regulation (Betts et al., 2018; Costanza et al., 2014; Lin et al., 2020). And, while the service of thermal buffering has been discussed within the context of urban heat island effects (e.g. Jenerette et al., 2011; Livesley et al., 2016; Rafiee et al., 2016) and green-roofs (e.g. MacIvor et al., 2016; Simmons et al., 2008), it has not yet been explored within the field of restoration ecology. Many large reviews and studies on ecosystem services affected by restoration have not included thermal buffering, or similar concepts, at all (e.g. Barral et al., 2015; Birch et al., 2010; Bullock et al., 2011); even though the value of thermal buffering to restoration funders and stakeholders could be high.

Quarries and pits, with their mainly bare ground, and often minimal soil cover (Chenot et al., 2018; Tomlinson et al., 2008) can become very hot, potentially influencing the climate of nearby ecosystems. One study has shown a mean summer surface temperature of a quarry to be 20 °C higher than nearby forests (Holbo and Luvall, 1989). Asphalt and gravel roads, which are similar in surface to quarries and pits, but usually narrower, have been shown to increase air-temperature up to 50 m into adjacent forests (e.g. Delgado et al., 2007; Pohlman et al., 2009; Tuff et al., 2016; Wright et al., 2010). In a study of temperature edge effects from agricultural fields, increased air-temperatures were found 80 m into the forest (Robinson et al., 2009). Agricultural fields are less narrow than roads, but have more soil and vegetation, and are often cooler than quarries and pits (Holbo and Luvall, 1989). It is therefore

possible that the temperature edge effect of quarries and pits on bordering ecosystems are even more pronounced than either roads or agricultural fields.

Temperature edge effects of quarries and pits on nearby ecosystems may change animal and plant habitat suitability as all organisms have a certain temperature tolerance range and climate envelope ([Hijmans and Graham, 2006](#); [Jiguet et al., 2007](#); [Tuff et al., 2016](#)). One review argues that temperature edge effects especially reduce habitat for animals that require stable temperatures, including ectothermic lizards and certain bird species (Tuff et al., 2016). If temperatures increase beyond species-specific tolerance levels, the increased heat can also directly stress and damage plants in both natural landscapes and agriculture (Michaletz et al., 2015; Wahid et al., 2007).

For the 6527 surrendered pits and quarries which need to be, or have been, restored in Ontario, 30.2 % are bordered by ‘natural’ terrestrial landscapes, and 15.6 % border residential areas (TOARC, 2020). Based on the results of our study, topsoil transfer restoration could mitigate the excess summer temperatures of a quarry or pit by 36 % (a reduction of 7.1 °C from 20 °C increase) in only two years, potentially decreasing the former quarry or pit’s effect on the ecosystems which surround it. With increasing evidence of the ability of restoration to increase the rate of thermal buffering from this study and others (Aerts et al., 2004; Hamberg et al., 2020), thermal buffering should be taken into consideration as an ecosystem service when accounting for the benefits of restoration projects. With their ability to thermally image the same site multiple times per day, UAV mounted thermal cameras could become useful tools for evaluating the positive impact of local thermal buffering of a restoration project at different times of the day. Thermal imaging from UAV to evaluate restoration benefits may be particularly relevant for projects and experiments where the narrowest part of the site is > 100 m (at or below most thermal imaging satellite pixel size). These could include linear restoration projects of former power-corridors, roads, stream-banks and seismic lines for oil-exploration (Finnegan et al., 2019), or projects where the individual sites of restoration are small, such as compensatory restoration of smaller farm fields, quarries, and pits.

To operationalize restoration evaluation and monitoring through UAV thermal imaging more research is needed. Further research needs to test the technology’s reliability and accuracy in detecting ecosystem state change in different types of diverse non-production ecosystems, as well as detecting common ecological issues, including tree disease, water-stress, and invasive species. Studies are also needed on how practitioners would use this technology and what they see as major barriers (e.g. cost, knowledge, awareness) to implementation.

#### 4.4.2. Surface temperature and its relationship with ecological attributes

Low variation in % ground covered by living plants in the 2015 reforestation may explain why it was the only site with no significant correlation between % plant ground cover and temperature (Table 4.5). Conversely, the higher variation in measures of tree size in the same site may explain why it was the only site in this study with significant correlations between tree size and temperature. The higher variation in cumulative tree size in plots in the 2015 reforestation site can be explained by the site having been established for one year longer than the 2016 reforestation site. The 2015 reforestation site also had one more year without active management that attempts to reduce mortality and competition (irrigation, mowing, application of glyphosate herbicide, and trapping of rodents) than the 2016 reforestation site. The more varied terrain of the 2015 reforestation site could also have affected the soil-moisture and sunlight available to the trees throughout the site, leading to more uneven growth than in the flatter 2016 reforestation site.

Higher native forest species richness was correlated with lower normalized surface temperature at all sites in this study, but species richness and diversity of all plant species were not. As discussed in the introduction of this study, and in Hamberg et al. (2020) (Chapter 2, Section 4), the connection between native species richness or diversity and temperature is not direct but is possibly related to variation in structure (e.g. leaf-shape and root depth), function (e.g. types of photosynthesis), and trophic complexity (e.g. mycorrhizae associations) (Hooper et al., 2005; Loreau, 1995). These variations in structure and function in turn increase potential transpiration (Baldocchi, 2005) and other plant activity that absorbs and transforms energy, and thereby decreases temperatures (Schneider and Kay, 1994).

In this study, many of the non-native and/or non-forest plant species already present in the recipient sites before topsoil transfer were herbaceous and graminoid ruderal field species. In comparison, the native forest species introduced through soil transfer included herbaceous plants, ferns, woody shrubs, and trees. On a continuum between an old-field ecosystem to mature forest, the understories of the non-topsoil control plots are located more towards the old field, and the topsoil recipient plots are closer to the mature forest. It has been observed that structural and functional diversity increases in succession of old fields towards forest (e.g. Bazzaz, 1975; Dölle et al., 2008; Guariguata and Ostertag, 2001; Perino and Risser, 1972; Tatonni and Roche, 1994). The assemblages of different types of native forest plants introduced from the forest topsoil may be structurally, and functionally, more diverse than the non-native and/or non-forest plants.

Another issue which warrants study and experimentation is how to separate the effect on surface temperature of cover and biomass from that of diversity. Both cover and diversity were correlated significantly, although not perfectly, with each other and they have direct and indirect effects on each

other. For example, more ground covered by living plants may allow for more species richness and diversity to be found in sampling, and higher species diversity may increase % of ground covered by plants as it may increase biomass (Hughes and Roughgarden, 2000; Tilman et al., 2001; Van Ruijven and Berendse, 2003). To disentangle the effects of the different ecological attributes, functions, structures, and mechanisms on surface temperature, at different times of the day and night, I suggest that the next step is to conduct UAV thermal-imaging of a physically controlled plot experiment. Long term plot experiments which vary species and functional diversity, but control other factors, such as those in the International Diversity Experiment Network with Trees (IDENT) (Tobner et al., 2014), may be ideal testing sites. These ideas are explored in more detail in Section 5.3.2 of Chapter 5.

## Chapter 5. Conclusion

*“You need coolin', baby, I'm not foolin'” – Whole Lotta Love, Led Zeppelin, 1969*

### 5.1.1. Summary of results on ecosystem change and surface temperature

Like much of ecology and remote sensing science, this dissertation was about finding the signal among the noise. In this case, the signal is produced by ecosystem change and recovery, assisted by restoration, causing differences in outgoing thermal energy radiation, as measured through thermally imaged surface temperature. There are many sources of noise that may obscure the signal, including wind, errors in imaging receptors, emissivity, terrain, clouds, and errors and uncertainties in ecological sampling. Overall, the results of this dissertation showed that the signal was both statistically and ecologically significant and detectable above the noise. The results are synthesized below by the questions from Chapter 1:

#### **1. Does restoration of wooded ecosystems decrease its daytime summer surface temperature, and if so, by how much, either over time or by type of restoration method?**

Yes, the results from both projects indicate a decrease in daytime temperature over time or by type of restoration method. The statistical analysis in Chapter 2 provides evidence that oak woodland restoration caused a significant reduction in normalized midday summer surface temperature in 30 out of 31 fields, by an average of 1.5 percentage points per year, translating to 4.5 °C in 12 years.

Chapter 3 showed that topsoil recipient plots in all sites in the second restoration project studied were more similar to the reference forest where the topsoil was taken from, than to the control plots in the recipient sites in terms of plant species community composition. This indicated that the topsoil transfer had caused ecological change and that we could expect to be able to find surface temperature differences. In Chapter 4, a consistent pattern of surface temperature differences was indeed found in 2 out of 3 topsoil recipient sites. In the gravel pit site, the mean temperature difference between topsoil transfer plots and control plots was 7.0 °C and in the younger reforestation site (2016) it was 2.0 °C. These differences were significant for flights at 12, 2, and 4 pm in July. In the older and more heterogeneous (2015) reforestation site, there was a significant reduction in normalized ‘net-temperature’ of topsoil plots only at 8 pm in July, with a difference of 0.7 °C compared to controls. Because there was no significant ecological change found in Chapter 3 in sub-plots that received additional woody debris or shrub plantings, these treatments were not included in surface temperature comparisons in Chapter 4.

