The Early Post-restoration Population Dynamics and Community Interactions of a Former Agricultural Field in the Carolinian Canada Life Zone

by

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Declaration

I hereby declare that I am the sole author of this 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.

Abstract

Monitoring ecological restoration at the community scale provides insight into how the population dynamics and community interactions are progressing through time in comparison to a restoration's goals and reference conditions. This study monitored the early post-restoration dynamics of a sand plain located in the Carolinian Canada ecozone called Lake Erie Farms. The restoration consisted of restoring three habitats via sculptured seeding: a mesic forest, oak woodland, and sand barren. The hypothesis of this study is that the restoration efforts have established population dynamics and community interactions consistent with successional patterns expected from comparative literature. Community ecology, directed succession, and alternative stable states are the underlying theories that provided a conceptual and theoretical lens from which to study the objectives and hypothesis.

To gain insight into the community dynamics at Lake Erie Farms the vegetation abundance, seed abundance and viability of the seedbank, seed viability of the 6 most dominant species (3 most dominant native species & 3 most dominant weedy species), and soil moisture & pH were monitored. The analysis was conducted using a RMANOVA of a nested design ($P > 0.001$, 0.01 and 0.05) to compare the variables in relation to the site (i.e. the sum of all the quadrats), the restoration treatment nested within the site, the field nested within the site, the transect nested within the restoration treatment and the quadrat nested within the restoration treatment.

The significant findings of this study include: i) the restoration treatments are producing similar results as those expected from the literature, though there is evidence of the sculptured seeding treatment accelerating the successional stage at Lake Erie Farms compared to abandoned agricultural fields in similar ecosystems because of the presence of later-successional species; ii) the control areas are less diverse than each of the restoration units ($P>0.05$); and iii) the soil moisture among the treatments is beginning to diverge into the desired restoration units.

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Chapter 1. Introduction

Human activities have decreased ecological integrity through resource exploitation, recreation, urban development and agriculture. Due to humanity's intrinsic need for the environment to continue producing resources for our survival and comfort, we need to learn how to restore ecosystems that have lost their ecological integrity. Ecological integrity ``is the state or condition of an ecosystem that displays the biodiversity characteristic of the reference… and is fully capable of sustaining normal ecosystem functioning" (Society for Ecological Restoration International Science & Policy Working Group, 2004). To re-establish ecological integrity, we must learn to understand the earth's ecosystems to our fullest ability to develop sound techniques for ecological restoration. The term ecological restoration refers to activities aiming at repairing degraded ecosystems (Hobbs & Norton, 1996; Halle & Fattorini, 2004). A degraded ecosystem has reduced ecological integrity (Society for Ecological Restoration International Science & Policy Working Group, 2004). To be able to repair degraded ecosystems, a restoration ecologist must understand the existing communities (i.e. interacting populations) within and surrounding the degraded ecosystem (Palmer et al., 1997; White & Walker, 1997). Due to the complexity of ecosystems, this is no small task.

There are several possible successional trajectories for an ecosystem to follow in restoration. Knowing the possible trajectories that lead to emulating the goals of a restoration project is essential to developing the desired ecosystem (Parker, 1997; White & Walker, 1997; van de Koppel et al., 2001; Beisner, 2003; Choi, 2004; Society for Ecological Restoration International Science & Policy Working Group, 2004; Suding et al., 2004; Schröder et al., 2005). There are several approaches to establishing why restoration projects progress along different pathways. The approach used in this study is known as "Alternative Stable States" (Parker, 1997; van de Koppel et al., 2001; Beisner, 2003; Choi, 2004; Suding et al., 2004; Schröder et al., 2005). According to Temperton and Hobbs (2004), this approach uses a model based on "Community Assembly" drawing ideas from both stochastic and deterministic approaches. The deterministic approach portrays development as a consequence of physical and biotic factors, while the stochastic approach states that communities are structured in an essentially unpredictable, random process providing there is available habitat (Temperton & Hobbs, 2004). The "Alternative Stable States" hypothesis states that there is predictability because of constraints on community structuring, but one must be aware of the several possible trajectories due to chance or random events resulting in alternative endpoints (Beisner, 2003; Suding et al., 2004; Temperton & Hobbs, 2004). It is difficult to achieve the exact conditions laid out in a project's goals and reference conditions since each site has its own unique set of environmental conditions (Choi, 2004; Falk et al., 2006; Maetre et al., 2006). It is important, therefore, to look at the dynamics of each ecosystem and how it assembles into various stable states post-restoration. Studying the post-restoration dynamics will provide further insights into the complexities of ecological integrity, in both the short and long-term.

Directed succession is one of the methods used to restore degraded ecosystems and was the method used to restore the ecosystem of interest in this study (Rebele & Lehmann, 2002). Directed succession uses our knowledge of natural succession and advances the temporal scale of the degraded ecosystem by adding later successional species, processes (i.e. nutrient cycles, hydrology, or disturbance regimes) and physical conditions (Luken, 1990; Rebele & Lehmann, 2002). In this study directed succession was achieved by sowing various seed mixtures that were determined by existing environmental gradients, such as soil types, moisture content, or topography. This technique is known as sculptured seeding (Jacobson et al., 2004). It involves developing seed mixtures at are species rich with mid- to later-successional species that are typical of the habitat being restored, which are applied with specific seeding rates (Jacobson et al., 2004). These seed mixes and seeding rates are carefully matched to existing and desired site conditions (Jacobson et al., 2004).

Increasing our knowledge of ecosystem dynamics would aid in the development of a conceptual framework for restoration ecology, which would assist practitioners and researchers alike (Halle & Fattorini, 2004). This framework would advance the discipline by providing general concepts, rules and guidelines that could be referred to from a broad range of ecosystems, which would then uncover patterns from the outcomes observed and create feedback loops to make the framework even more accurate (Hobbs & Norton, 1996). Monitoring is therefore crucial to restoration as it provides data on ecosystem patterns. Unfortunately, it is often neglected because of time constraints, poor funding and minimal protocols. Monitoring data could provide insight into the specific components and processes that could be universally influential to completing restoration projects successfully (Hobbs & Norton, 1996; Higgs, 1997; Ruiz-Jaen & Aide, 2005). This study will contribute to the literature pertaining to monitoring ecosystem dynamics, assembly and integrity.

Another key element of ecological restoration is the ability to evaluate the success of a treatment (Lockwood & Pimm, 1999; Society for Ecological Restoration International Science & Policy Working Group, 2004; Ruiz-Jaen & Aide, 2005). Goals and desired reference conditions are required for restoration projects because they ascertain whether the project is succeeding (Lockwood & Pimm, 1999; Ruiz-Jaen & Aide, 2005; Hobbs, 2007). Lefler (2006) and Maetre et al. (2006) suggested the success of a restoration treatment could be based on three main attributes: the ratio of native to exotic species; ecosystem processes (e.g., hydrology and nutrient cycling); and species diversity at all spatial scales. Reference sites are also commonly used to compare the outcome and progression of a restoration project because they can be used as models for degraded ecosystems to emulate (Hobbs & Norton, 1996; White & Walker, 1997; SER, 2004; Ruiz-Jaen & Aide, 2005). They are often used to evaluate restoration projects post-restoration by comparing similar attributes of both ecosystems to determine the successional stage of the restoration site (Hobbs & Norton, 1996; White & Walker, 1997). There are often problems associated with using reference sites, such as differing temporal scales and historical land uses that affect the reference site's successional trajectory differently from the restoration site (Brewer & Menzel, 2009). Fortunately, an accepted practice is to develop expectations and goals from peer-reviewed literature, which was used for the purposes of this study (Hobbs, 2007; Brewer & Menzel, 2009).

1.1 Restoration of Agricultural Fields

In the last few decades in developed nations, there has been an increase in the abandonment of agricultural lands (Ramankutty & Foley, 1999; Cramer et al., 2008). This trend is providing massive potential for the recovery of degraded ecosystems. For instance, forest cover in southwestern Ontario, Canada was estimated at approximately 95 % before European settlement, and has decreased to less than 5 % cover because of agriculture and urban expansion (Lefler, 2006). If agricultural lands are being abandoned, than at least some forest cover can be restored. The Nature Conservancy of Canada, a non-profit organization geared towards environmental conservation and restoration, believes this is a possibility and has acquired agricultural lands in southwestern Ontario for restoration purposes. This study will investigate the ecological recovery of such lands.

1.2 Objectives of the Study

The Carolinian Canada ecozone is the most biologically diverse ecozone in Canada and has become a target for restoration projects by various groups (Carolinian Canada Coalition, 2004). This ecozone is located in southwestern Ontario and the northern boundary is defined by the species typical of the deciduous forest region in eastern North America, which are usually found in close proximity to the Great Lakes (see Appendix for map). There are very few to no published studies regarding the restoration of agricultural fields in the Carolinian Canada ecozone, particularly those that encompass sculptured seeding and include data from the first three years post-restoration. That being said, the expectations for the population dynamics and community interaction were derived from peer-reviewed literature regarding abandoned agricultural fields within the first three years post-abandonment and this information was used to compare with the findings of this study to show the affects of sculptured seeding.

This study improves the understanding of early post-restoration effects of sculptured seeding on a former agricultural field on the Carolinian Canada sand plains. To be more specific:

- The objective of this thesis is to measure early successional dynamics in a Carolinian ecosystem, post-restoration.

- The primary objective of this study for the Nature Conservancy of Canada is to assess the differences between the treatments and the control areas, as well as the differences between the treatments in terms of population and community dynamics.

- The hypothesis of this thesis is that the restoration efforts have established population dynamics and community interactions consistent with successional patterns expected from comparative literature (the null hypothesis is the dynamics and interactions are not similar).

To achieve these objectives, environmental variables that were indicative of key community dynamics were selected for monitoring (e.g. seed abundance and viability of both the seedbank and harvested seeds; above-ground vegetation abundance and diversity; and soil moisture and pH). These variables were analyzed to determine the current population dynamics and community interactions, and also to see what species are likely to emerge in future generations (i.e. forecasting the future species composition). A transparent depiction of this study shows the conceptual framework of this thesis (Figure 1).

1.3 Conceptual Construct of the Study

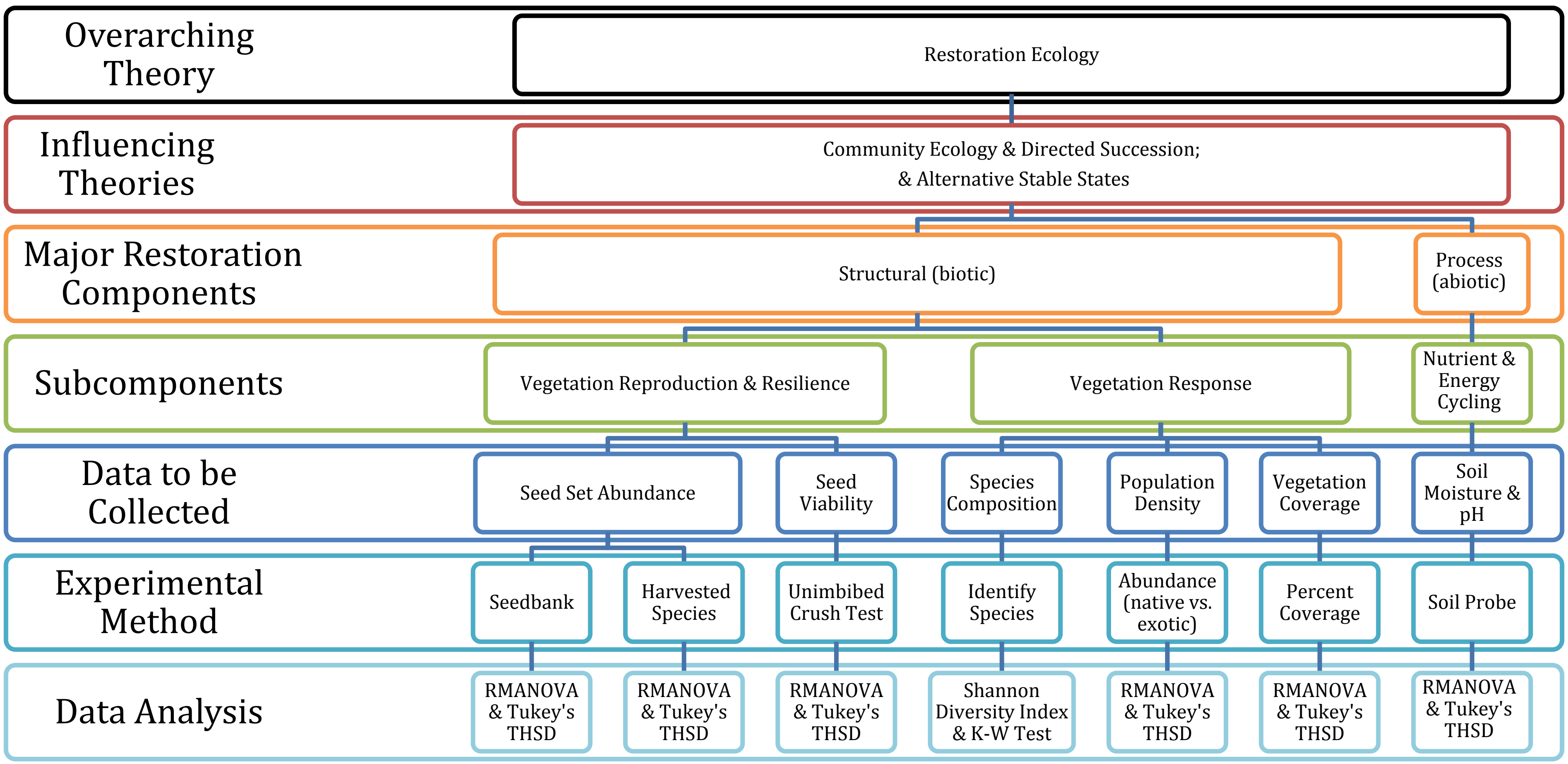

Figure 1 - This conceptual framework shows how I will achieve my objectives, and acts as a road map of this thesis. It starts by looking at the theoretical concepts underlying this study, then focuses on the components within those theories that were observed at the site, followed by how those components were studied and analyzed. Acronyms: RMANOVA – Repeated Measures Analysis of Variance; Tukey's THSD – Tukey's Test for Honestly Significant Differences.

Chapter 2. Literature Review

2.1 Introduction

Ecological restoration is the activity of "assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" with respect to its health, integrity and sustainability (Society for Ecological Restoration International Science & Policy Working Group, 2004). The practice of ecological restoration is becoming more popular with today's overuse of natural resources (Falk et al., 2006). It can recover a damaged ecosystem to a state that is within more acceptable limits compared to less damaged ecosystems. Thus the practice of ecological restoration is an attempt to recover acceptable ranges of ecosystem compositions, structures and dynamics (Falk et al., 2006). To achieve this, there must be clear goals of how to restore ecological integrity (Palmer et al., 1997; Hobbs & Harris, 2001; SER, 2004; Hobbs, 2007). One must test, therefore, the effectiveness of different restoration techniques through careful planning and ongoing monitoring (Palmer et al., 1997; Society for Ecological Restoration International Science & Policy Working Group, 2004; Ruiz-Jaen & Aide, 2005).

Another expression of ecological restoration is "directed succession" (Luken, 1990; Murphy, 2008). Ecological succession refers to the ecological community's change through time (Luken, 1990). Using "directed succession" means altering components in the ecosystem, such as species composition, nutrient dynamics, hydrodynamics, or geomorphology to change an undesirable trajectory into a desirable one (Falk et al., 2006). Another common method of changing undesirable trajectories is to remove perturbations (e.g. pollutants or invasive species), thus allowing the ecosystem to naturally heal itself (Falk et al., 2006).

Once a restoration treatment is initiated, the goal is to ensure that the ecological structures and processes that begin to self-organize will do so in a manner directly comparable to a reference site or predetermined goals and outcomes (Hobbs & Norton, 1996; White & Walker, 1997; Society for Ecological Restoration International Science & Policy Working Group, 2004; Ruiz-Jaen & Aide, 2005). Since appropriate reference sites are often not available, one may have to compare observed successional dynamics to those expected from published, peer reviewed literature (Murphy, 2005; Brewer & Menzel, 2009). These expectations derived from the literature are used to create clear, concise goals pretreatment when reference sites are nonexistent or poorly matched to the project. In addition, comparing sites that are still in a degraded state to the restored site often helps to determine if the restored site is changing in a desired direction (Brewer & Menzel, 2009). When choosing reference sites, there is still the question of whether restoring to historical conditions is appropriate, or even possible given species extinctions, invasions of exotic species, large scale changes to ecosystems, and climate change (Hobbs & Norton, 1996; Choi, 2004). Ecosystems are highly complex, creating the challenge of being able to clarify, empirically or from the literature, what the plausible trajectories are that lead to the emulation of the desirable state, which is essential to developing a successful restoration project (Parker, 1997; van de Koppel et al., 2001; Beisner, 2003; Choi, 2004; Suding et al., 2004; Schröder et al., 2005).

To determine if the goals are being met, ecological monitoring is necessary (Hobbs & Norton, 1996; Michener, 1997; Elzinga et al., 1998; Roberts-Pichette & Gillespie, 1999; Hobbs & Harris, 2001; Yoccoz et al., 2001; Sarr & Puettmann, 2008). Trends can be detected from the data obtained from monitoring, and often become more apparent the longer monitoring activities continue (Laughlin et al., 2008). Short-term monitoring (i.e. early post-restoration monitoring), however, is often overlooked during the evaluation of restoration success, yet could potentially show trends that reveal easily manageable and cost-effective repairs to the initial restoration treatments if undesirable community dynamics appear.

2.2 Approaches to Repairing Damaged Ecosystems

There is an ongoing debate in restoration ecology regarding the restoration of structural biotic components or the ecosystem's functions and processes (Korthals et al., 2001; Loreau et al., 2002).

2.2.1 Restoring Populations and Communities (Biotic)

The most common restoration practice is to restore the biotic community in terms of species biodiversity (Palmer et al., 1997; Lockwood & Pimm, 1999; Naeem, 2006; Oliver et al., 2007; Reinhardt & Galatowitsch, 2008). The main observation made when there is ecosystem decline is the loss of species; thus the driving goal is often to restore biodiversity (Allen, 2003; Young et al., 2005; Menninger & Palmer, 2006). Restoration ecologists believe that if the desired species composition is achieved, than the supporting processes will remain and may strengthen (Menninger & Palmer, 2006). This is supported by the fact the communities with more biodiversity tend to have more resilience to perturbations (Meiners et al., 2001; Meiners et al., 2004; Menninger & Palmer, 2006).

2.2.2 Restoring Function and Processes (Abiotic)

The underlying processes and geophysical constructs supporting species composition is another focus for restoration projects. If the functions and process of an ecosystem can be restored then the expectation is that the species will arrive through natural dispersion and migration (Benjamin et al., 2005; Falk et al., 2006). If the resources are available for a species to survive after natural dispersion, then it will establish itself into the population, demonstrating that restoring the abiotic conditions may play a more important role than restoring the biotic compositions (Drake, 1990; Hobbs & Harris, 2001; Bartha et al., 2003; Falk et al., 2006; Halle, 2007).

2.2.3 Restoring Biotic and Abiotic Based on the Causal Conditions

Only restoring the abiotic conditions may not be enough to prevent invasive species from colonizing instead of native species; preventing the desired species composition (Price & Weltzin, 2003; Huebner, 2007). It also does not restore rare and/or poor disperser species when restoration sites are longer distances from seed sources (Hewitt & Kellman, 2002). On the other hand, species compositions may not by achieved without proper resources and habitat. Depending on the causal conditions that degraded the ecosystem, restoring biotic, abiotic or both should be undertaken to achieve a successful restoration (Higgs, 1997; Gunderson, 2000; Diggelen et al., 2001; Hobbs & Harris, 2001; Choi, 2004; Maetre et al., 2006; Naeem, 2006).

2.3 Theoretical Framework of this Study

Many authors have developed conceptual frameworks for ecological restoration (Hobbs & Norton, 1996; Ehrenfeld & Toth, 1997; Hobbs & Harris, 2001; Halle & Fattorini, 2004; Hobbs & Halle, 2004; Temperton et al., 2004; Young et al., 2005; Andel & Aronson, 2006; Falk et al., 2006; Halle, 2007). Some of the common factors of these frameworks are shown in Table 1. The gap in the literature is the creation of one solid, singular framework that is strongly supported by the majority of restoration ecologists, both academics and practitioners alike.

Table 1 – Common elements of developing frameworks within the peer reviewed literature

Common Element	Supporting Literature
Setting restoration goals before treatment (sometimes with the use of reference sites)	(Aronson & Le Floc'h, 1996; Hobbs & Norton, 1996; Ehrenfeld & Toth, 1997; Palmer et al., 1997; White & Walker, 1997; Lockwood & Pimm, 1999; Diggelen et al., 2001; Hobbs & Harris, 2001; Choi, 2004; Society for Ecological Restoration International Science & Policy Working Group, 2004; Ruiz-Jaen & Aide, 2005; Hobbs, 2007; Brewer & Menzel, 2009)
Being aware of alternative trajectories and successional pathways	(White & Walker, 1997; van de Koppel et al., 2001; Beisner, 2003; Choi, 2004; Society for Ecological Restoration International Science & Policy Working Group, 2004; Suding et al., 2004; Ruiz-Jaen & Aide, 2005; Schröder et al., 2005; Suding & Gross, 2006)
The need for a conceptual foundation for restoration ecology	(Hobbs & Norton, 1996; Allen et al., 1997; Clewell & Rieger, 1997; Michener, 1997; Hobbs & Harris, 2001; Young et al., 2005; King & Hobbs, 2006)
Monitoring and evaluation is critical to determining restoration success and failure	(Hobbs & Norton, 1996; Michener, 1997; Elzinga et al., 1998; Roberts-Pichette & Gillespie, 1999; Hobbs & Harris, 2001; Yoccoz et al., 2001; Sarr & Puettmann, 2008)
Monitoring and evaluation data should be readily available to other restoration ecologists for comparisons and educational development	(Clewell & Rieger, 1997; Michener, 1997; Roberts-Pichette & Gillespie, 1999; Yoccoz et al., 2001)
The need to develop restoration standards – such as success criteria	(Hobbs & Norton, 1996; Clewell & Rieger, 1997; Oliver et al., 2007)

2.4 Area of Study

For this study, “community ecology” and “directed succession” are used as the basis for understanding ecological restoration and the ecosystems it affects since both concepts focus on observing community dynamics. In addition, the theories of “Assembly”, “Alternative Stable States” and “Resilience” add insight into how communities are organized and why random, unforeseen events and trajectories occur. These theories will inform my hypotheses and objectives by providing a conceptual and theoretical lens from which to interpret the results of my data analysis.

2.4.1 Community Ecology

Community Ecology is the study of species assemblages and their interactions with each other and their environment (Kikkawa & Anderson, 1986; Putman, 1994; Palmer et al., 1997). This area of study was founded from several concepts, such as natural history, plant geography, evolutionary ecology, ecological succession, species diversity, and ecosystematics (ecosystems and energy flow) (Kikkawa & Anderson, 1986). Community ecology is based on population-scaled studies, i.e. community membership (e.g. resource and threshold limitations), biotic relationships between species (e.g. competition, mutualism and predation), and patterns of these over time and space (Putman, 1994; Palmer et al., 1997). Niche theory is a fundamental aspect of community ecology in that species have evolved to fill different ecological functions based on selection pressures from biotic and abiotic factors (Putman, 1994).

According to Roughgarden (2009), community ecology does not have a general law about how species are distributed and how they interact with each other and their environment on a global or even regional scale. There are many notions and ideas regarding population dynamics, but there is not a general law, such as in physics (Simberloff, 2004; Roughgarden, 2009). Roughgarden (2009) believes that most community ecologists look at the structure of a community and base how they are formed on that structure. Perhaps if community ecology were based on community formation, a general theory could be: "local interactions act upon the species arriving at the community's boundary to produce a diversity of communities" (New South Wales Department of Primary Industries, 2005; Roughgarden, 2009). This means that dispersal and physical transport processes would supply the species, then various interactions would determine the species' entrance into the community, determine abundances and the potential change or disappearance of other species (New South Wales Department of Primary Industries, 2005; Roughgarden, 2009).

Finding general laws may not be an appropriate approach to studying community ecology (Simberloff, 2004). Simberloff (2004) argues that community ecology is a complex science where understanding will arise from being able to answer questions about natural patterns and phenomena. Due to the inherent complexity of individual species and how they interact, every study is essentially independent, meaning that general laws are likely impossible to form (Simberloff, 2004; Roughgarden, 2009). By building up a catalogue of case studies that point to rough generalizations regarding patterns, relationships between entities and processes, and causes of the patterns (i.e. mechanisms), community ecology may not require general laws

(Simberloff, 2004). Perhaps through this means community ecologists could find rules that apply across guilds of species at regional or even global scales.

2.4.2 Directed Succession

"Directed succession" is the acceleration or reversal of natural, spontaneous succession through the manipulation of an ecosystem (Prach et al., 2007). The goal of "directed succession" is to manipulate succession at the community level by changing the management of the site to enhance the desirable components and control or even eradicate the undesirable ones, such as exotic weedy species, on an ongoing basis (Prach et al., 2007).

Spatial and temporal scales are important to consider when directing succession (Yoccoz et al., 2001). Spatially, the landscape context of a restoration site is important to understand the proximity of local propagule sources, the land use history and the spatial patterns (Prach et al., 2007). The spatial scale is most often manipulated via population dynamics (as measured using variables such as growth, survivorship, and reproduction) and community interactions (as measured by testing for competition or mutualisms). There are two central spatial manipulations: i) physiochemical manipulations, which rely on changing the shape and chemical properties of the site, and ii) biotic manipulations, which consist of adding biota or controlling species establishment (Prach et al., 2007).

Temporally, the timing of ecosystem manipulations is important. For instance, seeding should be carried out in colonization windows for the species being planted, which are usually hard to predict because of several factors such as species competition or severe abiotic conditions (Prach et al., 2001; Bartha et al., 2003; Prach et al., 2007). It should also be taken into consideration that many seeds require cold stratification to break their dormancy period (Baskin & Baskin, 1998). This can be done either artificially in coolers, or naturally by seeding in the fall (Baskin & Baskin, 1998). There are techniques used for accelerating spontaneous succession such as planting relatively high densities of mid-successional species in an early succession ecosystem, which was done at this study site.

2.5 Other Contributing Theories

2.5.1 Assembly Theory

"Assembly Theory" is strongly related to "community ecology" in terms of their conceptual frameworks (Young et al., 2005). Assembly is the organization of the biota through a "filtering" process, i.e. constraints placed on community composition (Drake et al., 1999; Weiher & Keddy, 1999; Murphy, 2008). For instance, species invade an ecosystem either from an external species pool (i.e. seed dispersal) or through an internal species pool (i.e. seedbank) and try to establish in a suitable habitat with required resources (abiotic filters). If the species cannot survive, then other species (i.e. the biotic filter) will invade and compete for that niche until a local species pool is established. For a visual description of this, see Figure 2 which shows Fattorini and Halle's (2004) Dynamic Environmental Filter Model. Once the details of these filters are discovered, it will be possible to examine how these either affect the sequence in which species arrive and become extinct or how stochastic factors override the filters (Fattorini & Halle, 2004; Murphy, 2008). Long term monitoring of population dynamics and community interactions are a possible method to help identify these filters (Cramer et al., 2008). If assembly rules can be modeled or determined empirically, the final community composition could be predicted (Drake, 1990; Weiher & Keddy, 1999; Lockwood & Samuels, 2004; Falk et al., 2006). If the goal of "Assembly Theory" is to find filters, i.e. rules, that govern how ecosystem communities are created, then the goal is to look at the successional history for patterns pertaining to those filters (Lockwood & Samuels, 2004; Temperton & Hobbs, 2004).

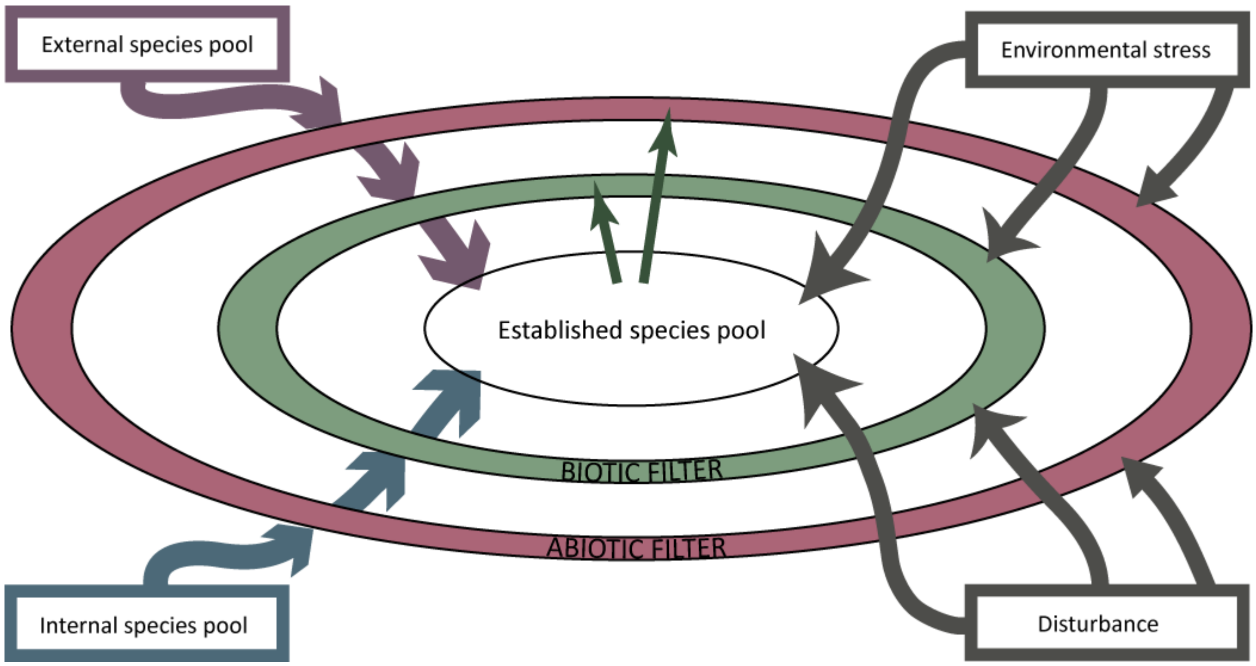

Figure 2 - Fattorini and Halle's (2004) Dynamic Environmental Filter Model

2.5.2 Alternative Stable States and Resilience

The theory of "Alternative Stable States" provides insight into the trajectories of ecosystems and why they sometimes stay in a particular state regardless of restoration actions (Parker, 1997; van de Koppel et al., 2001; Beisner, 2003; Choi, 2004; Suding et al., 2004; Schröder et al., 2005). The basic idea behind "Alternative Stable States" is that ecosystems are dynamic, meaning they move through succession towards stable states, and are perturbed into moving toward the same or another stable state (Gunderson, 2000; Beisner, 2003; Suding et al., 2004; Suding & Gross, 2006). To discover what pathway the restoration treatment caused the ecosystem to take, or if it was too resilient[1] to move out of its degraded state, it important to monitor post-restoration recovery (Gunderson, 2000; Suding & Gross, 2006). The challenge is to recognize desirable pathways and what to do when undesirable pathways emerge. Monitoring will add to the knowledge of how ecosystems respond and change over time to particular

[1] Resilience is measured by the magnitude of a disturbance or perturbation that can be absorbed before the ecosystem shifts its structures and processes that control the ecosystem's behaviour. (Gunderson, L. H. 2000. Ecological Resilience - In Theory and Application. Annual Review of Ecology and Systematics **31:**425-439.)

treatments, and will shape what interventions are appropriate for specific trajectories and ecosystem states, and when to use them.

The main relevance of "Alternative Stable States" and "Assembly Theory" to my study is to be aware of the several possible trajectories caused by random events, but to know that the constraints on community structuring from community assembly rules could create predictability (Beisner, 2003; Suding et al., 2004; Temperton & Hobbs, 2004). It is difficult to achieve the exact conditions laid out in a project's reference conditions since each site has its own unique set of environmental conditions (Choi, 2004; Falk et al., 2006; Maetre et al., 2006). It is still possible, however, to discern general, overarching patterns through monitoring to create guidelines for more effective and efficient restoration efforts (e.g. similar to the guidelines that make forensic scientists more easily unfold the events of a crime, although each crime has its own unique scene).

2.5.3 Conclusion

The gap in all of these theories is a lack of data on the dynamic recovery of habitats under self-organized or human assisted (restored) conditions. Ultimately, a tractable approach to closing these types of gaps is studying the post-restoration successional dynamics, then examine how deterministic factors (primarily the restoration efforts) and stochastic factors (weather, migration, dispersal, etc) are affecting the successional pathway of an ecosystem (Temperton & Hobbs, 2004). In other words, can we determine if and how "directed succession" is developing the expected post-restoration population and community dynamics (Murphy, 2005)?

2.6 Importance of Monitoring

Post-restoration monitoring is critical to the development of the knowledge of restoration ecology and to determine whether or not a project has been successful (Prach et al., 2001; Choi, 2004; Fattorini & Halle, 2004; Society for Ecological Restoration International Science & Policy Working Group, 2004). If the purpose of ecological restoration is to assist the recovery of degraded ecosystems, then restoration ecologists must have clear goals and objectives (Hobbs & Norton, 1996; Hobbs & Harris, 2001; Fattorini & Halle, 2004; Society for Ecological Restoration International Science & Policy Working Group, 2004; Hobbs, 2007). Monitoring is the key to validating whether a degraded ecosystem is recovering to meet those goals (Fattorini & Halle, 2004), and can also provide feedback for predictions, allow restoration programs to become adaptive, and serve as a base for other research (Prach et al., 2001). Restoration is often on a limited budget (particularly when completed by non-profit organizations as in this study) which means it is critical to assess through monitoring initiatives which restoration treatments work and which to avoid funding in the future.

Few ecological restoration projects or experiments actually examine the effects of "directed succession". Once restoration occurs, the habitat is often never monitored (Halle & Fattorini, 2004). My study will be one of the few that tests whether the successional dynamics of early restoration efforts are similar to those expected from the literature, and how directed succession has affected those successional dynamics.

2.6.1 Choosing Variables to Measure Community Dynamics

There are many possible variables to measure population and community dynamics resulting from "directed succession" in ecological restoration. The choice of variables depends on a thorough screening of the most commonly used variables that are the most successful indicators of population and community dynamics. The most elaborate example of this can be found in Tegler et al.'s (2001) article. They evaluated existing monitoring variables from a variety of sources (journals, experts, existing EMAN protocols) to find environmental variables used in hypothesis testing (Tegler et al., 2001). They found over 1770 variables and reduced that number to 188 by consulting an expert panel and using three broad criteria that the variables had to meet (Tegler et al., 2001). The next step involved screening the remaining 188 variables through a set of 20 detailed criteria that assessed the variables' "data quality, applicability, data collection methods, data analysis and interpretation, existing data and programs and cost effectiveness" (Tegler et al., 2001). 92 core monitoring variables were then tested for their effectiveness to detect environmental change, and were finally used to form a framework for monitoring a range of ecosystem components (Tegler et al., 2001).

According to Tegler et al. (2001) the variables shown in Table 2 are those that are most commonly used and indicative of population dynamics and community interactions on former agricultural lands.