**2. When during the day is the thermal buffering effect of the restoration the most pronounced and could restoration increase relative nighttime temperatures, thereby decreasing diurnal temperature variation?**

In Chapter 4, the maximum difference/buffering occurred at 2 pm in July (mid growing season) for both the gravel pit site (8.7 °C) and the 2016 reforestation site (2.1 °C). In these two sites, there was no significant difference/buffering at 8 pm in July. Conversely, as noted in the answer above, in the 2015 reforestation site 8 pm was the only time when the temperature of topsoil recipient plots was significantly lower than controls. Later in the growing season, in September, only the 2016 reforestation site showed a significant difference/buffering, where topsoil plots were 1.8 and 1.7 °C cooler at 2 pm and 4 pm respectively.

In Chapter 2, a significant increase in normalized summer nighttime surface temperature of 0.2 °C per year since restoration was found. The decrease in diurnal temperature variation per year since restoration was 4 percentage points, which translated to a total mean reduction of 5 °C in 8 years.

**3. What is the relationship between plant species diversity (native and exotic) and surface temperature, when taking into account total plant cover and other ecological attributes?**

In Chapter 4, the two sites where thermal buffering of topsoil recipient plots was most pronounced (gravel pit site and 2016 reforestation site) were also the ones where there were significant differences in plant species richness, diversity, or % plant cover (as measured and tested in Chapter 3). When tested through Spearman correlation analysis, all three sites in the study showed a significant negative monotonic correlation between native forest plant species richness and surface temperature. Some, but not all, sites showed significant correlations between surface temperature and % of plant ground cover, tree height, tree diameter, and all plant species richness and diversity.

In Chapter 2, in the three oak woodland restoration fields that had been in-situ sampled, there was a significant relationship between relative summer daytime surface temperature and plant species diversity over time, in a multiple linear regression model. Plant species diversity had a more significant relationship with surface temperature than plant ground cover, canopy cover, and the number of woody stems. One additional species corresponded with a decrease of summer daytime surface temperature of 5.06 percentage points, which translates to approximately 0.3 °C. When comparing native and non-native plant species separately, only the native species had a significant relationship with surface temperature, of -5.09 percentage points decrease in temperature per additional native species.

Together, Chapters 2 and 4 provide evidence to support the existence of the hypothesized, but relatively untested (Cushman, 2015), relationship between increased plant species diversity and ecosystem thermal buffering (Fraser and Kay, 2004; Schneider and Kay, 1994). The studies also find that the direction of the relationship between plant species diversity and surface temperature is negative and

that this was in many cases more relevant than the relationship between surface temperature and other measured ecological attributes. And, going beyond earlier hypotheses, the evidence suggests further that surface temperature is reduced more by later succession native forest plant species diversity than by all plant species diversity or by exotic and ruderal plant species diversity. This evidence should be considered preliminary, and I argue that the connection between temperature and ‘nativeness’ of plants is indirect. It may be due to diversity of functional traits of the mostly later succession native species compared to the mostly ruderal non-native species.

### **5.1.2. Summary of ecological effects of adding topsoil, woody debris, and shade**

Beyond the importance of Chapter 3 in providing the necessary ecological data for Chapter 4, it also asked its own questions, whose answers may improve methods for rapid forest restoration. We found that the topsoil transfer increased native forest plant species diversity significantly in the recipient plots as compared to control plots in the same sites for all three sites in the study (two reforestation sites and the gravel pit site). Topsoil transfer plots in all sites did not have significantly lower native forest plant species diversity than the forest sites from which the topsoil was donated. In terms of plant species communities, the topsoil recipient plots contained species from donor and recipient sites and were therefore significantly different from both. The addition of woody debris, planting of shrubs, and erection of shade-shelters had no significant effect on plant species diversity and community in the second season after their application. Any potential effect of these treatments may take longer or affect measures other than plant species diversity and community composition (e.g. soil-chemistry, microbes, and animal habitat).

The analysis of tree species mortality, damage, and growth between 2018 and 2019 in the two reforestation sites showed no significant increase in damage or mortality from the topsoil application. There was a marginally ( $p = 0.0495$ ) significant increase in diameter growth of 23 % over 2 growing seasons for trees in topsoil recipient plots compared to those in control plots. This shows that topsoil can be safely transferred to areas where trees have already been planted, allowing for quicker canopy closure compared to methods where trees are planted after topsoil transfer.

### **5.1.3. The contribution of using large quantities of interdisciplinary and multi-platform data**

On reflection, I submit that the overall contribution of this dissertation is not just in testing hypotheses of a relationship between diversity and surface temperature, or advancing new potential applications for thermal imaging in ecosystem restoration, but also how it does so with a comparatively large amount of interdisciplinary data. It is common practice, so far, in the field of thermal imaging using UAVs to



analyze and publish data from relatively few imaging flights. Comparable example studies include Izuka et al. (2018), which used 4 imaging flights of one site over 2 days, Smigaj et al. (2019), which flew and imaged 1 site 6 times during 1 day, and Ludovisi et al. (2017) which flew and imaged 1 time over 1 site. Results from a few flights are seemingly acceptable even though weather, time of day, and time of season effects on temperature dynamics can be significant (e.g. Smigaj et al., 2019). In contrast, Chapter 4 of this dissertation presents data on 20-40 imaging flights each for three sites over 9 days. The larger amount of data analyzed here, I believe, will improve the chance of detecting real ecological change, and decrease the effect of weather variations causing patterns which are not due to actual differences in ecosystem state.

Similarly, studies of ecosystems using moderate resolution satellite thermal imaging from the last decade have often used a relatively small number of images over a short time-span from one instrument, substituting space for time. Relevant examples include Avelar et al. (2020), which used 13 images over 2 years from Landsat 8, and Vlassova et al. (2014) which used 13 images over 3 years from Landsat 5. One large review of using Landsat data for studies of ecological change (longitudinal or otherwise) only recognizes thermal imaging in one revealing sentence: “Greater use of time-series thermal data from Landsat may also provide important new insights into ecological systems, but to date, these data have been underutilized.” (Kennedy et al., 2014). In contrast, Chapter 2 analyzed a total of 53 images from 4 instruments, longitudinally over 12 years, only using space-for-time substitution where necessary in one experiment, as the equipment needed (ECOSTRESS) was not launched until 2018. The in-situ sampled ecological data used was also substantial – 59 2x2 m plots sampled every year for 12 years in Chapter 2, and 150 2x0.5 m frames sampled, and 467 trees measured in Chapter 4.

By using both space-based and UAV-borne thermal imagers, this dissertation demonstrates their relative strengths and use-cases. The medium spatial resolution (60-120 m<sup>2</sup>) space-based instruments used can cover 150-400 km in one image, and historic data going back as far as the instrument has been running are available (Hook et al., 2019; Zanter, 2019). Space-based instruments thereby allow for comparisons of large and distributed projects over time. In contrast, the UAV allows for both higher spatial resolution (<10 m<sup>2</sup>) and on-demand usage, which, depending on site size, can allow for multiple imaging flights per hour. This makes the UAV ideal for physically smaller or narrower restoration project sites, and it allows for testing and comparing the variation of thermal buffering hour-by-hour over multiple days, which satellites are unable to do.

## **5.2. Areas of future research**

The findings in this dissertation raise many new questions and avenues of inquiry. In this section, I will focus the discussion on the two lines of inquiry which I think are the most relevant in expanding both our

theoretical understanding of the connections between ecosystem change and temperature, and the practical application of that knowledge.

The first, and more theory-based line of inquiry, aims to identify which ecosystem attributes and traits matter to temperature. It is in turn divided into three parts. The first part discusses how we can utilize the concept of functional traits to test if the relationship between species diversity (particularly later succession native species) and temperature depends on specific functional traits, and if so which ones. The second part considers how we can disentangle the effect on the temperature of several different ecological attributes, including biomass and diversity, through the thermal imaging of already established and highly controlled plot-based ecosystem experiments. The third part discusses the potential for also comparing ecosystem complexity against temperature, but how this could probably not be done in controlled plot experiments.

The second, and more application-focused line of inquiry, discusses what steps need to be taken to operationalize thermal imaging as a tool for restoration managers to evaluate the benefits of, and monitor for issues in, their projects.

### **5.3. Further interrogating the diversity-temperature connection**

The studies in this dissertation found a statistical relationship between increased plant species diversity and decreased surface temperature, when controlling for biomass or cover. However, the mechanisms and interactions of these, and other ecosystem attributes (e.g. native provenance, functional diversity, and complexity) remain unknown. For one, plant species diversity can increase cover and biomass (Cardinale et al., 2007; Hooper et al., 2005), and statistically, an increase in cover and biomass can correlate with a higher chance of discovering more plant species in the sampling process. These relationships may also vary depending on climate, nutrient availability, soil-water levels, and type of plant community (Allan et al., 2013; Hooper et al., 2005).

#### **5.3.1. Functional diversity, native plant species, and surface temperature**

While plant species diversity, as used in this dissertation, quantifies the amount of taxonomic variation that exists in the ecosystem, many species can be very similar in their functions (Tilman, 2001). Functional diversity instead measures the variation of functional traits. Functional traits have been defined as ‘what organisms do, rather than their evolutionary history’ (Petchey and Gaston, 2006). These include, for example, the ability to uptake nitrogen in different forms (Mason et al., 2005), and the seasonal timing of leaf-out and flowering (Petchey and Gaston, 2006). Functional traits also commonly include what in common language could be considered structural; for example, functional traits often include plant height (e.g. Falster et al., 2018) and root depth (Zhou et al., 2020). The best way to consider these ‘structural

functions' is that they are traits inherent in the genetics of the plant and it is their interaction with the environment that begins the process of growth (function) that in turn creates the structure.

The variation in functional traits may be more important to surface temperature than taxonomic variation as it provides a measure of niche complementarity, and difference in resource use (Mason et al., 2005). For example, consider an ecosystem which has one species with the functional trait of seasonally early leaf-out and one with late leaf-out – this may allow for more transpiration, and lower temperatures over time as a larger portion of the season is utilized, than an in an ecosystem with two species that both leaf out late.

The evidence from Chapters 2 and 4 suggests that native and later succession forest plant species seem to have a stronger effect on surface temperature than non-native and ruderal ones. I have argued that this may be due both to a larger variation in functional traits of later succession species and to evolution of the functions of native species to the environmental conditions in their native range. Evolutionary changes may have allowed native plants to be more effective at absorbing and transforming incoming solar energy than exotic ones. In Chapter 2, I provide the examples that native mid-succession tallgrass prairie plants seeded in the oak woodland restoration have deeper roots than ruderal exotic plants and utilize different photosynthetic pathways (Howe, 1994; Nippert et al., 2012). These traits may aid their transpiration in the sandy soil and dry conditions of the mid-growing season. In one study which compared similar native and exotic plant species in a tallgrass prairie, the exotic species had a higher specific leaf area – i.e. thinner leaves with more surface (Smith and Knapp, 2001). This may help them take up energy and spread early in the season compared to the native plants. However, the native species in the same study had higher total leaf area, and up to three times as much total biomass by the end of the season (Smith and Knapp, 2001), indicating that their architecture may have allowed for more water retention and more effective energy transformation through the mid and late season, potentially allowing for more cooling.

This is only a preliminary hypothesis so far, but it could be tested further by exploring functional trait diversity. To test the effect of functional trait diversity on surface temperature, a future study could use functional trait categories from the literature and apply them to the species data already assembled in this dissertation. The functional trait diversity could then be compared with the temperature measurements. The challenge is figuring out and selecting the right functional traits to focus on. There are multiple large function and trait databases available. For example, the TRY Plant Trait Database, one of the largest databases of plant traits includes information on 2100 traits of 160,000 different plant taxa (Kattge et al., 2020). With this amount of functional traits available to choose from, it would be unwise to try to conduct analysis by throwing all traits or functions into statistical analysis against temperature, as this would undoubtedly yield falsely positive (Type-1 error) results due to the large number of

comparisons made. Instead, any study on functional diversity and surface temperature must be firmly based in theory.

There is at least one study of functional traits and ecosystem thermal buffering, wherein plant species were given traits as per Grime's CSR (Competitive, Stress-tolerant, Ruderal) model (Norris et al., 2012). A significant thermal buffering effect was found for systems with a larger portion of species considered 'competitive' or 'stress-tolerant'. There was no further breakdown of traits, and sites were only compared over space, for one season using ground-based surface temperature data-loggers.

Moving beyond the CSR model, and basing our selection of functional traits for diversity analysis in ecosystem energetics and thermodynamic theory, the traits affecting transpiration may be most important, as transpiration transforms the largest amount of energy in the ecosystem (Baldocchi, 2005; Gosz et al., 1978; Monteith, 1981). Focusing on transpiration, some of the traits to be included would be rooting depth, stomatal conductance, and xylem flow. Other traits to be selected may have to do with respiration, productivity, and the albedo and emissivity of different plants. A major gap is that the 2,100 functional traits in the TRY database do not include albedo or emissivity values. These traits would instead have been sourced from elsewhere, including the ECOSTRESS spectral library (Meerdink et al., 2019), which holds reflectivity and emissivity values for 61 common tree and shrub genera.

Statistical analysis to compare functional diversity and surface temperature can shed light on what type of functional traits matters to temperature change, and if native plant species show more functional diversity than exotic ones. However, many of these functional traits will also interact in complex ways, affecting both diversity and biomass in ways that are hard to untangle by statistical methods alone.

### **5.3.2. Controlled experiment to separate the effects of different attributes and functions**

Acquiring UAV thermal imaging of a more controlled ecological experiment, where several ecosystem attributes (e.g. biomass, cover, height) are physically controlled for, while varying the attribute of concern (e.g. species or functional diversity), could potentially isolate these ecological factors' separate effects on surface temperature. There are many long-term ecological experiments where this could be done. Some examples include the JENA grassland experiment in Germany (Weisser et al., 2017), and the Sabah Biodiversity Experiment of tropical forests on Borneo, Malaysia (Hector et al., 2011). In Canada, three long term plot experiments of forest restoration exist as part of The International Diversity Experiment Network with Trees (IDENT) (Tobner et al., 2014). One IDENT site, located in Sault Ste. Marie, Ontario, contains 408 plots, with 4 levels of species diversity, 3 levels of functional diversity, and plots with either angiosperm or gymnosperm trees. The same site also has controlled variations in water-availability. Thermally imaging this site, and other ecosystem diversity experiments throughout the world, frequently

(hourly) throughout multiple day-night cycles and at different seasons, could go a long way towards answering several questions about the relationship between species and functional diversity and temperature. Including:

1. Is the relationship between species diversity and surface temperature linear or non-linear? And is this relationship different for different combinations of species?
2. What are the limits of the relationship between surface temperature and species or functional diversity? – I.e. when is the ecosystem ‘saturated’ in terms of the possible effects of diversity?
3. How strong is the relative effect of cover and biomass as compared to the effect of species or functional diversity, and does it change with time, age, or size of trees?
4. How generalizable is the diversity-temperature relationship between different tree species communities? Do certain combinations of trees cool more than others?
5. Which combination of functions have the strongest effect on surface temperature; and how do different functional diversity compare in terms of their cooling effect?
6. How do these effects of species and functional diversity differ during the day and season? Do more diverse tree assemblages also increase the nighttime temperature more than less diverse ones with the same biomass and cover?
7. With varying water levels – how important is transpiration for cooling relative to all other factors (e.g. thermal inertia, respiration, albedo, etc.), and how does this vary between different tree assemblages?

### **5.3.3. Limits of controlled experiments – ecosystem complexity and temperature**

Thermally imaging controlled plot experiments, such as the IDENT sites, do not preclude the need to also continue to thermally image natural ecosystems, as plants in controlled plots may behave differently from those in complex natural ecosystems (Wardle, 2016). Experimental plots are generally small – the Sault Ste Marie ones are 10 m<sup>2</sup> – and are therefore potentially affected by wind and other edge effects. The experiments also only vary the number of trees, with no shrubs or herbaceous plants included, and they exclude large herbivores through fencing. Landscape context (Zirbel et al., 2019) and the nutrient cycling done by the many species of animals, fungi, and microbial decomposers (Loreau, 1995) can be significant for the impact of diversity in ecosystem functions, and therefore temperature (Schneider and Kay, 1994). The interactions and connections between the different species and their environment over space and time is what creates ecological complexity (Cadenasso et al., 2006; Kay and Schneider, 1994; Norris et al., 2012; Parrott, 2010; Riede et al., 2010; Vihervaara et al., 2019).

Authors have argued from a thermodynamic basis, that increased complexity should decrease surface temperature (Fraser and Kay, 2004; Kay, 2000; Schneider and Kay, 1994). The argument is based on the idea that more complexity would provide more pathways for energy transformation and storage. Measuring complexity in ecosystems has proven difficult, in part because there are many kinds of complexity, including trophic (e.g. Riede et al., 2010), functional (e.g. Cadenasso et al., 2006), structural (e.g. Parrott, 2010), and informational (e.g. Brunetti et al., 2018; Vihervaara et al., 2019). However, methods of measuring expressions of complexity are emerging, including those which can be done through remote sensing – for example by looking at heterogeneity in structure or greenness of ecosystems (Parrott, 2010). One study (Avelar et al., 2020) has so far compared this type of remotely sensed complexity measure against thermal imaging, but only through comparing different sites at the same point in time. A possible next step would be to conduct a more longitudinal study of in-situ measured diversity and spatio-temporal complexity, measured remotely or in-situ, against surface temperature.

In summary, I would argue that the path forward is to look for generalizable patterns by thermally imaging both plot experiments and natural ecosystems in as many sites and ecosystem types as possible, utilizing the strengths of each: the ease of measurements and ability to physically control certain factors in plot experiments, and the more realistic, but harder to measure, conditions in natural ecosystems.