Table 2 - Modified framework from Tegler et al.'s article *Ecological monitoring and assessment network's (EMAN) proposed core monitoring variables: An early warning of environmental change* (2001) showing the environmental variables which can be used to assess early restoration success of terrestrial ecosystems

Components	Structure	Monitoring Variable	Function (and process)	Monitoring Variable
Abiotic components			Sunlight & Moisture Infiltration	Stratification, total incoming & soil moisture content
			Temperature	Mean soil temperature
Biotic Community	Species richness & diversity (incl. indicator/exotic species)	Plants	Reproduction	Seedbank & seed viability
	Community biomass	Vegetation coverage/biomass	Community productivity	Vegetation coverage/biomass
			Nutrient cycling	Soil fertility & mycorrhizae presence
			Decomposition rate	Leaf litter depth

Other influential literature on the commonly used indicators and monitoring variables emphasize selecting variables based on the objectives of the monitoring program, and on answering the hypotheses on community dynamics (Elzinga et al., 1998; Yoccoz et al., 2001; Drysdale et al., 2007). A corollary is to understand that the choices of variables for these purposes will be influenced by cost, spatial variation (i.e. sampling design based on the scale of the variable being measured) and detectability (i.e. when sampling a population, some species are likely to be undetected, and unaccounted for) (Yoccoz et al., 2001; Murphy, 2008). Others, such as Oliver et al. (2007), have noted that a researcher often has to decide which scale to examine (e.g. spatially explicit and thus landscape scale, or community and population scale). While it may be desirable to examine all scales, this approach is more appropriate to multi-authored, decades-long series of studies. Therefore this study will look at the community and population scale.

By distilling the variables chosen by the authors in this literature review - and the extensive literature review conducted therein by those authors - the criteria for choosing the final set of monitoring variables for this study were:

1. The variable must be correlated with physical components and/or processes that are associated with the mechanisms influencing community dynamics
2. The variable should be correlated with typical restoration goals, such as vegetation species composition, biomass, soil stability, decomposition rates, interactions, etc.
3. The variable can be applied to a wide range of ecosystem types
4. Each variable should be operationally practical and scientifically defensible
    a. The variable should be easy to monitor and cost efficient: techniques are simple and can be learnt through a guide, manual or simple training course; equipment should be common and inexpensive; if the cost is high, the frequency of monitoring will be low to offset the cost
    b. The variable should be widely used in research projects and by several experts to demonstrate that it is scientifically defensible
    c. The variable should be measurable within the timeframe of a Master's degree (i.e. 1-2 years)

On that basis, the following are the monitoring variables (summarized in Table 3) used to measure early post-restoration dynamics on a former agricultural field, and allow for the comparison of measured dynamics with the ideal references developed from the published literature:

- The relative species diversity, abundance, and dominance - as classified in terms of "guilds" (herbaceous, shrub and trees) - is often measured because it captures much of the community scale interactions and responses (Tegler et al., 2001; Lefler, 2006). One must be careful not to infer too much about the ecological integrity of a habitat even if there is evidence relating to it (e.g. increased species diversity is related to expected mid-successional dynamics and improved ecological integrity) (Booth et al., 2003). It is also relatively easy to measure these variables, but the limitation associated with this ease is the difficulty to perform quantitative analyses; the analysis will be mostly descriptive with minimal quantitative findings. Fortunately, multiple scales can be quantitatively analyzed from these measurements (e.g. population scales using individual-scale variables)

- The ability of restored populations to reproduce is critical to restoration (Matsumura & Washitani, 2000; Menges, 2000; Tomimatsu & Ohara, 2002; Kolb, 2005; Moir et al., 2005). In a plant community, establishing a seedbank is often required to provide future generations (Menges, 2000; Tomimatsu & Ohara, 2002). Once established, the seedbank's regenerative capabilities will reduce degradation and invasion through the seeds' ability to outcompete exotic species and to germinate when conditions are appropriate, creating resilience to disturbances as well (Menges, 2000; Loreau et al., 2002; Andel & Aronson, 2006). For these reasons, measuring the abundance of the fruit and seed sets directly from the plants, in the litter layer, and in the top 5 cm of soil were incorporated into my study. After determining if the seedbank is established or establishing, I determined seed viability (Menges, 2000; Sawma & Mohler, 2002; Borza et al., 2007). In this step, I used the unimbibed seed crush test to assess whether the seeds are viable or not (Menges, 2000; Sawma & Mohler, 2002). If the seedbank is establishing with seeds of low viability, then the resilience of the seedbank will be low, indicating that further restoration may be required.

- Soil moisture & pH were measured to assess whether the treatment is failing because of the lack of moisture, extreme pH or the ineffective treatments. According to Zhanag et al. (2005), soil moisture is a major factor that constrains processes such as the beginning of the growing season, root activities (e.g. growth, interactions with organisms), decomposition rates and nutrient cycling. Measuring soil moisture and pH will also ensure the other monitoring variables are providing data on the responses to the restoration treatment rather than other stochastic events (e.g. drought).

Table 3 - Variables used to test and monitor early post-restoration population dynamics and community interactions on a former agricultural field

Abiotic	Soil pH
	Soil moisture content
Biotic	Species diversity, abundance & dominance
	Seedbank abundance and seed viability
	Harvested seed viability from dominant species

2.7 *Expectations of Early Post-restoration Community Dynamics*

Since there were no appropriate reference sites that could be compared to the temporal scale and structural components of the site used for this study, expectations of the community dynamics at this stage post-restoration were derived from the literature to determine the effects of directed succession. Due to the literature gap regarding early post-restoration monitoring, however, there was little information on the expected conditions of a site in the Carolinian zone 3-5 years post-restoration. Fortunately, some comparable information was found from a successional study in the Hutcheson Memorial Forest in central New Jersey, USA, which is also located along the northern edge of the Carolinian Canada Life Zone (Cary Institute of Ecosystem Studies, 2004). This study began when 10 agricultural fields were abandoned, either being plowed or left alone (no active restoration treatments), in the spring and fall of 1958 and 1959 (Cary Institute of Ecosystem Studies, 2004). After each field was abandoned, the Buell-Small Succession Study began monitoring the ecological community (Cary Institute of Ecosystem Studies, 2004).

The outcomes that are of relevance to this thesis are the vegetation's percent cover taken from the 2x0.5 m quadrats. What is of particular interest are the species found in the first 20 years as shown in Table 4 and when their percent cover was the most dominant (Pickett, 1982):

Table 4 - Buell-Small Successional Study dominant species list (Pickett, 1982)

Phase	Species	Year	Phase	Species	Year
1 (year1-4)	*Ambrosia artemisiifolia*	1		*Lepidium campestre*	10
	Mollugo verticillata	1		*Trifolium pratense*	11
	Digitaria sanguinalis	2		*Convolvulus sepium*	13
	Barbarea vulgaris	3		*Poa pratensis*	15
	Erigeron canadensis	4		*Agrostis alba*	12
	E. annuus	5	**3 (year 16-20)**	*Rhus glabra*	19-20
	Plantago lanceolata	6		*Lonicera japonica*	17
	P. rugellii	2-3		*Juniperus virginiana*	19
	Oxalis stricta	3		*Acer rubrum*	20
2 (year5-15)	*Rumex acetosella*	5		*Poa compressa*	17
	Daucus carota	5		*Acer negundo*	18
	Aster spp	7		*Solidago graminifolia*	19
	Chrysanthemum leucanthemum	8		*Rhus radicans*	20
	Hieracium pratense	10		*Rosa multiflora*	20
	Hieracium florentinum	12		*Solidago juncea*	19

According to Myster and Pickett (1988), the four most important species in early successional communities are: *Ambrosia artemisiifolia* L., *Chenopodium album* L., *Hedeoma pulegoides* L., and *Barbarea vulgaris* L. because they best define the early oldfield community. They also found a pattern that early species populations have early peaks where the total percent cover exceeded 100%, with persistent tails, and that there is much species overlap at a given time (Myster & Pickett, 1988; Bartha et al., 2003). Myster and Pickett (1988) found that similar initial field conditions produced similar species patterns, although the year of abandonment also affected these patterns. Essentially, population patterns are individualistic (Myster & Pickett, 1988).

Another interesting outcome from the Buell-Small Successional Study is the temporal decline of dominant early successional species. For instance, *Erigeron annus* L. decreased from a mean of 25(+)% cover to less than 5% cover after 5 years; *Digitaria sanguinalis* (L.) Scop. was nearly gone after 3 years while *Ambrosia artemisiifolia* L. was almost gone after 15 years (Pickett, 1982). It will be interesting to see if the population patterns at this study site will be similar.

Although not as similar to the site used for this study as the Buell-Small Successional Study, Blatt et al. (2005) conducted an oldfield succession study at two abandoned agricultural fields in southeastern Ontario. The expert botanists and ecologists working on this study from the Queen's University created an early successional species list found in Table 5 (Blatt et al., 2005). It could be expected that these species should be present at this study site.

Table 5 - Early succession species list created by an expert panel of botanists and ecologists from Queen's University (Blatt et al., 2005)

Species		
Agropyron repens	*Melilotus alba*	*Polygonum convolvulus*
Ambrosia artemisiifolia	*Melilotus officionalis*	*Polygonum persicaria*
Barbarea vulgaris	*Oenothera spp.*	*Rumex acetosella*
Cerastium vulgatum	*Oxalis stricta*	*Setaria glauca*
Daucus carota	*Panicum capillare*	*Stellaria media*
Echium vulgare	*Phleum pratense*	*Thlaspi arvense*
Erigeron annuus	*Plantago lanceolata*	*Verbascum thapsus*
Lepidium campestre	*Poa annua*	
Medicago lupulina	*Poa pratensis*	

Blatt et al. (2005) found that succession progressed quicker on wetter soils, which is verified in Bornkamm's study (1981). Soil saturation is how nutrients are transported to the plants. Therefore moister soils that are not oversaturated will provide nutrient more readily to plant roots. Plants growing on sandy soils, however, would have a harder time finding nutrients since the soil has a poor water retention capacity (Zwart, 2006; Verhallen, 2009). Sandy soils hold approximately 100 mm (4 in) of available water within the depth of typical root penetration, which is about 1.20 m (48 in) (Yaremcio & Tames, 2001). Soil fertility is also affected by soil pH, where the ideal pH for nutrient availability is 6.0-7.0 (Zwart, 2006; Verhallen, 2009). Many of the soils in Ontario typically have a pH of 7 or higher because of their calcareous base, whereas sandy soils tend to have a lower pH because of their rapid drainage (Verhallen, 2009).

Pickett and Bazzaz (1978) conducted a study of the 6 dominant weedy species of Illinois (*Ambrosia artemisiifolia* and *Polygonum pensylvanicum* (native weeds); *Amaranthus retroflexus* & *Abutilon theophrasti*, *Chenopodium album* and *Setaria faberii* (exotics). The study consisted of having these six species compete for space on a soil moisture gradient in a glasshouse experiment. The results showed that the biomass of the whole assemblage peaked around 40% soil moisture, then rapidly declined at less than 10% soil moisture (Pickett & Bazzaz, 1978). Generally, all of these species occupy broad niches, with a great deal of species overlap, but the conditions must remain broad in order for these plants to survive (Pickett & Bazzaz, 1978).

Reinhardt & Galatowitsch's (2008) study shows that sowing high native seed densities will decrease exotic species interference. This is verified by a study conducted by Murphy (2005) where restoration sites transplanted with high densities of native plant species show the ability to suppress invasive and non-native species. Also, vegetation traits linked to competitive ability, growth and seedbank persistence tend to persevere over time (Pywell et al., 2003). One of the main goals for my study site is to strongly discourage exotic species; therefore the Nature Conservancy of Canada planted 92 native species using sculptured seeding (i.e. sowing particular species based on existing environmental gradients).

2.8 Gaps in the Literature

There is a lack of clear standards for the practice of restoration and evaluation for success, an issue that the Society of Ecological Restoration is trying to rectify (Hobbs & Norton, 1996; Allen et al., 1997; Clewell & Rieger, 1997; Oliver et al., 2007). This issue is directly related to the lack of a clear conceptual basis for which restoration ecologists can plan their projects and research studies, though many individuals and groups trying to develop one (Hobbs & Norton, 1996; Ehrenfeld & Toth, 1997; Hobbs & Harris, 2001; Hobbs & Halle, 2004; Halle, 2007).

A short fall in the literature is information on long-term monitoring and evaluations to determine where restoration efforts are meeting their predetermined goals. Some positive outcomes from evaluating restoration success are: increasing the understanding of ecosystem dynamics, site-specific conditions and successions, and general successional trajectories; and developing better methods to ensure restoration success based on past successes and failures (Hobbs & Norton, 1996; Clewell & Rieger, 1997; Falk et al., 2006). If monitoring data were readily available to restoration ecologists, meta-analyses could be performed to increase our understanding of the practice and theory from site-specific strategies to broad theoretical hypothesis testing (Bakker et al., 2002). It would also allow for comparisons of similar sites at similar temporal scales, and the comparison of successful or unsuccessful restoration techniques. A drawback of not having monitoring data available along with the restoration techniques is that practitioners and academics are becoming more isolated rather than working together to increase the knowledge of the field (Clewell & Rieger, 1997).

There is a scarcity of long-term successional research (Michener, 1997; Bakker et al., 2002; Choi, 2004). Though there are some long-term plot-based studies, such as the Buell-Small Succession Study in New Jersey, the research in this area is limited (Bakker et al., 2002). An extension of this is the monitoring of restored agricultural fields, and their successional activities post-restoration. Information is needed to restore agricultural fields back into their natural state successfully, something that long-term monitoring data could provide (Bakker et al., 2002). This study will directly fill a gap in the literature regarding monitoring data from the succession of a restored agricultural field in the Carolinian Canada Zone.

2.9 Contributions from this Study

2.9.1 Contributing to the Literature

A major contribution of this study to the literature is adding information on early post-restoration successional and community dynamics created from sculptured seeding within the Carolinian Canada Zone, where there is currently limited to no knowledge or comparative data. Knowledge of how to restore lands in this ecozone and the results from continued monitoring at this study site will be a major contribution to the literature from the Nature Conservancy of

Canada if they continue their monitoring efforts and create publications regarding their monitoring data, restoration techniques, and evaluation outcomes.

This study provides data that could be used for comparisons for future studies at this site, which will help develop trends for restored agricultural fields restored in the Carolinian Canada Life Zone. It also provides data to compare to restoration sites that are in similar successional stages post-restoration in similar ecosystems, such as the Buell-Small Succession Study (Cary Institute of Ecosystem Studies, 2004).

2.9.2 Contributing to the Practitioners

Ecological restoration practitioners can use this study to compare to other sites that have used sculptured seeding to determine the validity of this technique. This study also adds to the knowledge base of early post-restoration indicators of community dynamics. In the future, this study could be used to determine whether further restoration interventions should have been completed to avoid an undesirable future state. This study also provides information on the typical soil moisture & pH measurements at this site and within the Carolinian Canada ecozone.

Chapter 3. Methods

3.1 Study Site

This study was conducted at “Lake Erie Farms”, a former agricultural field where the Nature Conservancy of Canada initiated an ecological restoration project in the spring of 2006. Lake Erie Farms is located in Southwestern Ontario in the Canadian Carolinian ecozone next to the South Walsingham Sand Ridges Natural Area, which has been identified by the Ontario Ministry of Natural Resources as having provincially or regionally significant representative ecological features(see Appendix for a map of the surrounding natural areas). The Lake Erie Farms property includes approximately 61 ha of agricultural fields that produced tobacco and corn until 2005. There are also 4 ha of the farm’s footprint. Following the conservation targets for this area, it is the intent of the Nature Conservancy of Canada to restore the agricultural fields to a diverse mix of native communities compatible in structure and function with surrounding Carolinian Canada habitats. More specifically, as outlined in the Lake Erie Farms Property Management Plan, the goal of the restoration project is to “restore agricultural fields to a mosaic of woodland and sand barren communities” (Nature Conservancy of Canada, 2006).

In my study, I did not add any further treatments, but in 2008 and early 2009 monitored the response of the vegetation community to the restoration treatments.

3.1.1 Norfolk County History and Environment (adapted from Presant & Acton, 1984)

Lake Erie Farms is situated in the South Walsingham Township of Norfolk County in southern Ontario (see Appendix for map). This county was settled by Europeans in the early 1800s. During settlement, the land was heavily deforested for residential areas and agriculture. By 1921, 64% of the developed land was being used for growing crops, while approximately 30% was being used for pastures and idle or fallow land, while the remaining 5% was used for growing fruits and vegetables. Around this time, the nutrients of the sandy soils typical to the county were depleted and the wind began eroding the land where there was exposed soil. This caused abandonment of the fields and led to some reforestation. In 1925, flue-cured tobacco was first planted in the fields, and by 1930 5,666 ha were in tobacco, which stabilized the sands. The addition of windbreaks, reforestation and strip cropping increased the agricultural value of the area. In 1981, 84% of cleared land was used for crops and 0.5% for pasture. In 2006, of Norfolk County's 160,700 ha, approximately 24% was forest cover and 74% was farmland (NEAC, 2006).

LANDFORMS

Klinkenberg (2002) conducted a study on the landforms represented in significant natural areas, and how they reflect the rare plant records of this ecozone (Table 6). A map of the top five representative landforms in the Carolinian zone of Canada can be found in the Appendix. Lake Erie Farms is primarily on the Sand Plain, but is relatively close to the Clay Plain (Klinkenberg, 2002). Klinkenberg (2002) found that the Sand Plains supported the highest number of rare plants in the Carolinian zone of Canada, making this restoration particularly important to increase the area of refuge for those rare plants.

Table 6 – Representation of Landforms within the Carolinian Canada Zone (Klinkenberg, 2002)

Landform	% of study region	Landform	% of study region
Escarpments	0.51	**Limestone plains**	1.61
Till moraines	10.09	**Shale plains**	0.46
Spillways	8.38	**Sand plains**	17.59
Kame moraines	3.97	**Clay plains**	16.36
Till plains (undrumlinized)	11.40	**Eskers**	0.12
Till plains (drumlinized)	12.66	**Beaches & shorecliffs**	0.34
Drumlins	0.04	**Peat & muck**	1.42
Bevelled till plains	14.88	**Water bodies**	0.16

SOILS (Presant & Acton, 1984)

Norfolk County is composed primarily of sandy soils. Most of the sands in the county were originally deposited by deeper water glacial lakes, and have been eroded by wind which created eolian landscapes. Lake Erie Farms is located on the sand plains of Norfolk County where the sands are "thought to have been deposited in shallow water stages of glacial lakes… [ranging] in thickness from less than 1 m to over 20 m". Aside from the occasional eolian dune and moraine, the sand plains are quite flat, with the area gently sloping from the northwest toward Lake Erie causing slow runoff from precipitation. Lake Erie Farms is sitting on a Dundee formed bedrock at ranges from 3-115 m below the sand, which consists of weakly cherty, fossiliferous limestone and minor shale.