## **5.4. Operationalizing thermal imaging for restoration and conservation managers**

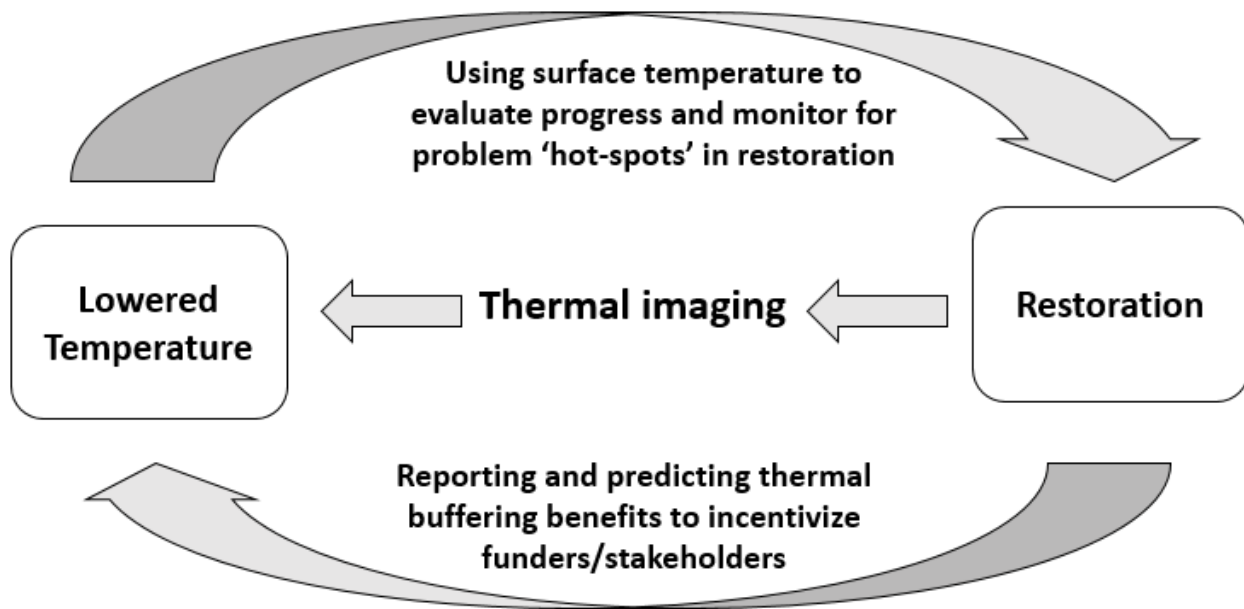
### **5.4.1. Accounting for the benefits of cooling provided by restoration**

Showing the benefits of earlier restoration, and predicting the benefits of future restoration, in terms of the local cooling effects can be a potent tool for increasing popular support and funding for new restoration projects (Figure 5.1). Most people have an intuitive understanding of temperature and have experienced the sensation of being uncomfortably hot.

Considering the continued global warming effect of climate change causing more extreme heat events (Mann et al., 2017; Stott, 2016), the idea of adaptation by reduction of local temperatures will only become more important. For the region of Southern Ontario, Canada where this study was conducted, it is estimated that mean summer temperature will increase by 2.1 °C, and the number of days a year with above 30 °C weather will increase from 5 to 16, between the periods 1971-2005 and 2021-2050 (Prairie Climate Centre, 2018). In this context, showing that restoration of gravel pits and agricultural fields to wooded ecosystems in or near a village, town, or city can reduce local afternoon summer surface temperature significantly, can be a strong motivator for funding and action at national, provincial, and local levels.

Increasing the use of temperature benefit accounting requires outreach to restoration practitioners, developing more intuitive tools for historical evaluations, and gathering more data for predictive modeling. Through presenting the research in this dissertation to practitioners, I have managed to reach out to multiple conservation authorities, which manage the conservation and restoration of lands in Ontario. There is a definite interest, but often a lack of funding and knowledge. After one presentation, managers at Credit Valley Conservation Authority (Peel Region, Ontario), conducted a pilot study of the benefits of their plantings, finding a temperature decrease of 1-3 C° for agricultural fields turned into coniferous plantations (Schuurman, 2020). In aiding this pilot study, I found that the major hurdles for practitioners were accessing, processing, and analyzing historical thermal imaging satellite data. NASA Earthdata, USGS EarthExplorer, and Google Earth Engine provide platforms for accessing and downloading thermal data, but part of the processing and the analysis and comparisons of temperatures over time still needs to be done manually through GIS or programming software (e.g. ArcGIS, QGIS, and R). A next step in applied research would be to build a dedicated platform where thermal images from different years, of the same restoration site, could be normalized, adjusted, and analyzed in a way that is easily understandable and accessible to practitioners while remaining accurate.

This kind of tool would still only work for evaluating historic restoration sites, as temperature change takes time. To instead model and predict temperature change into the future, more data is needed. A large scale campaign of collecting thermal imaging data of well-documented and monitored historical restoration projects could provide a basis for a model that can predict the amount of cooling that can be expected in a certain amount of time, depending on starting landscape, climate zone, method of restoration and goal ecosystem.



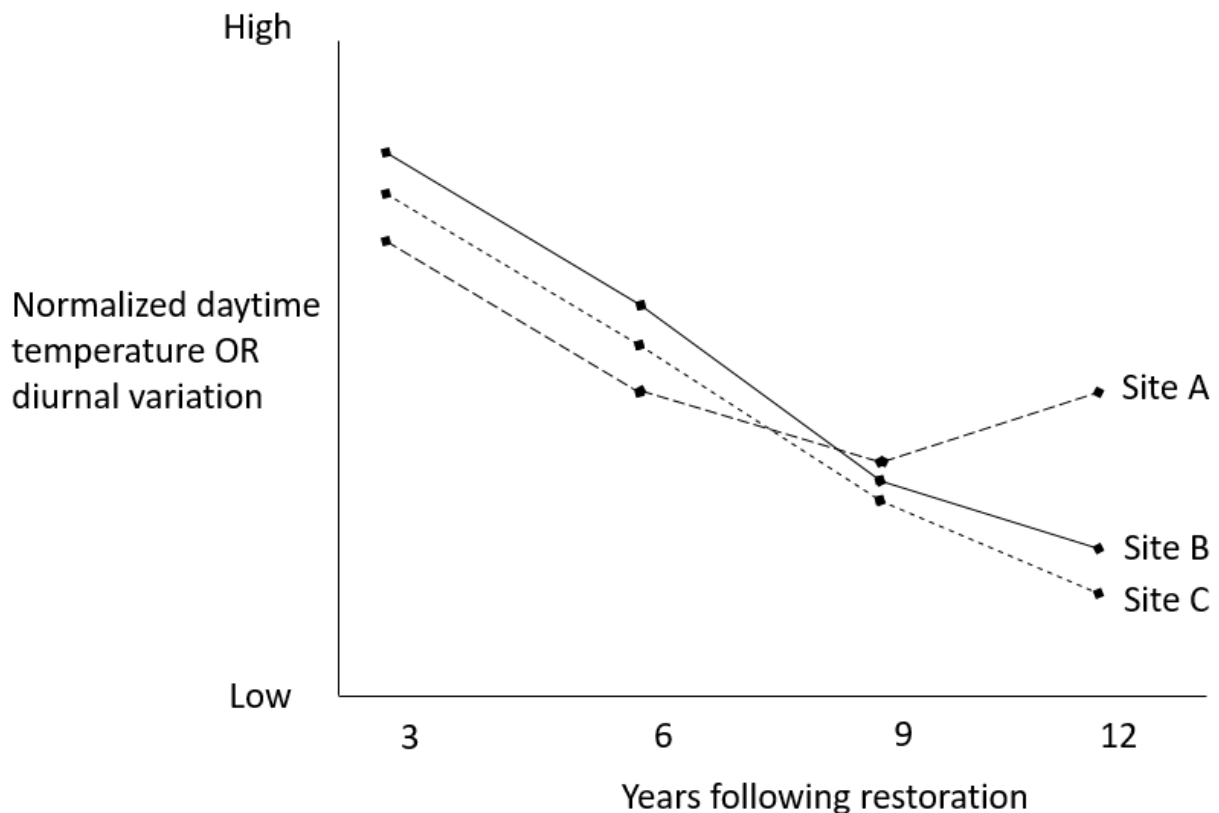
**Figure 5.1: Thermal imaging of ecosystems undergoing restoration can be useful to quantify the thermal buffering benefit of restoration (section 5.4.1), and also as a way to evaluate restoration progress and monitor for issues (section 5.4.2).**

#### **5.4.2. Thermal imaging for monitoring ‘hot-spots’ in restoration projects**

Beyond measuring and predicting benefits, a thermal mapping tool can also be used for identifying literal hot-spot areas, indicating potential issues that may affect the trajectory of restoration projects (Figure 5.1). As shown in figure 2.6, maps can be used to find areas where the temperature has not been decreasing, as compared to other areas around it. If satellite data are used, then the issue hot-spot can be detected down to an area the size of the pixel resolution – 60 to 100 m<sup>2</sup>. This may seem like a large area, but considering projects with sites that can be thousands of hectare in size (e.g. Cao et al., 2011; Melo et al., 2013), and which may also be remote, difficult to access, and difficult to traverse on foot, narrowing down problems to a hectare or less can help guide and speed up in-situ monitoring and adaptive management greatly.

In case resources limit the number of restoration project sites that can be in-situ monitored in a year, as was the case for the study in Chapter 2, then what I have termed the ‘mean thermal recovery trajectory’ of each site (Figure 5.2.) can be used to triage which sites need attention first. This would be done by comparing the normalized mean summertime daytime temperature (similar to figure 2.5 but without linear regression), or the mean diurnal temperature variation (similar to figure 2.7 but without linear regression) over time and against the other project sites. Normalization of temperatures would need to be conducted against a mature and stable forest area, as was done in Chapter 2.





**Figure 5.2: Illustration of the relative thermal recovery trajectory of three sites in an imagined restoration project. All sites are experiencing a downward trend in normalized summer daytime temperature or diurnal temperature variation, until year 9. After year 9, site A deviates from the other sites, increasing in temperature or its diurnal variation, indicating an ecological issue at the site, moving it up on the triage list of fields that require further in-situ monitoring and adaptive management.**

Temperature change maps and trajectories could be monitored manually, but they could also potentially be automated to look for hot-spots and deviations that stand out against their surroundings, other fields, or over time. A future collaborative project would be to use historical data to train a thermal imaging mapping tool to spot actual problem areas, such as disease outbreaks, droughts, or encroachment, as compared to natural variations in temperature.

If sites are smaller, or higher spatial and temporal resolution monitoring is needed in order to be effective, then UAV- mounted thermal imaging could be used by practitioners. This method will be more expensive than publicly available satellite imagery, and there will be no historical comparisons to be made, thermal imagery could potentially guide personnel on the ground to find out what the issue is that is affecting the temperature of a single tree or colony of plants. To operationalize UAV thermal imaging for restoration projects cheaper and more accurate methods of stitching thermal imaging are needed, along with standardized protocols for when (weather and time during the day and season) and how to accurately gather and compare the data. At the moment stitching together thermal imagery from UAV is difficult due

to the smaller variation in each image, as compared to visual imagery, making it harder for software to find overall. Another option would be to fly at higher altitudes (e.g. 200-500 m), where resolution would suffer but still be good enough to find single trees or 1 m<sup>2</sup> spots of stressed vegetation within unstitched imagery that could cover over a hectare in one image. UAV thermal imaging monitoring may sound expensive, but depending on the size and accessibility of a site, it may still be cheaper than hiring ground-personnel to wander the field without direction. UAV monitoring would also be less disruptive, as it would avoid trampling or spread of invasive species from pants, boots, or land-vehicles. To provide this type of monitoring as an attractive method to practitioners, further studies are needed of the cost-effectiveness of monitoring by UAV thermal imaging, comparing it both to alternative monitoring methods, but also to the opportunity cost of not monitoring at all, and thereby increasing the risk of failure.

## **5.5. Final thoughts**

The practice of ecosystem restoration is already expanding rapidly in terms of numbers and size of projects, including mega-reforestation projects spanning millions of hectares in China (e.g. Cao et al., 2011), Brazil (e.g. Melo et al., 2013), and south of the Sahara (e.g. O'Connor and Ford, 2014). And, if intentions of the Bonn Challenge (Stanturf et al., 2019; Suding et al., 2015) and the UN Decade of Ecosystem Restoration (Aronson et al., 2020; Young and Schwartz, 2019), to restore 350 ha of deforested and degraded land by 2030 are realized, the pace of expansion of restoration will only increase.

Unfortunately, many restoration projects fail, both due to lack of knowledge and because the existing knowledge may not flow between practitioners and scientists (Cooke et al., 2019; Menz et al., 2013; Suding et al., 2015; Suding, 2011). The larger the projects become, the higher the stakes and the amount of resources potentially squandered in case of a failure (Cooke et al., 2019; De Groot et al., 2013). It is within this context that the search for new and improved methods for implementing, evaluating, and monitoring restoration, including thermal imaging, is urgently needed. It is not efficient or plausible for field ecologists to become thermal remote sensing specialists, or vice versa. Instead, the way forward should be through collaboration between fields of study and usage of the open data that is already available in order to test our assumptions, and to create reliable tools in a format that can be rapidly disseminated to practitioners who need them as aids in restoring a more sustainable world.

Much of the data analyzed in this dissertation would not have been possible to acquire until the last few years. For example, Chapter 2 was published as one of the first studies using the new ECOSTRESS equipment on the International Space Station which was made operational in July 2018. The launch of ECOSTRESS enabled the Chapter 2 study to possibly be the first to use nighttime thermal imaging to image ecosystems from space with a < 1 km<sup>2</sup> resolution. Beyond ECOSTRESS, the type of

high-resolution uncooled microbolometer thermal imager used in the UAV has only recently become affordable enough for research (Sagan et al., 2019). The time-series of thermal images from Landsat satellites grows each year, allowing for longer retrospective studies of restoration projects. The possibilities for thermal imaging of ecosystems will only grow in the near future with UAVs and their thermal imagers becoming cheaper and more accessible and with the planned commissioning of multiple new moderate spatial resolution thermal imaging satellites, including the NASA-led SBG Observable (Poulter et al., 2019) and the French-Indian TRISHNA (Lagouarde et al., 2018). I look forward to being part of growing the collaboration, combination, and understanding of new thermal imaging tools, and the usage of their output to aid in the future success of ecosystem restoration globally.

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## Appendix 2A: Details of processing and adjustment of Landsat thermal imagery

The Digital Number to Land Surface Temperature (DNtoLST) model clips the images to the area of interest, removes pixels missing values due to the failed Scan Line Corrector on Landsat 7 and removes pixels within 200 m from any clouds, based on the cloud cover layer supplied with each image, so as to avoid effects of cloud shadow on temperature. The quantified and calibrated digital number ( $Q_{cal}$ ) of the thermal band was translated into top-of-atmosphere radiance ( $L_a$ ) (Jiménez-Muñoz et al., 2009; Sobrino et al., 2004; Yu et al., 2014). The DNtoLST model then estimates land surface temperature ( $T_s$ ) from top-of-atmosphere radiance using the single channel method from Jiménez-Muñoz and Sobrino (2003) as revised by Jiménez-Muñoz et al. (2009). The single-channel method is an approximation of the radiative transfer equation and includes corrections for atmospheric conditions in atmospheric transmission and upwelling and downwelling atmospheric radiance. These factors were estimated for the research area using the Atmospheric Correction Parameter Calculator (ACPC) (Barsi et al., 2005).

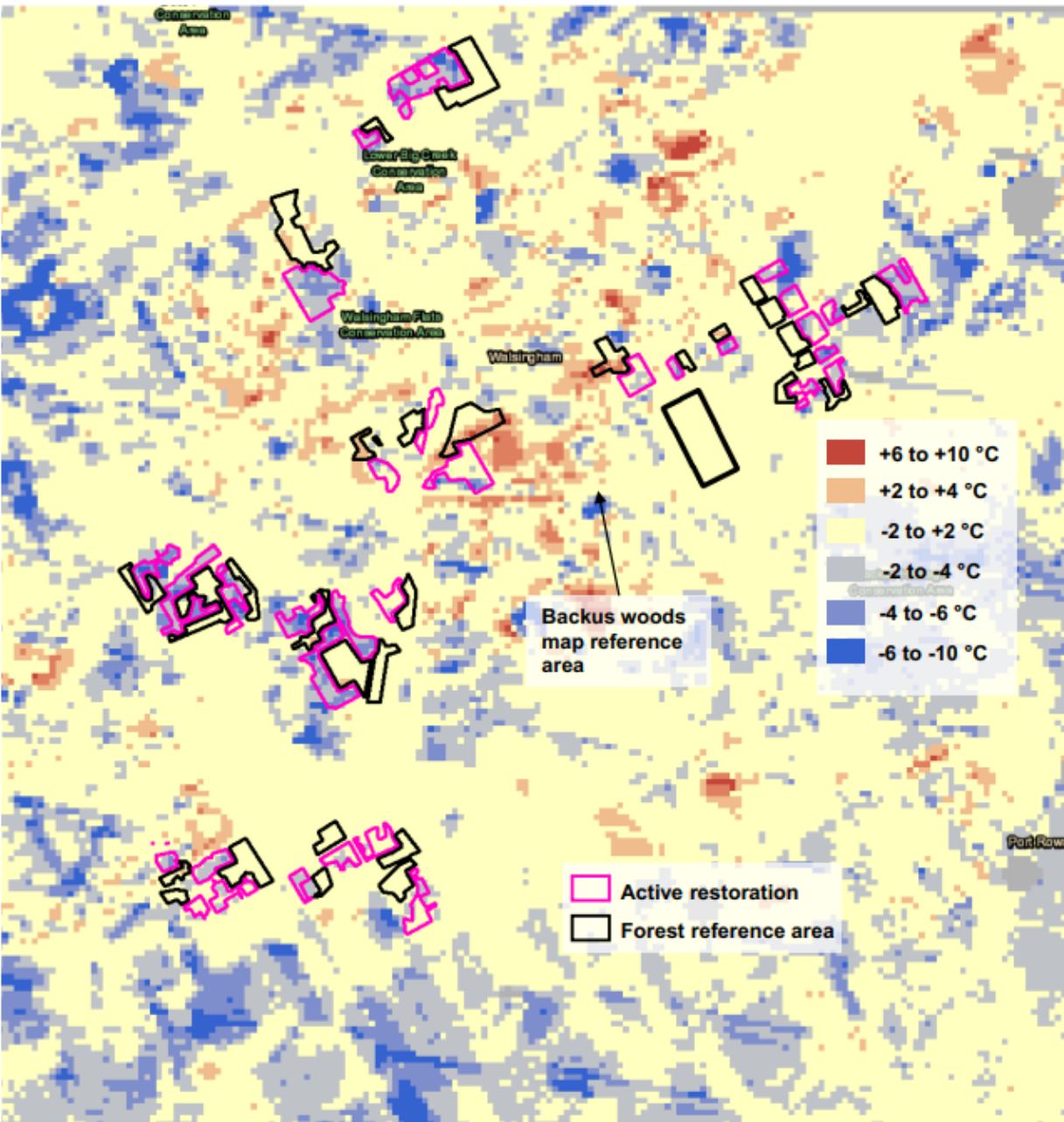
The emissivity term in the single-channel method equation was estimated using the NDVI Threshold Method (NDVI<sup>thm</sup>) (Sobrino et al., 2008; Yu et al., 2014). The criteria for the use of the NDVI<sup>thm</sup>, which this study fulfilled, were that only images in the 10-12  $\mu\text{m}$  wavelengths were used and that the site only contains soil and vegetation. In the NDVI<sup>thm</sup>, emissivity is based on proportion vegetated area, which in turn is based on NDVI. Based on NDVI results from areas of the study site known to be bare soil and closed canopy, NDVI values below 0.4 were assumed to be bare soil and values above 0.7 were assumed to be fully vegetated. While 0.4 is a higher threshold than the recommended standard, the literature recommends that when possible, NDVI threshold should be set to real site conditions (Jiménez-Muñoz et al., 2009; Sobrino et al., 2008, 2004; Van de Griend and Owe, 1993).