The north fields at Lake Erie Farms are primarily Plainfield Soil (PFD) (85%), with a small area of Plainfield Soil in dune phase (PFD.D) (12%), small areas of Brant Soil (BRT) (2%) and Waterin Soil (WRN) (1%); in the central fields, there is primarily PFD (65%) with some PFD.D (34%), and a small area of Wattford Soil (WAT) (1%); and in the southern fields, there is primarily PFD (90%) with some PFD.D (10%). All percentages are approximate coverage based on the observations of the soil map overlaid on the restoration fields at Lake Erie Farms (see Appendix for Lake Erie Farms Soil Map). Plainfield soils "have developed on a metre or more of windblown eolian sands", and the textures are mostly fine sand throughout, although there can be loamy fine sand and sand on the surface A horizons (Presant & Acton, 1984). These soils are rapidly to well-drained causing droughtiness that restricts plant growth. The organic content in the surface horizons is quite low, (less than 2%), which also restricts plant growth and water retention in the soil. As indicated by the Percent of Normal Moisture (Drought Model) from Agriculture and Agri-food Canada (2007), Lake Erie Farms is located within the 85-115% normal moisture zone, which is the average range for Canada's agricultural lands (see Appendix). These soils are also medium to slightly acidic in the surface and subsurface horizons. The PFD.D is very similar to PFD with the exception of having more fine sand in all the horizons. The soil classification is typically Brunisolic Gray Brown Luvisol.

Most forest tree species are limited by the low nutrient and moisture levels of these soils, with the exception of Black Walnut and Beech. According to Zwart (2006), the average pH of Ontario soils ranges from 6.5-7. The soils in Southern Ontario, however, are more calcareous and thus have pH readings of 7 or higher, although sandy soils tend to have lower pH because of their low buffering capacity (Verhallen, 2009).

CLIMATE AND WEATHER (Environment Canada)

The data from three Environment Canada climate stations within 25 km of Lake Erie Farms from 1951 until 2006 were compiled and analyzed in excel. There was no data readily available from any of the climate stations within 25 km of Lake Erie Farms after 2006; therefore the search was extended to stations within 50 km for data from 2006-2008. The mean annual temperature from 1951-2006 is 8 °C, with a maximum mean annual temperature of 9.4 °C and a minimum mean annual temperature of 6.7 °C. The average total annual precipitation from 1951-2006 is 967.3 mm, consisting of 846.1 mm of rainfall and 132.9 mm of snowfall. From 2006-2008 (the years of interest for this study), the Delhi CS weather station located 40 km from Lake Erie Farms recorded the average total annual precipitation as 1056.6 mm, which is 89.3 mm higher than the average from 1951-2006 (Figure 3). This station also recorded the mean annual temperature as 8.5 °C, which is half a degree higher than the mean from 1951-2006 at the closer station (Figure 4).

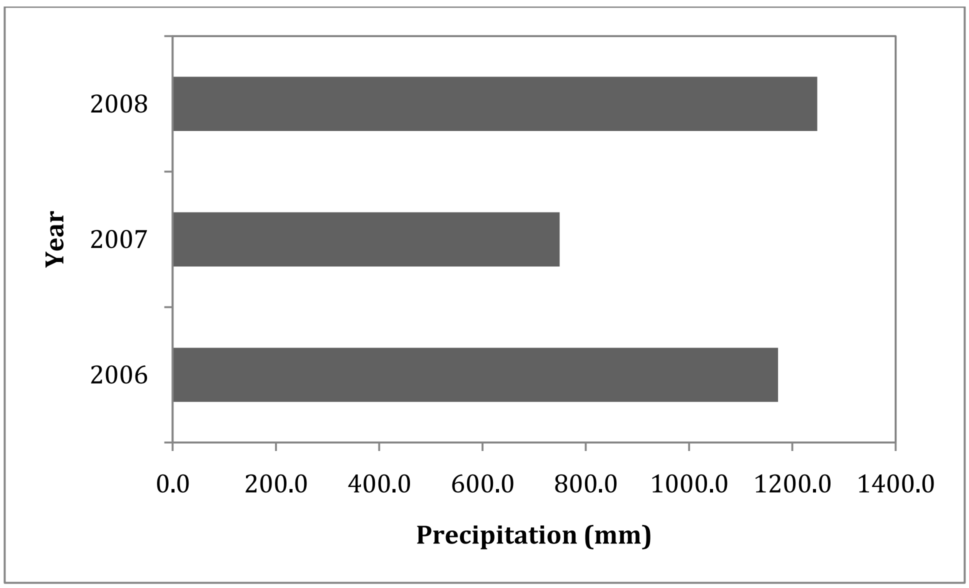

Figure 3 – The annual total precipitation (mm) from the Delhi CS weather station located 40 km from Lake Erie Farms

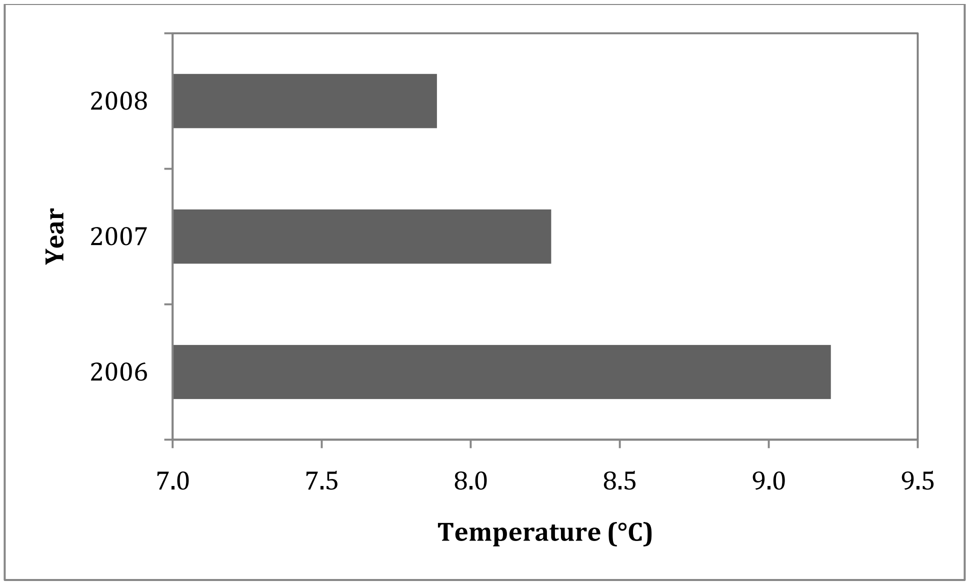

Figure 4 – The mean annual temperature (°C) from the Delhi CS weather station located 40 km from Lake Erie Farms

3.1.2 Lake Erie Farms Description

The adjacent 420 ha of the South Walsingham Sand Ridges Natural Area has a complex set of protected vegetation communities including sand barrens, sand prairies, savannas, oak woodlands and forests responding to several soil types, moisture regimes, topography and other disturbances (Carolinian Canada Coalition, 2004). These communities served as the basis for the goals developed by the Nature Conservancy of Canada for Lake Erie Farms (Nature Conservancy of Canada, 2006).

Fields and natural communities are both present at Lake Erie Farms (see Appendix for a map of Lake Erie Farms). All natural communities are no more than 200 m away from any given restoration area. The interior of the field was the primary focus of the restoration efforts, as the 10 m margin of the fields are expected to be naturally colonized by seeds and stolons dispersing from the natural communities. Throughout the restored parts of the property, five broad community types, referred to as restoration units, were chosen as the target communities based on environmental gradients of moisture, soil, aspect, elevation, drainage and surrounding vegetation (Nature Conservancy of Canada, 2006). These five restoration units were treated with a planting approach known as *sculpted seeding*: the restoration units are seeded according to their particular environmental gradients with locally-sourced seeds and cuttings planted in a random pattern within each unit (New South Wales Department of Primary Industries, 2005).

The Nature Conservancy of Canada (2006) expects subcategories of community composition (i.e. dry versus moist oak woodlands) to develop over time.

3.1.2.1 Restoration Units

As outlined by the Lake Erie Farms Property Management Plan, the restoration units are (Nature Conservancy of Canada, 2006):

Mesic Forest (4.6 ha, 3 plots): Drainage tiles were disabled in fields to restore natural hydrology. Mesic forest communities were planted in low lying areas. Target stem density for planted woody shrubs and trees is 2000 - 2500/ ha. The long term objective is to have a dense canopy closure (>75%) dominated by red maple, yellow birch, red oak, white oak, tulip tree, largetooth aspen, trembling aspen and black cherry. To achieve this goal, 80 species were included in a seed mixture sowed at 85.4 kg/ha. Of the 80 species 59% were woody, 6% grass, and 35% herbaceous.

Oak Woodland (40.7 ha, 25 plots): A multi-species oak woodland mix was planted in high, dry uplands, and was the dominant restoration unit across the fields. Target stem density for planted woody shrubs and trees is approximately 2000/ ha. This restoration unit will, through succession, consists of a mosaic of insipient sand barrens, sand prairies, and oak woodlands. Target canopy cover will be patchy and relatively open (25 – 60% cover). 75 species were included in the oak woodland seed mixture, including 38% woody, 15% grass, and 47% herbaceous species. This seed mixture was sown at 88.4 kg/ha.

Sand Barren (9.5 ha, 15 plots): The uppermost soil layer (A horizon) was removed or turned over on dry knolls to reduce competition from weedy species and provide habitat for sand barren associations. Height and shape of sand dunes was maintained by adding B or C horizon sand (e.g. from vernal pool and hibernacula excavations) to sites where the A horizon had been removed. Sand barren planting mix consisted mainly of forbs and grasses with sparse shrubs and black oak (target canopy cover <25%). 38 species were used in the sand barren seed mixture, including 21% woody, 21% grass, and 58% herbaceous species. This seed mixture was sown at 36.6 kg/ha. Several mosses colonize newly restored agricultural fields and may pose barriers to colonization by forbs and sand-dwelling insects. Adaptive management of extensive moss communities (e.g. mechanical disturbance or spot burning) may be required over time to maintain sand barren associations.

Experimental Control (6.2 ha; 15 plots): Long-term ecological monitoring plots were established in the fields. Experimental control treatments (i.e. unplanted field areas) represent approximately 10% of the field area. Data collected across the monitoring plots are used to determine the relative effectiveness of active (planted) and passive (unplanted) restoration approaches across taxa, and the use of specialized habitat features described above.

3.1.2.2 Plot Design

The Nature Conservancy of Canada placed 163 2x2 m quadrats on Lake Erie Farms within the four restoration units after completing the restoration treatments. The reason they chose a larger quadrat size than EMAN's standard of 1x1 m was related to the low initial density of plants. The larger quadrat size reflects the need to have representativeness within the samples (Nature Conservancy of Canada, 2006). Within each restoration unit, transects were systematically located to intersect a representative variety of topographic, aspect, slope and restoration unit parameters, typically starting 20 m from a forest edge through the restoration unit with quadrats placed every 20 m along the transects, no closer than 5 m from the edge of another restoration unit. Each quadrat is marked with a T-bar in the southwest corner.

3.1.2.3 Baseline Data

The Nature Conservancy of Canada provided their data collected from the year after the restoration treatments were conducted in 2006 until 2008, which includes the percent cover of bare ground and the percent cover of each species within each quadrat. The Nature Conservancy of Canada also provided a species list including the seed mixes and planting rate from their restoration treatments (See Appendix). Due to the difficulties of identifying some species in the month of June, the asters were categorized as one species for this summary: 39 out of the 87 species seeded were present in the data collected from 2006-2008. In 2006, there were 22 species present from the restoration's planted species list, 31 species in 2007, and 25 species in 2008.

An assumption that I am making is that the native seedbank was no longer present at Lake Erie Farms because of extensive, long-term agricultural practices (Feldman et al., 1997).

3.2 Experimental Design

3.2.1 Boundaries of the Study

Spatially, I need to know where certain elements of the environment exist and in what abundance. In order to gain this information, I conducted a population survey on the former agricultural lands at Lake Erie Farms to account for the population trends of the treated and control areas.

Temporally, I need to know when new components arrive, and if existing abundances change and where. Due to the limitations of the survey, I may not account for the presence of all populations on the property. Another temporal boundary is when I monitor. Timing is critical because of vegetation lag times for growth and reproduction (Price & Weltzin, 2003). All of the species, including spring ephemerals, should be present from late spring to early summer; therefore it would be appropriate to conduct the first monitoring survey during this time period, though it will be difficult to identify some of the species this early in the growing season.

For my study, I am using a categorical approach to studying the community dynamics, meaning I am observing the species and guilds in a snapshot in time, i.e. the extant community (Drake et al., 1999). I also have some components of the topological approach in my study by looking at the change in species through time to see if any patterns emerge, such as invasions or extinctions of specific species or guilds (i.e. assembly rules) (Temperton & Hobbs, 2004).

3.2.2 Sample Size

A power analysis was used to determine the number of plots to sample in order to have a robust data set. Out of the 163 plots available to sample on the site, the power analysis showed that 54 plots would be enough to provide robust findings (i.e. a 95% confidence interval, alpha = 0.05). A simple random sample, conducted in excel, of the 163 plots was taken to determine which 54 plots would be monitored in my study (Elzinga et al., 1998; Murphy, 2008).

3.2.3 General Analysis

Unless stated otherwise, the datasets collected from measuring the environmental variables below were analyzed with a Repeated Measures Analysis of Variance (RMANOVA) of a nested design. Before the dataset could be analyzed with the program, it first had to be tested to see if it met the assumptions of an RMANOVA. This was done using the Normality Test to ensure the data are normally distributed, and a Bartlett's Test to test for homoscedasticity. The RMANOVA was run using 'R' with a critical values (P-value) of 0.001, 0.01 and 0.05 (Murphy, 2008). Two post-hoc tests were used on the species abundance data: Tukey's test for Honestly Significant Differences ($HSD_{0.05}$) and Pillai's Trace.

In the RMANOVA the species found at Lake Erie Farms were divided into four major plant categories for comparison purposes: i) exotic weedy species, ii) native weedy species, iii) native non-planted species, and iv) planted (i.e. seeded) species. The 'planted species category' refers to species that the Nature Conservancy of Canada planted during the sculptured seeding treatment. The 'native non-planted species category' refers to native species that were not included in the sculptured seeding treatment, and have arrived via natural seed dispersal. Native species are those that are indigenous in origin to the Carolinian habitat rather than introduced by humans (i.e. exotic species) (Pysek et al., 2004). Weedy species are those that typically reproduce and spread aggressively, and are sometimes referred to as colonizing or invasive species (Pysek et al., 2004). Exotic species have a tendency to be weedy because they do not have natural controls or barriers in an alien environment (e.g. predation), and are thus able to reproduce and spread quickly (Murphy, 2008).

These categories were compared with the RMANOVA in relation to the site (i.e. the sum of all the quadrats), the restoration treatment nested within the site, the field nested within the site, the transect nested within the restoration treatment and the quadrat nested within the restoration treatment.

3.2.4 Assessing Species Diversity and Richness

Species diversity is often both an indicator and a goal in ecological restoration as it is the most common assessment of restoration success or failure (Ruiz-Jaen & Aide, 2005). Though comparative studies have been equivocal, restoring much of the species diversity may be important to ecological resilience by ensuring there is enough genetic material to adjust to disturbances, and to provide a good indication that the more complex structures and functions exist and are working (Booth et al., 2003).

For this study, the vegetation was identified and quantified in each 2x2 m plot to determine species abundance, evenness and richness. This was achieved on June 10th 2008 by conducting a 1x1 m herbaceous vegetation identification and abundance count within each plot, as well as a

5x5 m tree and shrub identification and abundance count as per the protocols laid out by the Ecological Monitoring and Assessment Network (Roberts-Pichette & Gillespie, 1999). Both of the 1x1 m and the 5x5 m sub-plots were measured from the t-bar, which is located in the northeast corner of each sample area.

The total species diversity within the community was determined using a proportional abundance model, which incorporates both evenness and richness into the model (Booth et al., 2003). The model is based on the Shannon Index. The assumptions of this index are that all individuals are sampled at random, are sampled from infinitely large populations, and that all of the species in the community will be sampled (Booth et al., 2003). The following equation was used to calculate the species diversity:

$$H^{'} = -\sum[p_i(\ln p_i)]$$

Where:

- $H^{'}$ is the Shannon diversity Index
- p_i is the proportional abundance of a given species (species "i")
    - $p_i = n_i/N$
    - Where n_i is the number of individuals in a species (i), and
    - N is the total number of individuals of all the species

To then find species evenness, the following equation was used:

$$E = H^{'}/\ln S$$

Where:

- S is the number of species found

The data showed the proportional abundance of the species within each plot, treatment, transect, field and site.

The results from my abundance count were pooled with the species found from the data collected by the Nature Conservancy of Canada annually mid to late July since the restoration treatment took place (for the pooled species list, see Appendix). The Nature Conservancy of Canada measured the percent cover of each species within 2x2 m quadrats.

For the Shannon Index to be analyzed with the RMANOVA, the data had to be log transformed to meet normality requirements.

3.2.5 Assessing Viability of Harvested Seeds and Fruit Sets

Seed viability is the ability of seeds to germinate and grow (Sawma & Mohler, 2002). In order to test viability, the three most dominant weedy species: *Conyza canadensis* (L.) Cronquist. (native weed), *Arenaria serpyllifolia* L. (exotic weed), and *Veronica arvensis* L. (exotic weed) and three most dominant native species: *Rudbeckia hirta* L. (planted species), *Monarda fistulosa* L. (planted species), and *Solidago canadensis* L. (native non-planted species) were determined from the abundance counts completed on June 10th 2008. 40 individuals of each of these six species were randomly selected from the 54 quadrats for seed harvesting. As these plants bloomed, 5-10% of the total inflorescence per individual plant was sampled using a bagging method to capture the inflorescence's seed set. This method involved bagging the inflorescence in a sheer material, in this case bridal veil, that is porous enough to not harm the plant (i.e. air, sunlight and moisture are able to pass through), while still being able to gather the fallen seeds for quantification and viability tests. The veil bags were placed just before the flowers blossomed, which was closely monitored to ensure quality in this measurement. These bags were collected on Oct 7th 2008 after the seeds had completely dropped.