Emissivity values for bare soil and vegetation for the different effective wavelengths were obtained from literature (Table 1, Sobrino et al., 2008; Yu et al., 2014). Due to similar effective wavelength and same bandwidth breadth, the Landsat 5 formula for estimating emissivity was also used for Landsat 7 imagery. Soil emissivity values were validated for local conditions against the ECOSTRESS (formerly ASTER) spectral library (Baldrige et al., 2009; Meerdink et al., 2019). The soil throughout the site is classified as Plainfield series fine, very well-drained, brown Aeolian sands in the Udipsamments (Nature Conservancy of Canada, 2006). This is an Entisol in the Psamments sub-order. Soil emissivity was then taken from the closest related soil in the ECOSTRESS library, which is Quartzipsamment, brown to dark brown Aeolian sand, in the same sub-order (Table).

**Table 1: Comparison of Landsat 5, 7 and 8 bandwidth, wavelength and the recommended emissivities for soil and vegetation according to the ECOSTRESS spectral library, Sobrino et al. 2008 and Yu et al. 2014 and Snyder et al 1998, showing general agreement on soil and vegetation emissivity.**

Landsat satellite #	Bandwidth ( $\mu\text{m}$ )	Effective wave-length	ECOSTRESS Soil emissivity	Soil emiss. from literature	Veg. emiss. from literature	Bare soil empirically derived equation
5	10.40 - 12.50	11.457	0.972	0.973	0.985	$0.979-0.035(\rho_{\text{RED}})$
7	10.40 - 12.50	11.269	0.972	0.973	0.985	$0.979-0.035(\rho_{\text{RED}})$
8 <sub>10</sub>	10.60 - 11.19	10.904	0.966	0.967	0.986	$0.973-0.047(\rho_{\text{RED}})$

## Appendix 2B: Full map of temperature change 2014-2018



Full map of restoration area temperature change for NCC sites in Norfolk County, 2014-2018

## **Appendix 3A: Details on methods of transfer of topsoil, plot selection criteria, and vegetation sampling**

### **1. Further details of transfer of topsoil**

Topsoil and overlaying matter were dug up by mini-excavators ( Takeuchi TB-138 and TB285, Takeuchi Mfg. Co., Ltd.) and moved directly to receiving sites using wheel-loaders (Takeuchi TW80), skid-steers with tracks (John Deere 270, Deere & Company; Case 95XT, Case Construction), dump trucks (Ford F800, Ford Motor Company) and a tractor (John Deere 2555). The topsoil was then loose-tipped in the experimental plots, and subsequently spread evenly by another mini-excavator. Trees and shrubs that were moved with the topsoil were stood back up and planted in plots where possible.

### **2. Plot selection methods and criteria**

#### **2.1. Reforestation site plot selection methods and criteria**

For the 2015 reforestation site 20 12.5 x 10 m plots plot locations were selected semi-randomly. Each potential plot-location was between 50 and 100 paces from each other, avoiding windrows and with minimum 10 m from forest edge, while taking into consideration equipment accessibility (Appendix D). Out of the 20 plot locations, 10 were randomly selected to be mowed between tree-rows and then sprayed with a glyphosate herbicide in fall 2017, 1-2 weeks before topsoil transfer took place in order to suppress ruderal field species. Out of the 10 mowed and sprayed plot locations, 5 were selected to receive forest topsoil. Selection of which of the mowed and sprayed plots should receive topsoil had to be done non-randomly and based on which plots were accessible for the soil-moving machinery due to a combination of mud and uneven terrain at the time of topsoil transfer. The 5 plot locations which had been mowed and sprayed but did not receive topsoil retained in the experiment as ‘mow-and-spray’ control plots and the 5 of the 10 plot locations not mowed and sprayed were retained as passive control plots.

For the 2016 reforestation site, 5 plot locations were randomly selected to be mowed and sprayed and then receive topsoil, using the same constraints as for the 2015 reforestation site. Another 5 plot locations were selected to be passive control plots. In order to maximize distance between plots with the same treatment, the locations of passive control plots were positioned in between the topsoil transfer plots but with the same constraints in terms of minimum distance to windrows and forest edge.

**2.1.1. Detailed plot selection criteria for reforestation sites (referred to as YA or S2) as described through email verbatim by post-doctoral fellow Paul Richardson 14th July 2020.**

“At YA (or S2) the approach was pretty much the same as for the other described sites, except that there was no constraint with respect to only selecting areas on a woody successional trajectory -- anywhere on the property was fair game, except for the windrows that had previously been applied with woody debris and other materials as part of Stantec's afforestation plan. The biggest constraint was accessibility for equipment, so I used random paces and headings (as described) to guide my path in from either outer perimeter points or the main trail. I wanted to avoid having all plots too near the field margins, however, or each other, and wanted to make use of the space at hand. Therefore, the range of paces selected from was 50-100 in the 2015-planted field (at plantions S3-S4, in contrast, space was much less available and paces reduced to 10-30). The plots in the 2016-planted field of YA had somewhat less space and paces were 20-40, I believe. In addition to the random paces, recall that the five plots in the 2015-planted field were randomly selected from the full set of 20 potential translocation-recipient plots that we assessed in 2017 (the 5 plots that received habit amendment treatments but not translocated living mulch were also selected at random from this pool).”

## **2.2. Gravel pit site plot selection methods and criteria**

Plot locations in the gravel pit site were selected semi-randomly. A random walk of 20-60 paces was taken from an arbitrary start position, and then from the previous plot from each other. At the end of the random walk, a plot outline was selected nearby to represent formerly mined conditions with a limited amount of spontaneous succession. This included having a minimum of 3 woody stems of at least 1 m height inside of it and be considered as not ‘too wet, too sparse or too harsh’ as compared to other potential plots in the same site (Paul Richardson - personal communication, 13/7/2020. Full methods of selection in Appendix D). A quarter-plot sized area, adjacent to the topsoil recipient plot was selected as a passive control. The exact location and configuration of the passive control quarter-plot was based on being similar in terms of vegetation cover, woody stems and vehicle tracks to that of its adjacent topsoil recipient plot (Figure 2). No mowing or spraying was considered necessary at the gravel pit site.

### **2.2.1. Detailed plot selection criteria for reforestation sites (referred to as S1) as described through email verbatim by post-doctoral fellow Paul Richardson 13th July 2020.**

“In S1, the constraints on block location included minimizing likelihood of potential influence from: i) other experimental blocks; ii) the active industrial usage of the property; and iii) margins of the property that were never mined for gravel and currently feature forest cover. More importantly, the block locations at S1 were explicitly selected to represent conditions at the mined site consistent with spontaneous natural succession towards woodland cover: if substrates at a prospective experimental block location were too sparse, harsh, wet etc. to support spontaneous woody growth, that location was not included in the final

selection (in practice, I would veto a location if fewer than 3 woody stems greater than 1 m height were spontaneously growing in the prospective 10 m x 12.5 m area). Meeting these constraints guided the "haphazard" factor in the selection; the "random" factor was facilitated using a table of random numbers to dictate number of paces separating blocks from each other, as well as the distance between an arbitrary starting point (approximate NW corner of site area considered) and the first block. The list of random numbers dictating paces taken ranged from 20 to 60, while heading taken was partially guided by random selection between 1 and 3 (veer left, go straight, veer right) but also partially guided by locations of major abiotic site constraints (i.e. avoid big bare patches with no soil and no trees; avoid big wet patch with no trees; avoid property margin; avoid other blocks, etc.). Once I paced appropriately to the designated location, it was evaluated as a potential central point for an experimental block. If vetoed (due to any of the reasons discussed above), a new random number of paces at semi-random heading was selected and taken.”