Smaller individuals, which were also spring ephemerals, were collected on June 14th, 2009. These were *Veronica arvensis* and *Arenaria serpyllifolia*. The seeds collected from the plants were stored in a cool, dry environment with the open paper bags to ensure the seeds would dry out and not mold until lab analysis could be conducted.

The seed and fruit sets were analyzed in the University of Waterloo Restoration Ecology Lab from October 2008 until June 2009. This analysis consisted of using the air dried seeds from the bagged samples collected July 22nd 2008, October 7th 2008 and June 14th 2009, and removing the seeds from the bags and if necessary from the fruit capsules by gently squeezing the capsule with a pair of tweezers. After the seeds were removed from the bags, 100 seeds were randomly selected if the individual sample consisted of more than 200 seeds (Sawma & Mohler, 2002). Anything under 200 seeds resulted in the entire sample being tested for viability (Sawma & Mohler, 2002). To test the seed's viability, the collected seeds underwent the unimbibed seed crush test (Sawma & Mohler, 2002; Borza et al., 2007). Generally, the data produced by this method was shown to be indistinguishable from other viability tests (e.g. tetrazolium and imbibed crush tests) (Sawma & Mohler, 2002). The unimbibed seed crush test is also a better technique to use for this thesis because it is less time consuming (e.g. than germinating or tetrazolium tests) (Sawma & Mohler, 2002; Borza et al., 2007). Another advantage of the unimbibed seed crush test, it that unlike the germination test, there is no need to break the dormancy of the seeds (Sawma & Mohler, 2002; Borza et al., 2007). To summarize, the unimbibed seed crush test is less laborious, expensive and time consuming; indicating this method is appropriate for my study (Sawma & Mohler, 2002).

To conduct the unimbibed seed crush test, the first step is to remove any seeds that appear to have deteriorated, and then randomly select a subsample of 25 seeds, with 4 replications (Sawma & Mohler, 2002). After this, I applied a firm pressure on the seed with the flat side of a pair of tweezers to test how easily the seed broke (Sawma & Mohler, 2002). Although Sawma and Mohler (2002) found that if the seed broke it is not viable, I added an extra analysis to ensure quality in this method: the float test. For the float test, the seeds that floated were considered not viable and were crushed to see what a non-viable seed looked like (e.g. oily, chalky, colour). This extra measure set a standard on what a non-viable seed looked like when crushed. This was repeated for the viable seeds that sank in the float test. Once the viability could be determined consistently via the crush test, it was used without the float test (approximately 3 to 5 samples per species).

Data were arcsine transformed to meet both the normality and homoscedasticity requirements**.**

3.2.6 Assessing Seedbank Seed Viability

Hempy-Mayer and Pyke (2008) recommended measuring the established seedbank by taking samples from the litter layer (25 cm^2) and the top 3 cm of the soil immediately below the litter sample. Due to the absence of the litter layer at Lake Erie Farms, only the soil sample was taken. During the field sampling on June 10th 2008 a 30 cm deep soil sample was taken in the north-eastern corner of each plot with the Oakfield Apparatus Company soil corer, which has a 2.5 cm exterior diameter and a 2 cm interior diameter. From each sample, the top 5 cm of the soil was placed in a paper bag to later be cleaned for seed collection, while the bottom 25 cm was placed in a plastic bag for freezing and possible nutrient sampling, which time did not permit. All of the soil samples were placed in a Beaumark dryer deep freeze until September 15th 2008, and then the paper bags containing the top 5 cm of the soil samples were thawed for lab analysis.

In the lab, the soil samples containing the top 5 cm of soil were individually emptied onto a sieve which allowed everything (i.e. sand, particulates, and debris) but the seeds to fall through (Hempy-Mayer & Pyke, 2008). I did not have to use water to help with this process, as the sandy soil was easy to separate through the sieve on its own. After separating out the sand and small particulates, I spent approximately 30 minutes per sample looking for seeds (among the larger particulates and rocks) on the sieve to keep consistency among the samples.

The seeds collected from the soil samples were identified under a Vista Vision model 11389-219 microscope and keyed primarily with Delorit's (1970) "Illustrated Taxonomy Manual of Weed Seeds", Martin's (1961) "Seed Identification Manual", and Montgomery's (1977) "Seeds and fruits of plants of Eastern Canada and Northeastern United States".

Two analyses were performed on the soil sample containing the seedbank sample: seed abundance and seed viability. To complete the seed abundance analysis, the data had to be modified with a logx+1 to ensure the data met normality requirements. To complete the seed viability analysis, the data were arcsine transformed to meet both the normality and homoscedasticity requirements.

3.2.7 Assessing Soil Conditions

At each quadrat, I measured the moisture and pH of the soil using a soil moisture and pH meter on June 10th, July 22nd and Oct 7th of 2008. The measurements were taken in the northeast corner of all 54 quadrats. This measurement assesses whether the treatments are working or failing because of the treatments themselves or because of uncontrollable environmental variables such as precipitation, soil water retention, or pH. It could also indicate whether the treatments are causing the soil condition to diverge into the conditions characteristic of each restoration unit.

Chapter 4. Results and Discussion

4.1 Species Diversity and Richness

4.1.1 General Summary Statistics

Based on the species abundance data that I collected in 2008 the following general statistics were derived:

Total Number of Species present per Treatment: The oak woodland and mesic forest treatments supported the highest number of species with 31 species in each treatment (75 planted in the oak woodland and 80 planted in the mesic forest), followed by the control treatment with 20 species (zero planted), and lastly the sand barren with 17 species (38 planted) (see Figure 5). See Appendix for plant list. The oak woodland treatment may have the same number of species as the mesic forest treatment three years post-restoration because it is a larger area and thus has a higher probability for species to naturally colonize. The sand barren treatments supported low species diversity likely because of the low soil moisture content (see Figure 14). The sand barren treatments have the lowest mean percent soil moisture content of all the treatments, as described in section 4.4 Soil Condition, page 54.

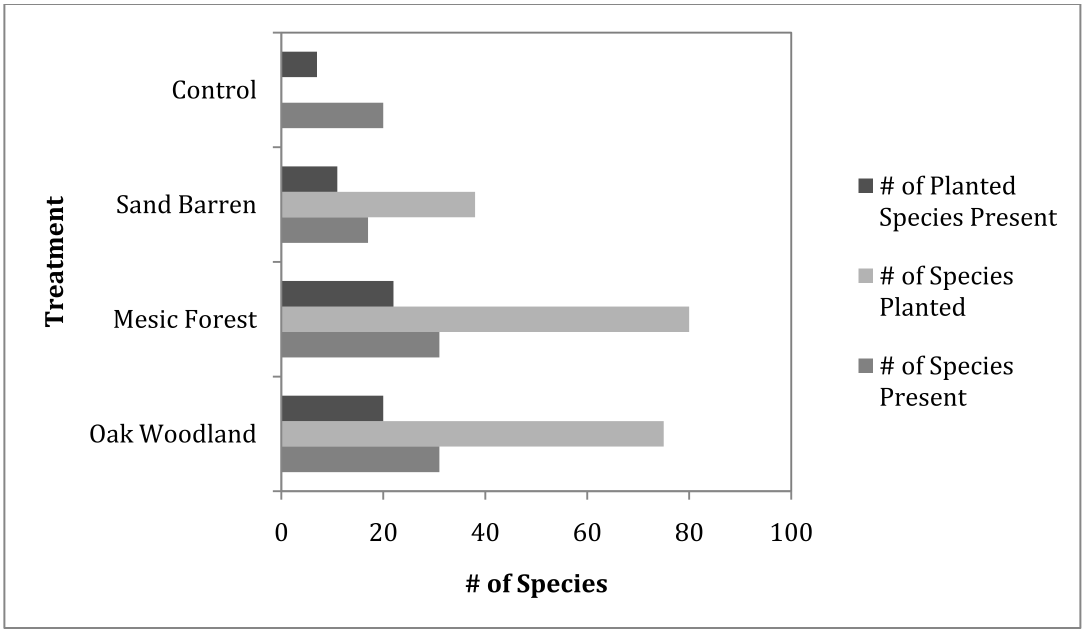

Figure 5 – The number of species found in the sampling areas of each treatment in 2008 compared to the number of species sowed in each treatment by the Nature Conservancy of Canada in 2006.

Number of Species present per Quadrat: The oak woodland treatment supported the highest species diversity (7 quadrats with 8+ species) followed by the mesic forest treatment (4 quadrats with 8+ species). There were no quadrats with 8+ species in the sand barren or control treatments.

According to MacDougall et al. (2008), Pywell et al. (2002) and Van der Putten (2000) increasing the species richness of a seeding mixture for primarily sandy soils will increase the likelihood of a restoration effort reaching its target compositional goals. In this regard, the results from Lake Erie Farms are following a similar pattern to these studies. Van der Putten (2000) and Pywell et al. (2002) used anywhere from 15-41 species in their species-rich seeding mixtures.

Proportion of Native Species to Weedy Species per Treatment: The oak woodland treatment supported 20 non-native species and 32 native species; the mesic forest supported 20 non-native species and 36 native species; the sand barren supported 15 non-native species and

17 native species; and the control areas supported 9 non-native species and 14 native species in 2008 (see Figure 6).

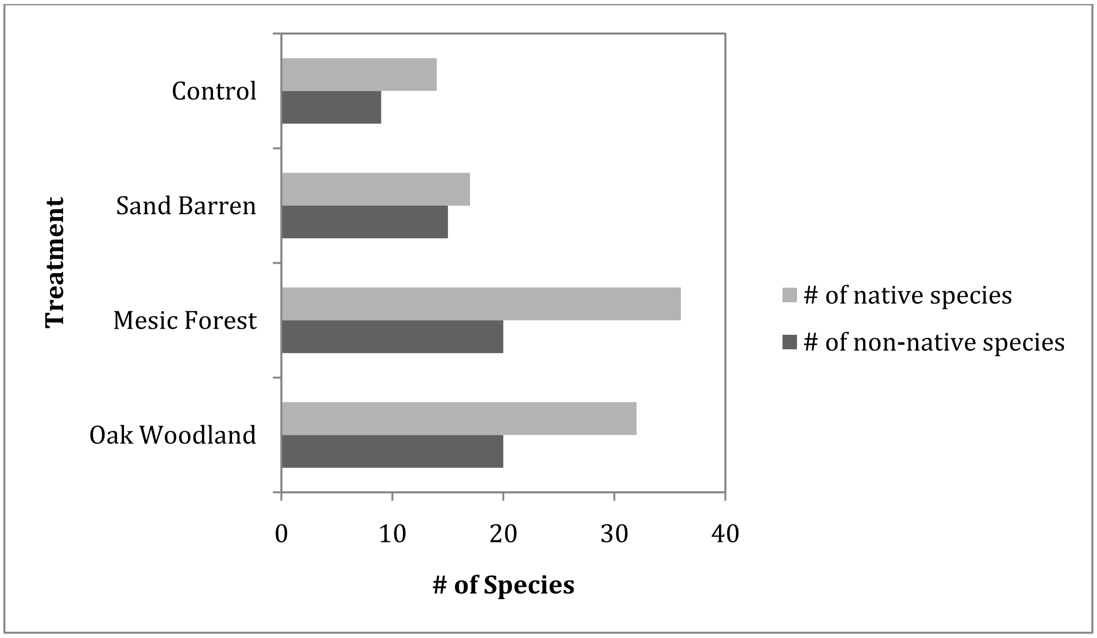

Figure 6 – The number of non-native species per treatment compared to the number of native species per treatment from the Nature Conservancy of Canada's monitoring data from 2008.

Van der Putten (2000) found weed suppression with a sown seed mixture of only 15 species. The weed suppression at Lake Erie Farms is noticeable in the ratio of weedy species to native species in 2008, and should be more noticeable in the near future.

4.1.2 Shannon Diversity Index

The results of the Shannon Diversity Index based on the data I collected can be seen in Table 7. The responses from the Shannon Diversity Index were compared with a nested RMANOVA between fields, treatments and the site as a whole; there were no statistically significant responses. For comparison purposes, the number of species the Nature Conservancy of Canada found within all of their plots from 2006-2008 is shown in Table 8.

Table 7 - The values from the Shannon Diversity Index based on the data collected by Katelyn Inlow in 2008 from 54 plots at Lake Erie Farms. The last column shows the H-value of the Shannon Diversity Index based on the Nature Conservancy of Canada's 2008 data for all the plots.

Location	# of Species	Diversity (H-value)	Evenness (E-value)	NCC Diversity (H-value)
Lake Erie Farms	36	1.778	0.496	2.32
North Field	16	0.907	0.327	2.34
Central Field	30	1.000	0.294	0.41
South Field	19	0.565	0.192	2.48
Control Plots	15	0.775	0.286	0.86
Oak Woodland Plots	23	0.997	0.318	2.35
Mesic Forest Plots	26	0.404	0.124	2.49
Sand Barren Plots	15	0.659	0.243	1.57

Table 8 - Number of species the Nature Conservancy of Canada found at Lake Erie Farms from 2006-2008

Location	Number of Species		
Year	**2006**	**2007**	**2008**
Lake Erie Farms	41	95	80
North Field	41	74	57
Central Field	n/a	42	24
South Field	n/a	47	39
Control Plots	12	26	23
Mesic Forest Plots	29	68	59
Oak Woodland Plots	32	73	55
Sand Barren Plots	18	45	27

Although not significant, the most diverse field is the central field, which is where the most diverse treatment (i.e. mesic forest) is located (Table 7). While this trend is consistent with the percent cover data collected by the Nature Conservancy of Canada in 2008 in terms of number of species present in those areas, it appears it is too soon after the restoration treatments to demonstrate a statistically significant outcome because of the varying species diversity in 2007 (Table 8). According to the data collected by the Nature Conservancy of Canada from 2006-2008, the quadrats from the control areas has a statistically significant ($P < 0.05$) lower species diversity than each of the restoration treatment quadrats using a Kruskal Wallis test (Table 8).

4.2 Species Abundance

After pooling my data with the Nature Conservancy of Canada's and testing the dataset with a RMANOVA, the abundance of planted, native non-planted and native weedy species showed a statistically significant response to the treatments (Table 9-Table 11). The abundance of the planted species was highest in 2007 across all the treatments except for the control area where the abundance remained the same from 2007 to 2008 (Table 10 and Table 11). The native non-planted species abundance was highest in 2008 across all treatments with the exception of the sand barren treatment where the abundance remained the same from 2007 to 2008 (Table 10 and Table 11). The abundance of native weedy species was highest in 2007 across all of the treatments. The exotic weedy species did not respond significantly to the treatments, but did significantly peak in abundance in 2007 across all the treatments (Table 9-Table 11). The planted and exotic weedy species are showing similar changes in abundance from year to year; the native non-planted species are steadily increasing in abundance; and the native weedy species have increased but are fluctuating (Table 11).

Table 9 - Significant responses of vegetation abundances within each species category nested within the treatments over time with a RMANOVA based on the data collected by the Nature Conservancy of Canada from 2006-2008. (* = p<0.05, ** = p<0.01, *** = p<0.001)

	Treatment			Time			Treatment x Time		
Species Category	**MS**	**F**	**P**	**Pillai**	**F**	**P**	**Pillai**	**F**	**P**
Planted	6.15	4.91	*	0.88	57.19	***	0.81	47.26	***
Native non-planted	7.19	5.47	*	0.80	45.13	***	0.85	53.65	***
Native weedy	6.88	5.11	*	0.82	49.02	***	0.83	50.92	***
Exotic weedy	4.17	3.08	0.153	0.74	42.81	***	0.77	45.81	***

Table 10 - The number of species within each treatment organized by species category from 2006-2008 using the Nature Conservancy of Canada data

Species	Mesic Forest			Oak Woodland			Sand Barren			Control		
Category	2006	2007	2008	2006	2007	2008	2006	2007	2008	2006	2007	2008
Planted	17	27	22	17	28	20	6	17	6	3	6	6
Native non-planted	1	12	13	2	11	12	0	3	3	1	3	6
Native weedy	1	4	2	2	5	3	2	3	3	1	3	2
Exotic weedy	9	21	19	11	25	18	10	20	12	7	14	8

Table 11 - The total number of species organized by species category from 2006-2008 using the Nature Conservancy of Canada's data

Species Category	Lake Erie Farms		
	2006	2007	2008
Planted	43	78	54
Native non-planted	4	29	34
Native weedy	6	15	10
Exotic weedy	37	80	57

The high abundance of planted species is likely because of the sculptured seeding conducted by the Nature Conservancy of Canada in 2006 (Table 11); actively seeding native species should increase their abundance and out-compete the weedy species, which is likely why the exotic weedy species are at comparable numbers to the planted species (instead of much higher than the planted species) (Pywell et al., 2002; MacDougall et al., 2008). The native non-planted and native weedy species have increased in abundance from 2006-2008 because of the proximity and quantity of natural areas surrounding the fields, which are a source for the dispersal native seeds (Prach et al., 2007). It is too early to make discernable conclusions from these small temporal scale patterns since the fluctuation of species composition in the first 10-15 years post-agriculture is typical, and seeding may not accelerate an ecosystem's successional trajectory past 10-15 years post-agriculture (Pickett, 1982; Myster & Pickett, 1994). Continued long-term monitoring at Lake Erie Farms will discern whether this acceleration has occurred.

An interesting outcome from the analysis is the presence, and increase in abundance of the planted species in the control quadrats; demonstrating that the planted species have dispersed to these areas in as little as three years. This should be kept in mind by the Nature Conservancy of Canada and anybody else that is conducting research with control plots near treatment areas as the data could be skewed by natural dispersal of the planted species. The data does show that the control areas are progressing more than expected from other abandoned agricultural fields that have received no treatment and are in an early successional stage (Pickett, 1982; Blatt et al., 2005).