### **2.3. Protected forest site plot selection methods and criteria**

#### **2.3.1. Detailed plot selection criteria for protected forest site (referred to as S5) as described through email verbatim by post-doctoral fellow Paul Richardson 13th July 2020.**

“I used a similar set of guidelines to select the blocks at S5, though the main environmental factor I was seeking to represent at the locations was forest succession outcome equivalent to that observed in the donor forests -- i.e. woodland patches with the same approximate canopy species composition and size structure, as well as similar soils, moisture, litter layers and understory composition to the donor forest. Major constraints on location selection included: i) staying as far back as possible from the section of forest expected to be removed as the quarry expands over the next few years (i.e. southern limit); ii) remaining as close as possible to the donor forest, both to ensure continuity and representation of the donor forest "type" within the S5 blocks and to minimize distance and obstacles to equipment during the bulk materials relocation (i.e. western limit); avoiding the transition from upland sugar maple forest (consistent with the donor forest) to more lowland area with more persistently-wet soils and shorter/younger canopy that trends away from sugar maple to include much more ash and silver maple (i.e. northern limit). Within the delimited "safe" area to consider establishing experimental LM translocation blocks, I started at an arbitrary point near NW corner and used random paces/headings table similarly to the approach used at S1 to select potential block centres, vetoing and reselecting as required (e.g. first block selected was too wet, so moved).”

### **3. Summer vegetation sampling details**

All sub-plot treatments in all plots in all sites in the experiment were sampled using two 2 x 0.5 m quadrats made of thick metal wire, following a similar sampling design to Tomlinson et al. (2008). The 2 x 0.5 m quadrat was sub-divided by thin metal wire into 16 squares measuring 25 cm<sup>2</sup> each. The frequency of occurrence of each species was estimated from a presence/absence count in an imagined 12 cm<sup>2</sup> quarter of each of the 25 cm<sup>2</sup> squares.

Species were considered present if the plant was alive and any part of it was within the 12 cm<sup>2</sup> quarter-square when looking straight down at the square at chest height, even if the plant was rooted outside of it. This count provided a presence/absence number between 0 and 64 for each species in each frame. These numbers were then converted to proportional frequency ratio (0 to 1) relative to all other species for diversity measurements. While the method was used to approximate abundance, or percentage cover, of plants, it was still based on presence/absence of species in very small (12.5x12.5 cm) squares. For this reason, this study uses the term ‘proportional frequency’ throughout.

Two quadrats were laid in each subplot-treatment in each plot in all areas, with one quadrat approximately 2.7 m from the plot edge and the other 2.7 m from the plot center (Figure 3). For the SS and DS treatments, the quadrat’s distance from the edge and center was adjusted so that the sampling frames were positioned under the actual treatments.

For the fall follow-up sampling, if a plant was identified to species that had only been identified to genus or family, it was updated in the list from the mid-summer sampling, but relative cover of that plant species was not changed. If a genus had been recorded and it turned out it was more than one species, the frequency from summer was divided based on the relative frequency in the fall.



## **Appendix 3B: Tests, diagnostic plots and assumption testing**

This appendix is a PDF file of R code output of statistical tests, diagnostic plots and assumption testing for all analysis in Chapter 3.

The file name of this PDF file is “Appendix 3B - Tests, diagnostic plots and assumption testing.pdf”.

If you accessed this thesis from a source other than the University of Waterloo, you may not have access to this file. You may access it by searching for this thesis on <https://uwspace.uwaterloo.ca/UWSpace>.

## **Appendix 3C: Species list and observations**

This appendix is a spreadsheet containing species list and all sampling observations of plants referred to in Chapter 3.

The spreadsheet data can be accessed through Dataverse at: <https://doi.org/10.5683/SP2/M4RTY0> in the dataset titled “Spreadsheet appendices (3C and 4B) for L. Jonas Hamberg dissertation”

## Appendix 3D: Further comparisons of plant community composition

### 1. Paired comparisons of plant community between plot-types and sites

#### 1.1. Spring pairwise PERMANOVA comparison between plot-types

**Table I:** F-statistic, R-square and adjusted p-value for post-hoc PERMANOVA on the Jaccard beta diversity of pairs of sites and plot-types. P-values were adjusted using the Benjamini-Hochberg procedure. P-values are approximate as they depend on the random start and number of combinations used. 100000 permutations used for each pairwise comparison. No measures from sub-plots with additional treatments beyond receiving topsoil were used. RF Control – all passive and mow-and-spray controls in both reforestation sites; RF Topsoil recipient – all topsoil recipient plots in both reforestation sites; , Protected Forest – all passive and ‘lift-and-drop’ plots in the protected forest site.

Paired plot-types :	F	R <sup>2</sup>	p	P-adjusted (BH)
RF Control (all) : Protected Forest (all)	20.1	0.48	<0.001	<0.001
RF Control (all) : RF Topsoil recipient (all)	12.2	0.36	<0.001	<0.001
RF Control (all) : Gravel Pit - Topsoil recipient	9.6	0.36	<0.001	<0.001
Protected Forest (all) : RF Topsoil recipient (all)	11.5	0.39	<0.001	<0.001
Protected Forest (all) : Gravel Pit - Topsoil recipient	6.4	0.33	<0.001	<0.001
RF Topsoil recipient (all) : Gravel Pit - Topsoil recipient	4.3	0.25	<0.001	<0.001

#### 1.2. Summer pairwise PERMANOVA comparison between plot types

**Table II:** Pairwise comparison of topsoil recipient plots by sites tested against control sites in the same site and against the relevant donor forest site. Test performed through pairwise PERMANOVA on dissimilarity index, with p-values adjusted using the Benjamini-Hochberg procedure.

Paired plot-types	F	R <sup>2</sup>	p	p-adjusted
RF 2015 - Topsoil recipient: RF 2015 - Mow-and-spray	4.2	0.34	0.0080	0.0096
RF 2015 - Topsoil recipient: RF 2015 - Passive	4.5	0.36	0.0080	0.0096
RF 2015 - Topsoil recipient: N. Donor Forest - Passive	11.1	0.58	0.0081	0.0096
RF 2016 - Topsoil recipient: RF 2016 - Passive	4.8	0.37	0.0082	0.0096
RF 2016 - Topsoil recipient: N. Donor Forest - Passive	7.9	0.50	0.0082	0.0096
Gravel Pit - Topsoil recipient: Gravel Pit - Passive Control	2.4	0.23	0.0244	0.0244
Gravel Pit - Topsoil recipient: S. Donor Forest - Passive	3.0	0.27	0.0075	0.0096

### 2. Comparisons of plant communities in sub-plot treatments

## 2.1. Spring NMDS and PERMANOVA for sub-plot treatments

A separate NMDS was performed to test if there were any differences in Jaccard beta diversity for the sub-plot treatments within any of the topsoil recipient plots in the 2015 and 2016 reforestation sites combined, as well as the gravel pit site (Figure 5). There is no apparent visual grouping of sub-plot treatments within either the combined reforestation sites, or the gravel pit as per the NMDS in Figure 5, and separate PERMANOVA tests of sub-plot treatments found no significant difference within either the combined reforestation sites ( $F = 0.9$   $R^2 = 0.07$   $p = 0.676$ ) or the gravel pit site ( $F = 0.8$   $R^2 = 0.12$   $p = 0.821$ ). Multivariate dispersion was similar in both the combined reforestation site ( $p = 0.554$ ) and the gravel pit site ( $p = 0.931$ ). In both tests multivariate dispersion was found to be similar with the ANOVA on dispersion for reforestation sites.