The increase in abundance of the planted and native non-planted species by the third year post-restoration suggests the treatments are progressing acceptably. One year after the treatments were conducted there was only a 1 % mean cover for both the planted and native non-planted species, which increased to 12 % and 28 % respectively in 2008 (Figure 7). A corresponding response is the decrease in native weedy species and exotic weedy species from 2006 to 2008, though it was statistically insignificant because of the high standard deviation of the means (Figure 7). The native weedy species had a 6 % mean cover while the exotic weedy species was 21 % in 2006, reducing to 5 % and 14 % respectively in 2008 (Figure 7). A cause for these responses could be that there was high species richness in the seeding treatments. High species richness can resist invasion by exotic species and thus achieve target species composition (Pywell et al., 2002; MacDougall et al., 2008).

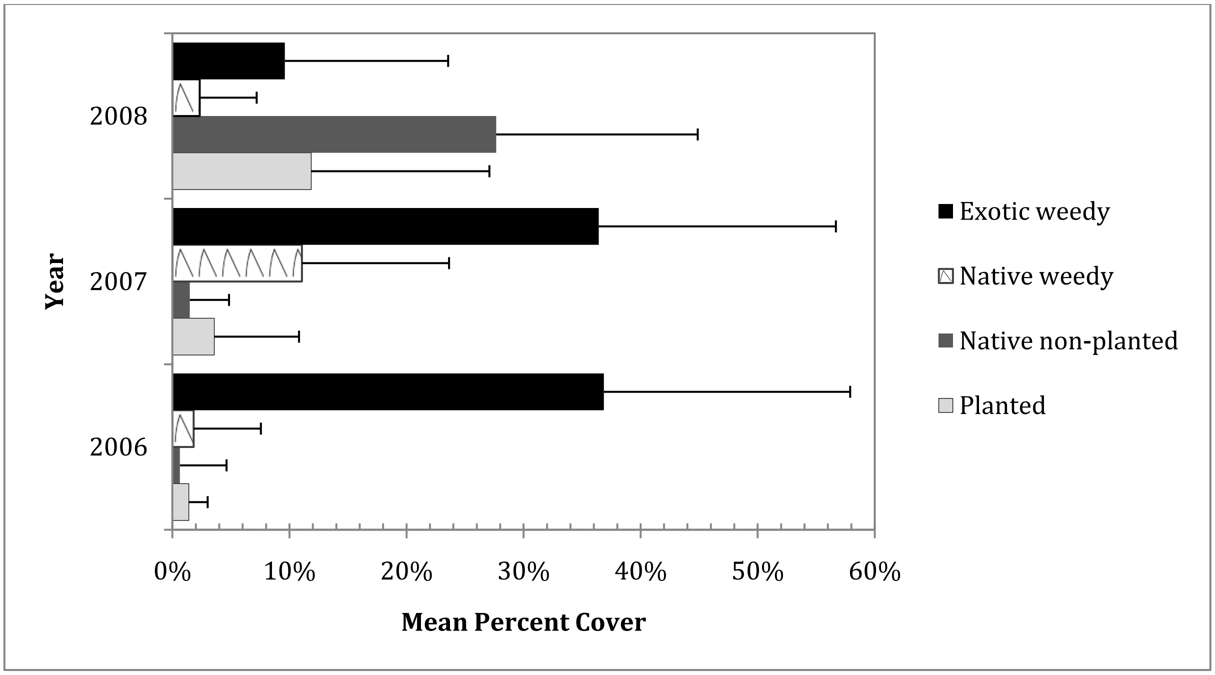

Figure 7 – Mean percent cover of each species category in the first three years post-restoration at Lake Erie Farms

Some of the species found at Lake Erie Farms had a statistically significant response to the treatments and the effect of the treatments over time (Table 12). The abundance of *Anthemis cotula* L. was highest in 2007 having a mean percent cover of 4.2 %, which decreased to 1.5 % in 2008 within the Oak Woodland treatment. This species often invades cultivated areas and is able to withstand competition and low resource availability (Erneberg, 1999). *Anthemis cotula* is, however, also unable to germinate under low light intensity; this is why it is usually less abundant at mid- to late-successional stages (Erneberg, 1999). Given this comparative information, it appears the sculptured seeding treatment has accelerated the successional stage of the oak woodland restoration unit to the point that these areas are no longer optimal for the growth of the species.

The abundance of *Salix bebbiana* Sarg. was highest in 2008 with a mean percent cover of 2.0 % within the Oak Woodland treatment. *Salix bebbiana* is typically a pioneer species in riparian habitats with moist sandy soil (Tesky, 1992). This species may have had a high abundance in 2008 due to high amount of precipitation in 2008 as shown in Figure 8 (Environment Canada).

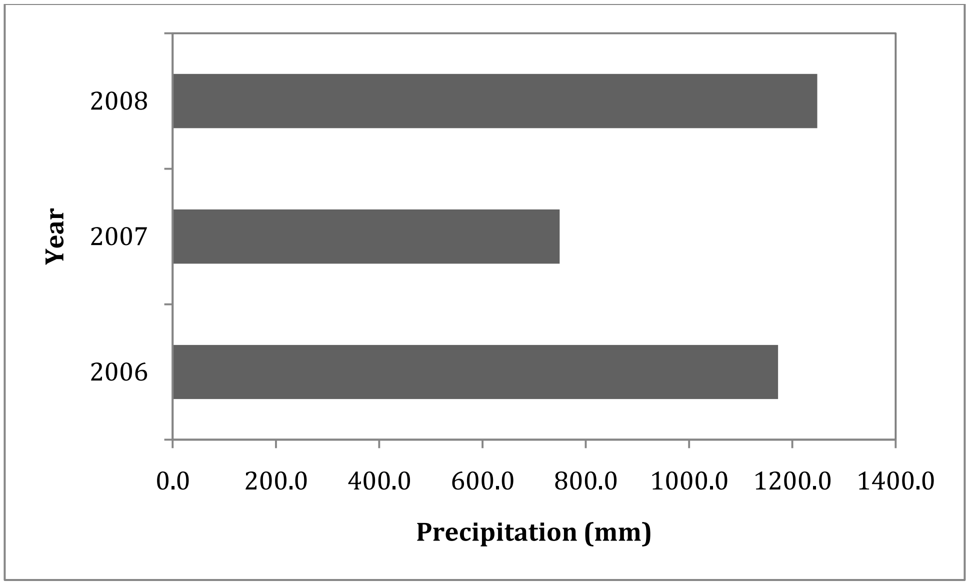

Figure 8 – Total annual precipitation from the Delhi CS weather station located 40 km from Lake Erie Farms.

The abundance of *Digitaria sanguinalis* (L.) Scop. significantly decreased across all treatments to zero above-ground presence in 2008 after having a mean percent cover of 3.3 % in 2007 and a mean percent cover of 54.0 % in 2006. Species such as *Digitaria sanguinalis* are generalist species that easily establish because they have fewer survival requirements (Pickett & Bazzaz, 1978; Pickett, 1982). They are easily outcompeted, however, explaining why this species went from covering over half of the quadrats to having no presence at all (Pickett & Bazzaz, 1978; Pickett, 1982).

The abundance of *Zea mays* L. was the highest in 2006, one year after the restoration treatments were conducted, with a mean percent cover of 16.7 %, then decreased drastically to a zero mean percent cover in both 2007 and 2008 across all treatments. This is likely because it was left over from the farming practices at Lake Erie Farms and is not a colonizing species. *Apera spica-venti* (L.) P. Beauv. (2.9 %), *Chenopodium album* L. (33.2 %), and *Lepidium campestre* (L.) W.T. Aiton (9.4 %) had their highest abundances in 2007 across all treatments, while the abundance of *Ambrosa trifida* L. (3.8 %), *Cerastium arvense* L. (44.8 %) and *Rhus typhina* L. (13.9 %) were the highest in 2008 across all treatments (Table 12). Of these species, only *Rhus typhina* is a later successional species. This species is likely present because it was planted during the sculpted seeding and is a good colonizing species (Foster & Gross, 1999).

Many of the early successional and/or weedy species found at Lake Erie Farms are likely to be out-competed (Pickett & Bazzaz, 1978; Putman, 1994; Kosola & Gross, 1999; Pywell et al., 2003; Prach et al., 2007). Most of the existing species exhibiting significant differences in abundance between the treatments are typically found on newly disturbed lands, lands with low resource availability, and high light intensity (Boutin & Harper, 1991; Erneberg, 1999; Greiling & Kichanan, 2002). As more and more species establish themselves, light levels will become less intense at ground level, resource availability will increase as plants decompose, soil microbes will establish and make nutrients more available to vegetation, and soil moisture will diverge between the restoration units causing a divergence in species that favour certain moisture levels: all of these mechanisms will advance the successional trajectory at Lake Erie Farms (Pickett et al., 1987; Putman, 1994; Bartha et al., 2003; Blumenthal et al., 2003; Greipsson & DiTommaso, 2006; McGill et al., 2006; Flinn & Marks, 2007; MacDougall & Turkington, 2007).

Table 12 - Significant responses of species abundance across all the quadrats combined with a RMANOVA when each species was compared by treatment, time and the effect of treatment over time. (* = p<0.05, ** = p<0.01, *** = p<0.001)

	Treatment			Time			Treatment x Time		
Species	**MS**	**F**	**P**	**Pillai**	**F**	**P**	**Pillai**	**F**	**P**
***Anthemis cotula* L.**	8.64	4.13	*	0.37	12.56	**	0.41	14.71	**
***Apera spica-venti* (L.) P. Beauv.**	1.27	0.92	0.431	0.57	11.16	**	0.12	0.09	0.512
***Chenopodium album* L.**	1.54	1.08	0.302	0.72	39.06	***	0.14	0.10	0.445
***Digitaria sanguinalis* (L.) Scop.**	1.99	1.67	0.224	0.54	10.85	**	0.44	17.90	**
***Lepidium campestre* (L.) W.T. Aiton**	2.04	1.81	0.202	0.31	7.54	*	0.15	0.10	0.441
***Zea mays* L.**	1.13	0.66	0.617	0.59	19.56	***	0.14	0.09	0.446
***Ambrosia trifida* L.**	1.15	0.69	0.595	0.25	5.97	*	0.11	0.08	0.571
***Cerastium arvense* L.**	2.51	1.99	0.183	0.66	31.57	***	0.08	0.04	0.667
***Rhus typhina* L.**	2.09	1.97	0.197	0.60	20.14	***	0.13	0.08	0.442
***Salix bebbiana* Sarg.**	8.91	4.47	*	0.35	12.03	**	0.38	12.85	**

Table 13 - Mean percent cover of the three most dominant species per year at Lake Erie Farms

Year	Species	Mean
2006	*Digitaria sanguinalis* (L.) Scop.	54.0%
	Zea mays L.	16.7%
	Chenopodium album L.	9.4%
2007	*Chenopodium album* L.	33.2%
	Conyza canadensis (L.) Cronquist	16.8%
	Lepidium campestre (L.) W.T. Aiton	9.4%
2008	*Cerastium arvense* L.	44.8%
	Rhus typhina L.	13.9%
	Verbascum thapsus L.	6.1%

4.2.1 Species Composition post-restoration at Lake Erie Farms Compared to Early Successional Species in the Literature

Two published studies can be compared to Lake Erie Farms in terms of species that should be present had the Nature Conservancy of Canada passively restored Lake Erie Farms (shown in Table 12 and Table 13). These comparisons will help demonstrate the effects of sculptured seeding as a restoration treatment. These comparisons show the species present during different successional stages post-agriculture without restoration treatments in the Buell-Small Successional Study, and an early successional species list generated by expert botanists and ecologists from Queen's University (Pickett, 1982; Blatt et al., 2005). There are only two species that are the same from the dominant species lists of Lake Erie Farms and Buell-Small Successional Study: *Ambrosia artemisiifolia* and *Digitaria sanguinalis*, which were both dominant in the first year post-agriculture at both sites (Table 14). According to Pickett (1978; 1982) these species are important to early population dynamics in that they are both generalists. They are also easily outcompeted by native species (Pickett & Bazzaz, 1978; Pickett, 1982). There are many similar species at Lake Erie Farms compared to the early succession species lists created by Queen's University, which suggests that the experts at Queen's chose species that are general to both their study area and to the Carolinian Canada ecozone (i.e. most are common agricultural weeds) (Table 15) (Blatt et al., 2005). It is of interest that there are species present in the first three years post-restoration at Lake Erie Farms that may not naturally appear for at least another 7-17 years because of the sculptured seeding, demonstrating again that this treatment is showing promise for restorations in the Carolinian Canada ecozone (Table 14).

Table 14 - Lake Erie Farms and the Buell-Small Successional Study's dominant species according to the total percent cover. * indicates most dominant species that year. Species found in both Lake Erie Farms within the Buell-Small Successional Study are highlighted.

LEF Dominant Species	Year	BSS Dominant Species	Year
Ambrosia artemisiifolia L.	1	*Ambrosia artemisiifolia* L.	1
Anthemis cotula L.	1	*Mollugo verticillata* L.	1
Chenopodium album L.	1	*Digitaria sanguinalis* (L.) Scop.	1
Digitaria sanguinalis (L.) Scop.*	1	*Barbarea vulgaris* W.T. Aiton	2
Zea mays L.	1	*Erigeron canadensis* (L.) Cronquist	2
Anthemis cotula L.	2	*Erigeron annuus* (L.) Pers.	3
Arenaria serpyllifolia L.	2	*Plantago lanceolata* L.	3
Chenopodium album L.*	2	*Plantago rugellii* Decne.	2-3
Conyza canadensis (L.) Cronquist	2	*Oxalis stricta* L.	3
Lepidium L. *spp*	2	*Rumex acetosella* L.	5
Ambrosia trifida L.	3	*Daucus carota* L.	5
Cerastium arvense L.*	3	*Aster* L. spp	7
Panicum miliaceum L.	3	*Chrysanthemum leucanthemum* L.	8
Rhus typhina L.	3	*Hieracium pratense* Tausch	10
Verbascum thapsus L.	3	*Hieracium florentinum* All.	12
		Lepidium campestre (L.) W.T. Aiton	10
		Trifolium pratense L.	11
		Convolvulus sepium (L.) R. Br.	13
		Poa pratensis L.	15
		Agrostis alba L.	12
		Rhus glabra L.	19-20
		Lonicera japonica Thunb.	17
		Juniperus virginiana L.	19
		Acer rubrum L.	20
		Poa compressa L.	17
		Acer negundo L.	18
		Solidago graminifolia (L.) Nutt.	19
		Rhus radicans L.	20
		Rosa multiflora Thunb.	20
		Solidago juncea Aiton	19

Table 15 - Early successional species list created by an expert panel of botanists and ecologists from Queen's University (Blatt et al., 2005). Species highlighted were found at Lake Erie Farms within the first three years post-restoration.

Early Successional Species		
Elymus repens (L.) Gould	*Medicago lupulina* L.	*Poa pratensis* L.
Ambrosia artemisiifolia L.	*Melilotus officionalis* (L.) Lam.	*Polygonum convolvulus* L.
Barbarea vulgaris W.T. Aiton	*Oenothera* L. spp.	*Polygonum persicaria* L.
Cerastium vulgatum L.	*Oxalis stricta* L.	*Rumex acetosella* L.
Daucus carota L.	*Panicum capillare* L.	*Setaria glauca* (L.) P.Beauv.
Echium vulgare L.	*Phleum pratense* L.	*Stellaria media* (L.) Vill.
Erigeron annuus (L.) Pers.	*Plantago lanceolata* L.	*Thlaspi arvense* L.
Lepidium campestre (L.) W.T. Aiton	*Poa annua* L.	*Verbascum Thapsus* L.

4.2.2 Viability of Harvested Seeds

There were no statistically significance responses in the viability of the harvested seeds from the three most dominant weedy species and the three most dominant native species. As with most plants, seed viability in these species usually have high standard deviations because of genetic and environmental variation, causing minimal statistically significant responses (Baskin & Baskin, 1998). Some trends were derived from the mean viability of each of the harvested species. *Veronica arvensis* had the highest mean viability followed by *Monarda fistulosa* and *Conyza canadensis* (Figure 9). *Veronica arvensis* typically has a high viability rate and a high abundance of seeds produced annually, and is therefore often found in the seedbank and above-ground (Boutin & Harper, 1991). The Nature Conservancy of Canada does not have this species in their identification list because *Veronica arvensis* emerges and flowers in early spring, while the identification field work conducted by the Nature Conservancy of Canada occurs in July after the species has died (Boutin & Harper, 1991). This was a limitation for seed harvesting in my study. Many of the species were absent on July 22nd from the quadrats I had identified during my field work on June 5th.

The purpose of assessing seed viability in experimental ecological restoration is to assess whether any delays in directed succession might be related to genetic or environmental limitations on seed production and germination. In this study, neither the dominant native nor dominant exotic species harvested showed any unusually high or low viability.

The results from this experiment provide a good base for further studies on the viability rates of these six species as there is limited information in the literature.

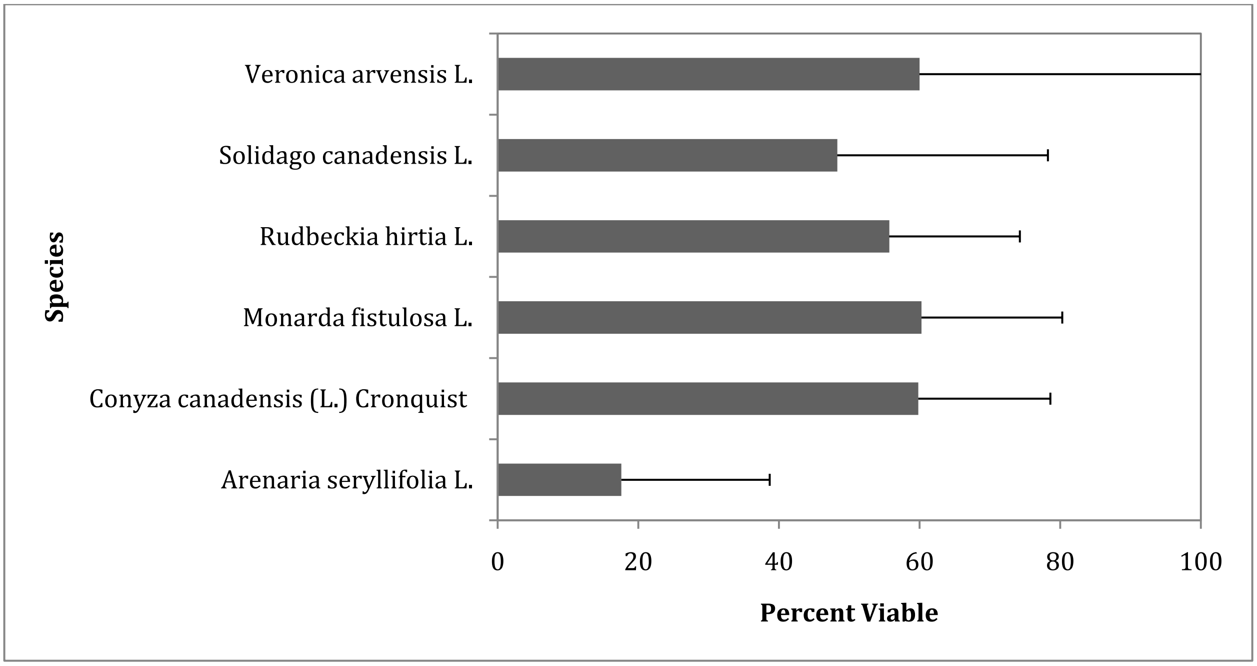

Figure 9 – Mean percent viability of harvested seeds: six species, 3 most dominant native species and 3 most dominant weedy species, were harvested for their seeds in the fall of 2008 and in the spring of 2009.

4.3 Seed Viability and Abundance

4.3.1 Seedbank Abundance

In the brief amount of time elapsed (3 years) since restoration via sculptured seeding of native species, the seedbank has remained dominated by exotic and native weeds as is expected in the early successional stages of sand plains (Pywell et al., 2002; Leicht-Young et al., 2009). The RMANOVA results indicated there were significantly greater numbers of exotic weedy seeds, followed by native weedy seeds. The mean number of native seeds either planted or naturally dispersed into the seedbank did not differ significantly from each other and were both significantly lower than the weedy species (Table 16).

Table 16 - Comparison of the mean seedbank abundance of each species category across all the quadrats using a RMANOVA. Results followed by the same letter are not significantly different according to the RMANOVA with a post-hoc Tukey's test for Honestly Significant Differences. (P> 0.05)

Type of Species	MS	F	Significant Differences
Exotic Weedy	36.2	8.1	A
Native Weedy	18.3	6.3	B
Native non-planted	12.5	8.0	C
Planted	10.0	7.2	C

The analysis of how the treatments are affecting seed abundance suggests that the planted species have not had sufficient time to establish themselves due to the site being in the early stages of restoration (Pywell et al., 2002; Leicht-Young et al., 2009). This is likely because the weedy species dominating the seedbank require fewer resources and can survive harsher environmental conditions (Pickett, 1982). It is promising that there is a presence of planted and naturally dispersed species in the seedbank. As is consistent with Catling & King (2007), Lawson et al. (1999) and Donohue et al. (2000), restoration sites that are close to natural areas are more likely to have higher restoration success. There are several reasons why the locally dispersed native species are beginning to establish at Lake Erie Farms:

- Lake Erie Farms is located next to the protected natural area, the South Walsingham Sand Ridges, which serves as a source population for species that are able to disperse propagules over longer distances
- There are many natural areas surrounding the former agricultural areas on this site
- The former agricultural fields are irregularly shaped, and any point of the field is no more than 200 m from the surrounding natural areas

In time, the planted species should also disperse and outcompete the weedy species for seedbank dominance, which can already be observed above-ground as shown in section 4.2 4.2Species Abundance on page 41.

There were a few species within the seedbank that had significantly high abundances. *Anthemis arvensis* had a significantly higher abundance in the central field than the north and south fields; *Arenaria serpyllifolia* had a significantly higher abundance in the north field than the other two fields, though the central field still had significantly high abundance than the south field; *Digitaria sanguinalis* had a significantly high abundance in the north field; and *Elymus repens* had a significantly high abundance in the oak woodland treatments in comparison to the other treatments (Table 17).

Table 17 - Statistically significant responses of species with higher seedbank abundances compared by treatment and by field with a RMANOVA and a post-hoc Tukey's test for Honestly Significant Differences. (* = p<0.001)**

	Treatment			**Field**		
Species	**MS**	**F**	**P**	**MS**	**F**	**P**
***Anthemis arvensis* L.**	1.13	0.90	0.615	29.27	24.61	***
***Arenaria serpyllifolia* L.**	4.27	1.66	0.118	22.19	18.53	***
***Digitaria sanguinalis* (L.) Scop.**	3.46	2.24	0.208	31.91	26.79	***
***Elymus repens* (L.) Gould**	29.04	26.78	***	5.19	3.27	0.09

When looking at the data from the Buell-Small Successional Study (2004), the seedbank is similar to the expected species that should be present at this stage post-abandonment on an agricultural field, although the above-ground species are somewhat further along the successional progression that occurred at the Buell-Small Successional Study because the Nature Conservancy of Canada actively restored the site with sculpted seeding rather than monitoring an abandoned agricultural field (Pickett, 1982).

4.3.2 Seedbank Viability

There were no significant differences in the seedbank viability at Lake Erie Farms. When looking at the mean percent viability of all the seeds found in the soil samples *Brassica nigra* had the highest mean percent viability followed by *Setaria viridis* (Figure 10).

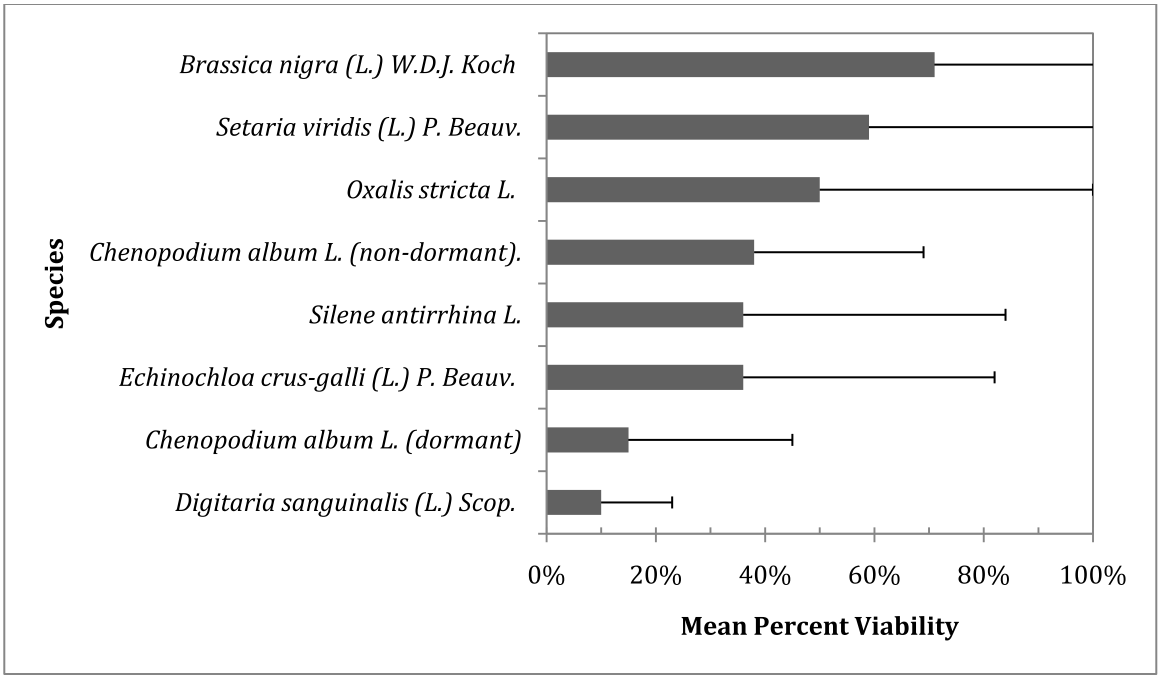

Figure 10 - Mean percent viability of seeds collected from seedbank: The most common seed found was non-dormant *Chenopodium album* (1091 seeds), while the least common seed found was *Brassica nigra* (9 seeds, which was the minimum abundance taken for this graph).

There were many *Chenopodium album* found in the seedbank, both dormant and not dormant. In contrast, there were few emergent individuals of this species. *Chenopodium album* is a wide-spread, highly tolerant plant and its germination is mainly affected by field temperature (i.e. soil & ambient temperatures), as well as light (Bouweester & Karssen, 1993). The lack of above-ground individuals implies that the environment has changed sufficiently, perhaps through the ecological restoration efforts, to alter the soil conditions and light availability so that it no longer favours seed germination, but rather dormancy (Pickett & Bazzaz, 1978; Pickett, 1982; Bouweester & Karssen, 1993; Van der Putten, 2000; Pywell et al., 2002).

The long-term outcome at Lake Erie Farms could help respond to the question that Foster and Timan (2003) posed regarding whether seed additions represent only transient coexistence or if they cause permanent changes to the community composition. At this time, the response is that the community structure above-ground is at a later successional stage than that expected from the literature while the seedbank is indicative of what should be growing at this stage after abandonment with species such as *Chenopodium album*, *Digitaria sanguinalis, Setaria ssp.* and

Oxalis spp. (Pickett, 1982; Blatt et al., 2005). Again, this is likely because of the sculptured seeding treatment at Lake Erie Farms.

4.4 Soil Condition

In June and July the soil moisture was significantly higher in the mesic forest treatment compared to the other treatments, where moisture levels did not differ significantly from each other (Table 18 and Figure 11). In October, the oak woodland and control treatments within the north field were significantly drier (Table 18 and Figure 12-10). That the mesic forest quadrats became wetter than the other treatments over the first three years indicates that the restoration of this area is progressing as desired. The drier oak woodland and even drier sand barren areas are also indicative of this successful progression. This demonstrates the beginning of a divergence of the restoration units into their own environmental conditions, and thus will start developing their own successional trajectories as desired by the Nature Conservancy of Canada.

Table 18 - Statistically significant responses of the soil moisture levels per sampling period compared by treatment and by field nested within the restoration treatments with a RMANOVA and a post-hoc Tukey's test for Honestly Significant Differences. (* = p<0.05, ** = p<0.01)

	Treatment		Field x Treatment	
Sampling Period	**F**	**P**	**F**	**P**
June 2008	11.25	**	1.47	0.378
July 2008	12.17	**	1.93	0.325
Oct 2008	2.43	0.291	7.44	*

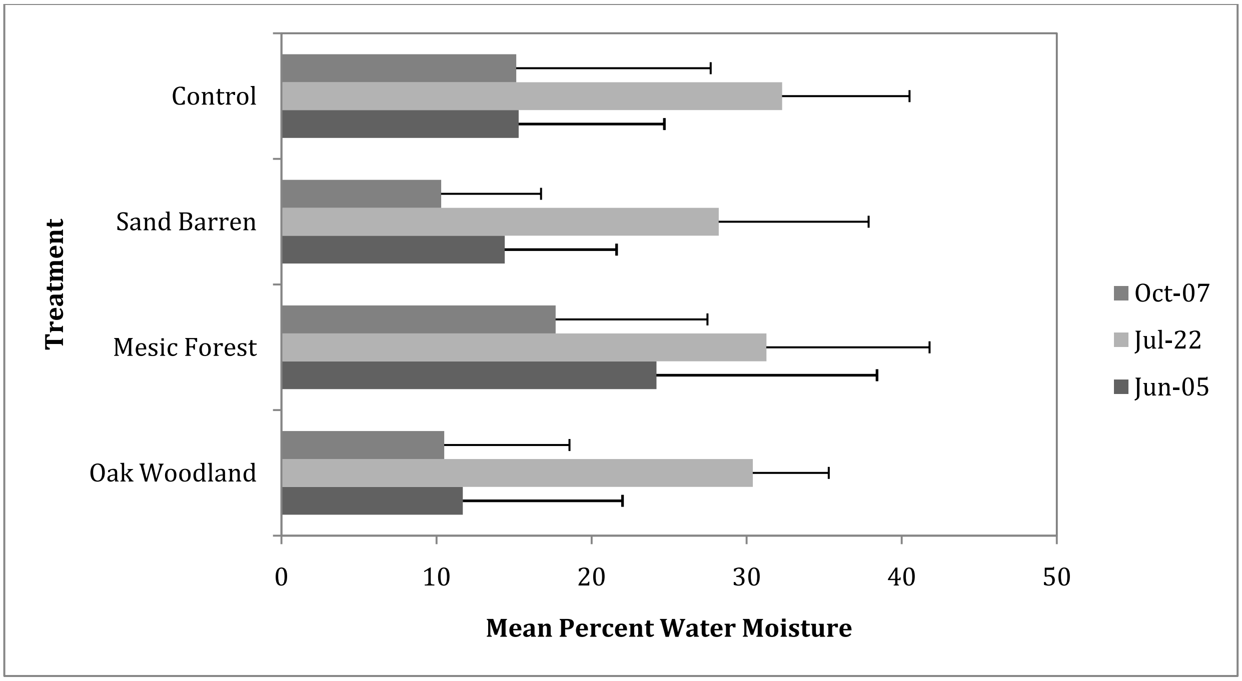

Figure 11 - Mean percent water moisture within the restoration treatments in 2008

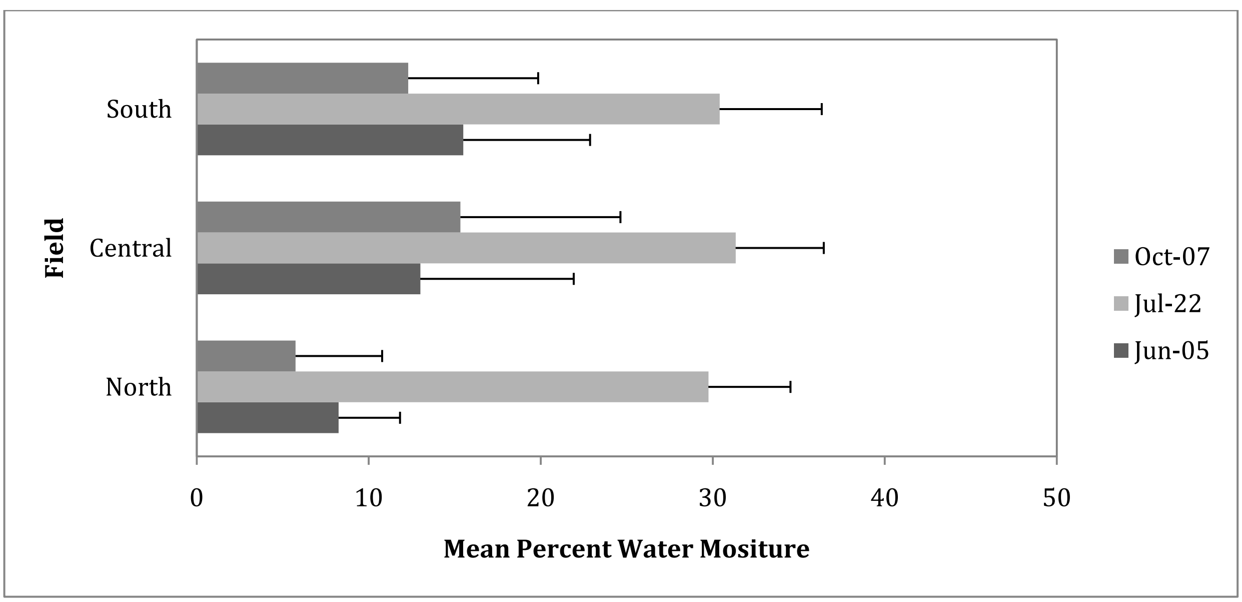

Figure 12 - Mean percent water moisture of the fields nested within the oak woodland treatments in 2008

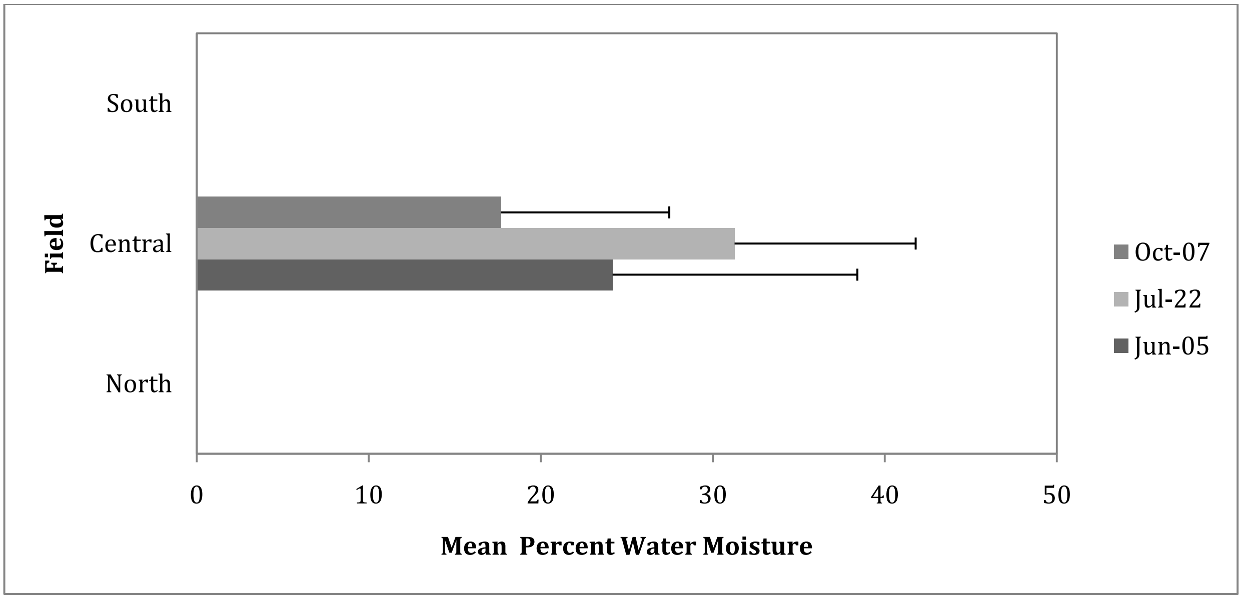

Figure 13 - Mean percent water moisture of fields within the mesic forest treatments in 2008