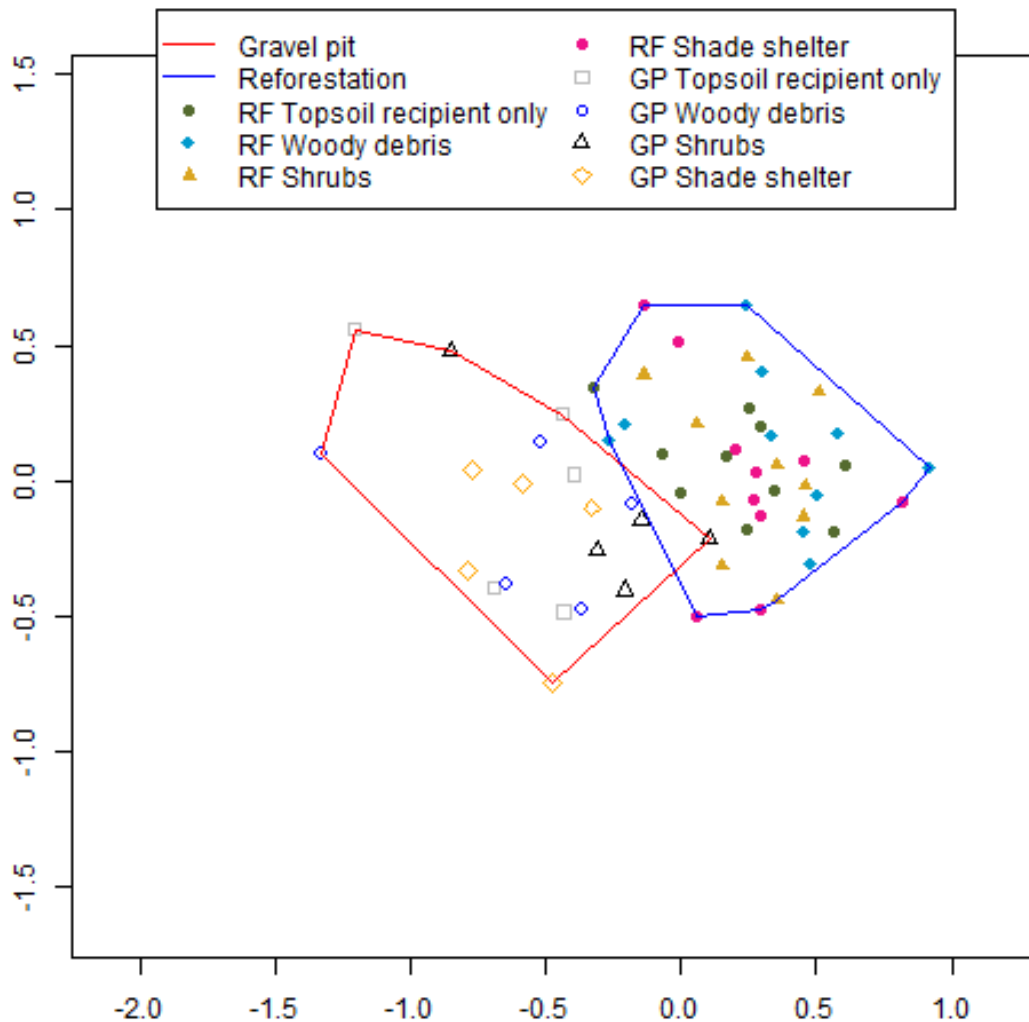
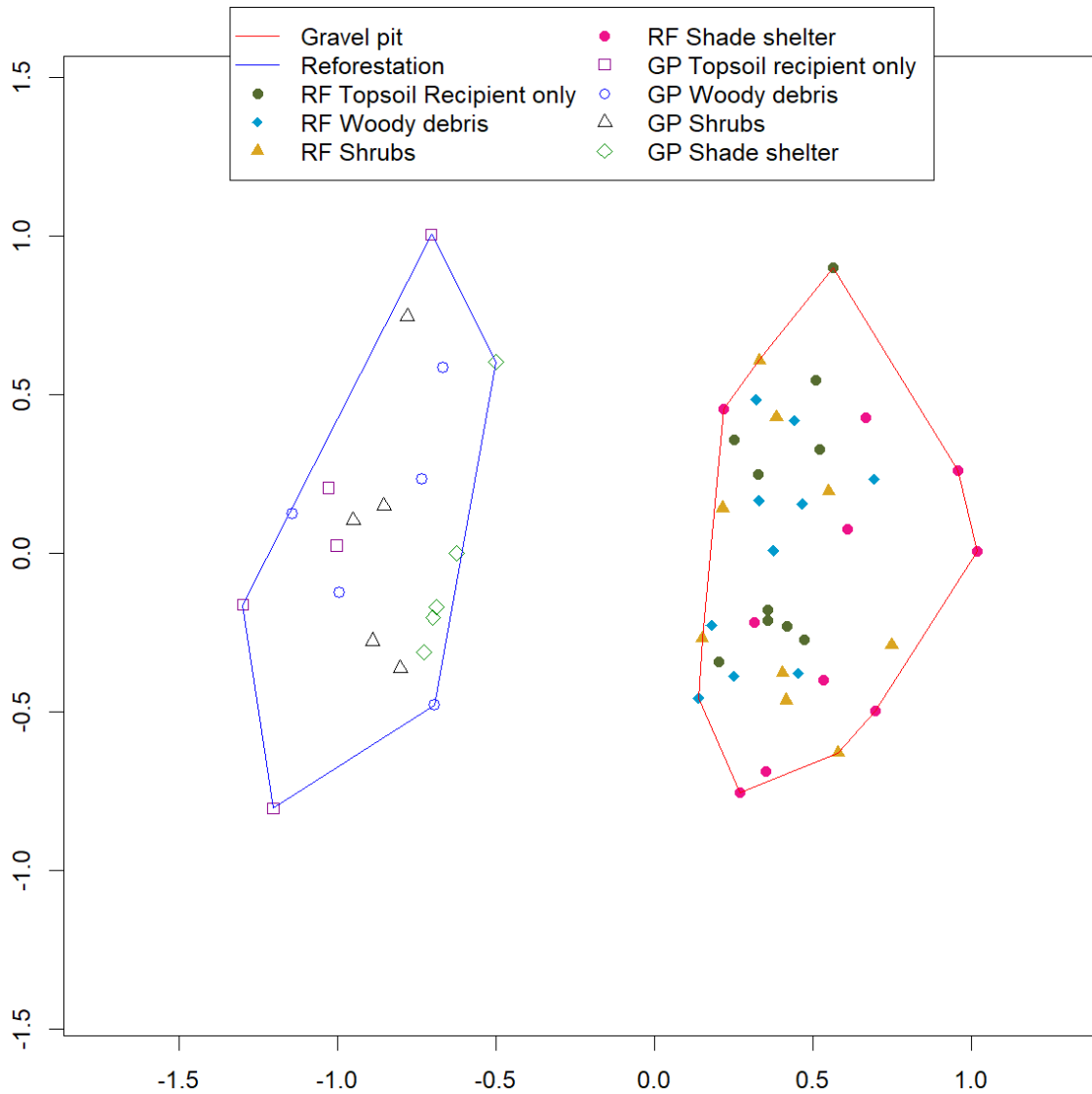


Figure I: NMDS using the Jaccard method of beta diversity of spring-sampled vegetation communities of sub-plot treatments within topsoil recipient plots in the two reforestation sites combined (blue outline) and the gravel pit (red outline). RF – 2015 & 2016 Reforestation, GP – Gravel pit.

## 2.2. Summer NMDS and PERMANOVA for sub-plot treatments

An NMDS on summer plant community dissimilarity was performed on sub-plot treatments within topsoil recipient plots in both the reforestation sites grouped together, and in the gravel pit site (Figure 7). Shade-shelter plots (green diamonds in figure 7) within the gravel pit site also seem to be visually distinct, but when testing all sub-plots by treatment within the gravel pit site through a PERMANOVA, there was no significant difference ( $F = 0.9$ ,  $R^2 = 0.17$ ,  $p = 0.474$ ). There was also no significant difference among treatments of sub-plots in topsoil recipient plots in the reforestation site ( $F = 1.3$ ,  $R^2 =$

0.10,  $p = 0.163$ ). In both cases, multivariate dispersion was not significantly dissimilar between different sub-plot treatment groups.



**Figure II: NMDS of Bray-Curtis dissimilarity of plant communities of subplot treatments in topsoil recipient plots in the reforestation sites (grouped) and the gravel pit. The NMDS was calculated on relative frequency of species in the summer sampling. Axes represent rank-ordered (non-metric) dissimilarity of plant communities. Red and blue lines mark the ‘hulls’ or furthest distribution of sub-plots within the gravel pit and reforestation site respectively in the two non-metric dimensions.**

## **Appendix 4A: Digital Elevation Model methods**

The Digital Elevation Model (DEM) data was acquired by Dr. Derek Robinson and Benjamin Meinen with assistance from L. Jonas Hamberg. The DEM were processed by Benjamin Meinen. It was not used for analysis, just background site information.

The methods below were written by Benjamin Meinen:

“A FLIR Systems R60 Skyranger UAV was used to collect optical imagery with the SR-3SHD payload. The payload captures 15 MP RGB 4608 x 3288 resolution images and has a 46-degree field-of-view. The UAV was flown at 90m above-ground-level and used parallel-axis flight lines with a 70% overlap to collect imagery. A ground control survey was done using 24 ground control points (GCPs) measured with Smartnet’s RTK-GNSS using a Leica Viva GS14 and Leica Viva CS15 field controller. UAV imagery and the ground control survey were processed in Pix4D to generate an optical mosaic and digital surface model.”

## **Appendix 4B: Flight time, temperature and vegetation data**

This appendix is a spreadsheet containing times and details of UAV flights undertaken in Chapter 4 as well as temperature measurements extracted and vegetation data used to compare with temperature in the same chapter.

The spreadsheet data can be accessed through Dataverse at: <https://doi.org/10.5683/SP2/M4RTY0> in the dataset titled “Spreadsheet appendices (3C and 4B) for L. Jonas Hamberg dissertation”



## **Appendix 4C: Tests, diagnostic plots and assumption testing**

This appendix is a PDF file of R code output of statistical tests, diagnostic plots and assumption testing for all analysis in Chapter 4.

The file name of this PDF file is “Appendix 4C - Tests, diagnostic plots and assumption testing.pdf”.

If you accessed this thesis from a source other than the University of Waterloo, you may not have access to this file. You may access it by searching for this thesis on <https://uwspace.uwaterloo.ca/UWSpace>.