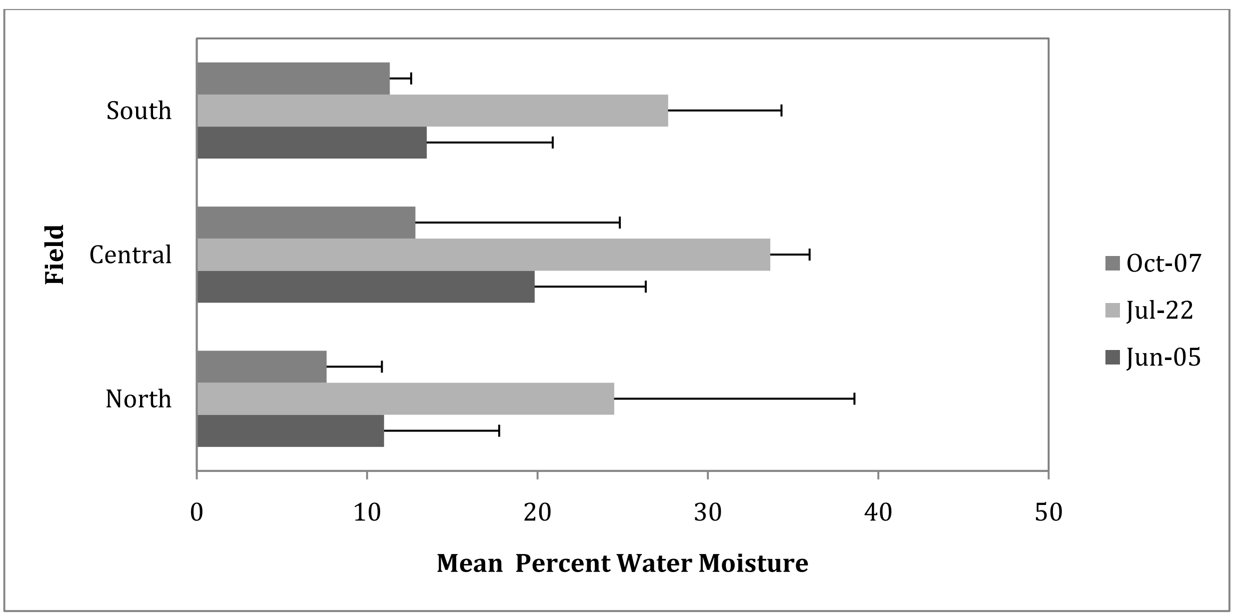

Figure 14 - Mean percent water moisture of fields within the sand barren treatments in 2008

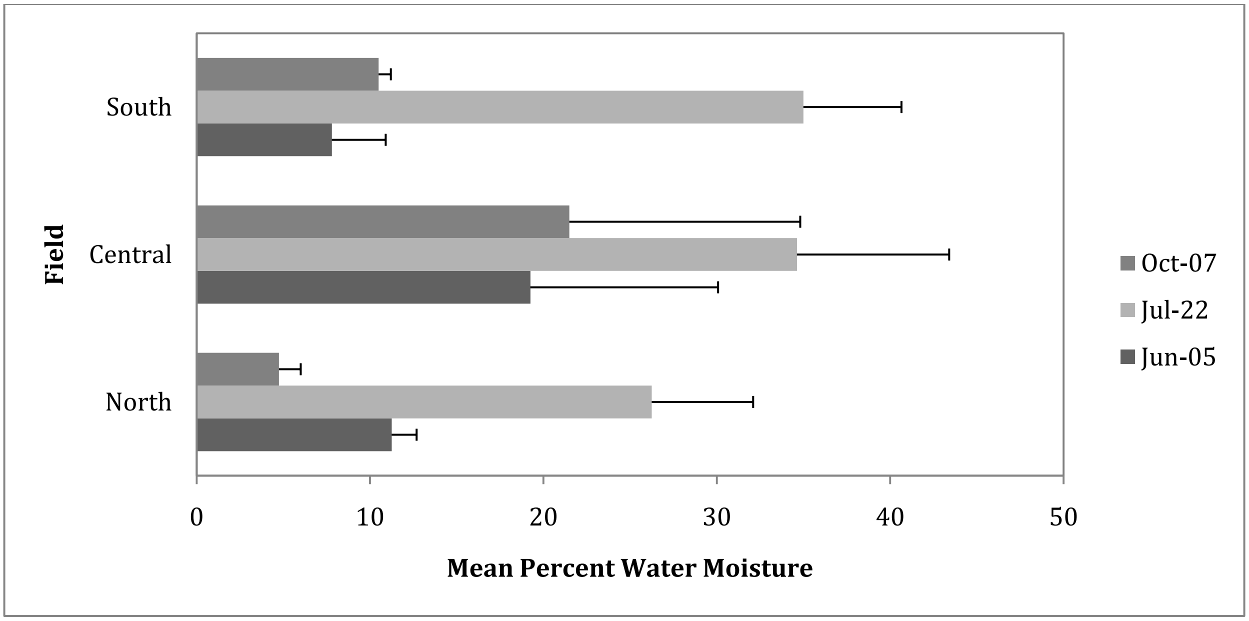

Figure 15 - Mean percent water moisture of fields within the control areas in 2008

There were no statistically significant responses in the soil pH comparisons (see Figure 16 for the mean pH within the treatments, and Figure 17-15 for the mean pH of the fields nested with the restoration treatments). The pH did not vary significantly from season to season in 2008, nor between treatments. Characteristic of sandy soils such as those of Lake Erie Farms, the pH was below the typical pH range for southwestern Ontario, which is 7 or above in most clays or loamy soils (Zwart, 2006; Verhallen, 2009).

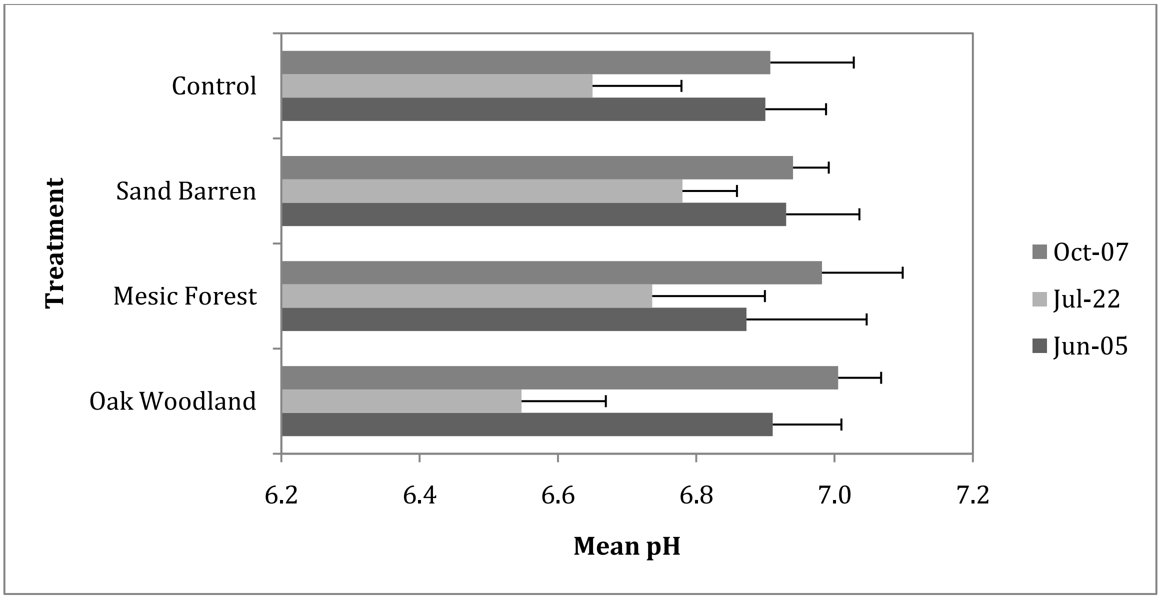

Figure 16 – Mean pH within treatments in 2008

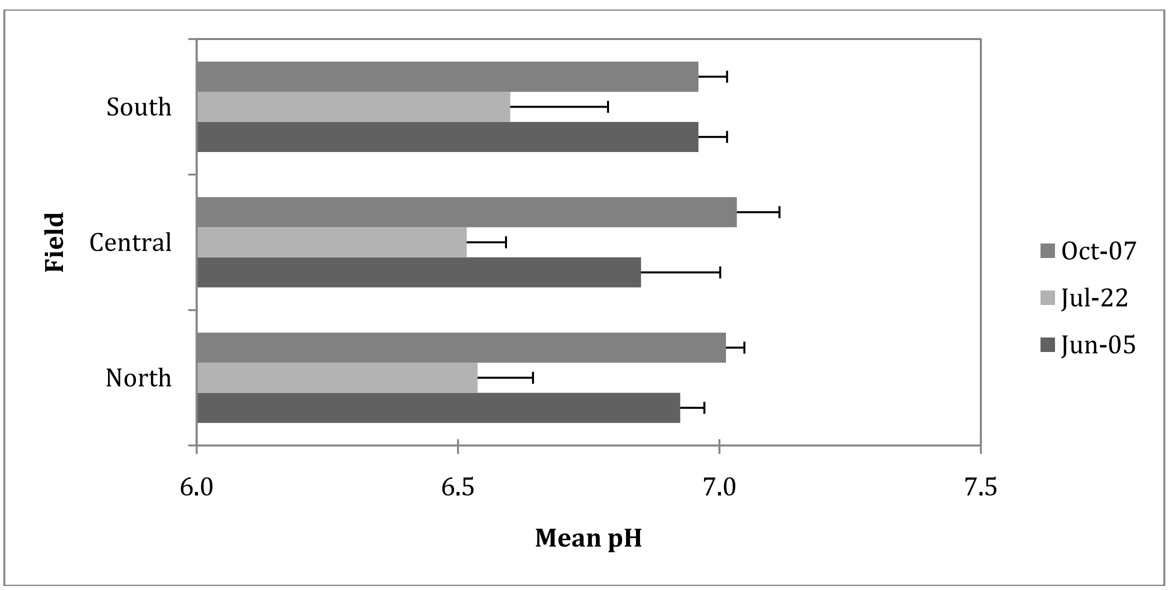

Figure 17 - Mean pH of the fields within the oak woodland treatments in 2008

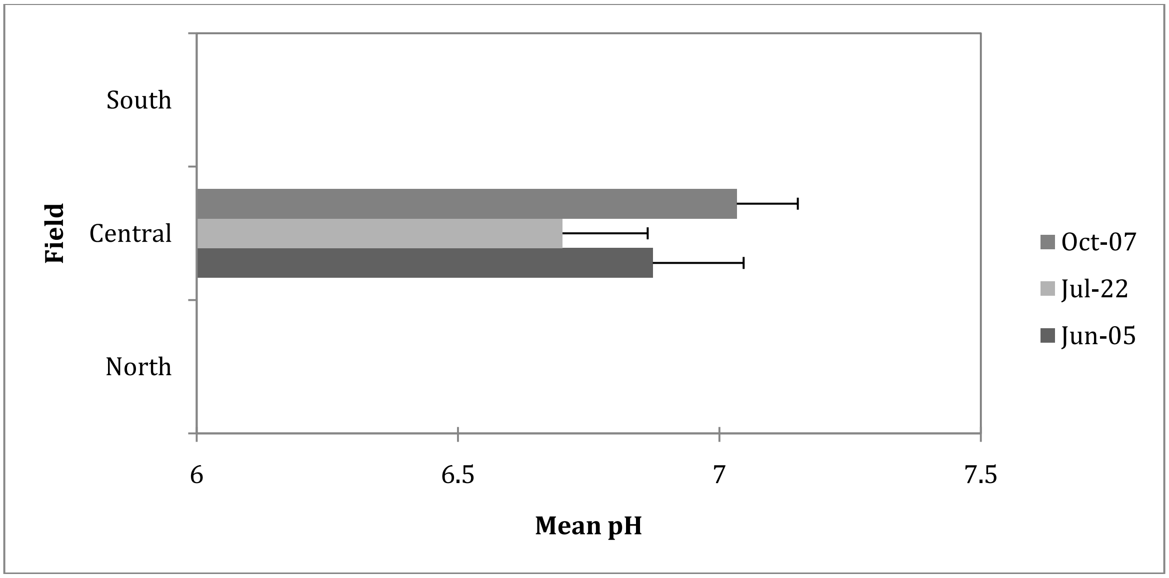

Figure 18 - Mean pH of the fields within the mesic forest treatments in 2008

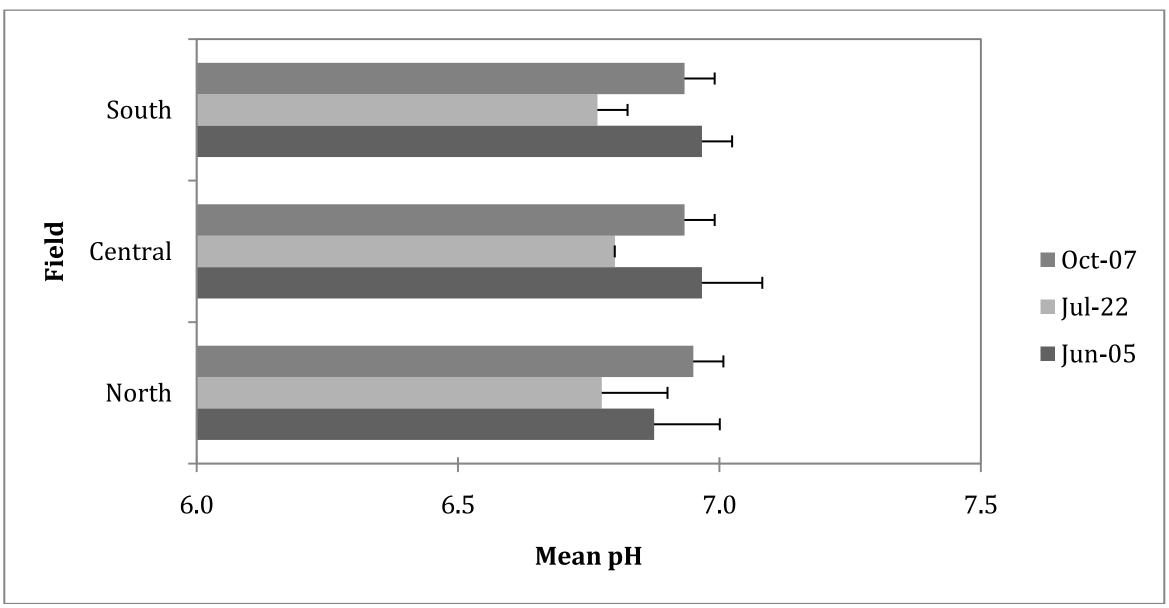

Figure 19 - Mean pH of the Fields within the sand barren treatments in 2008

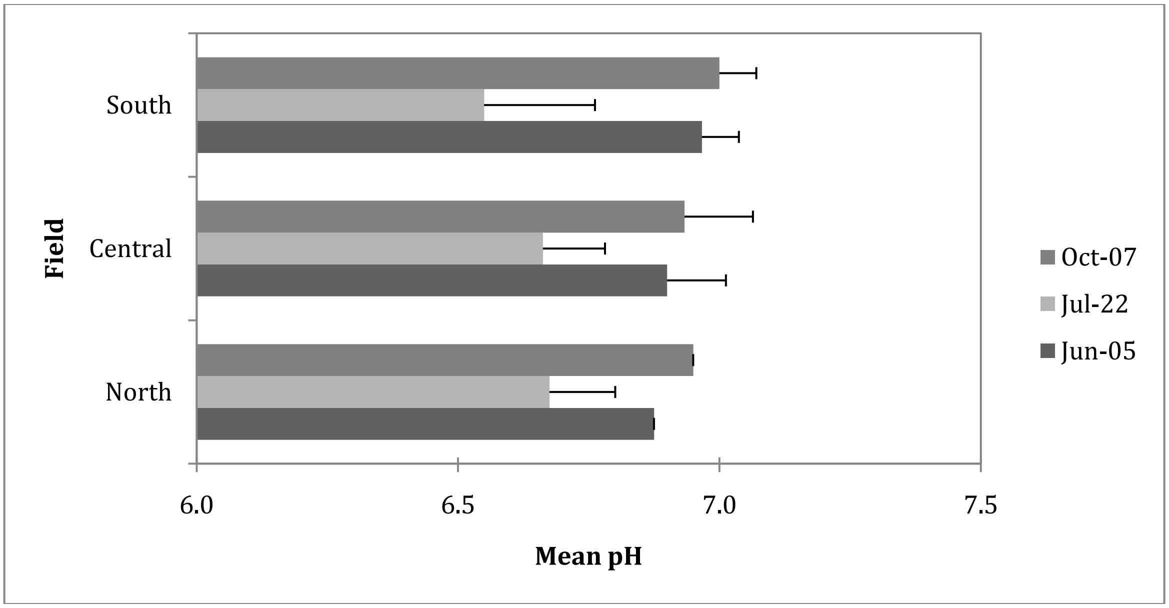

Figure 20 - Mean pH of the fields nested within the control areas in 2008

4.5 General Conclusions

General conclusions that can be derived from this study of population dynamics and community interactions in the third year post-restoration are:

- The restoration treatments are producing results similar to those expected from similar studies in the literature (Van der Putten, 2000; Pywell et al., 2002; MacDougall et al., 2008);
- It is too soon to make any judgements regarding the success or failure of the restoration treatments, as is expected from the literature,
    - The literature shows that the pattern of community dynamics will alter variably during the early successional stages before it stabilizes and becomes more consistent along a clear trajectory unless a major stochastic event occurs (e.g. drought) (Pickett, 1982; Bartha et al., 2003; Pywell et al., 2003; Cadenasso et al., 2006; Prach et al., 2007; Prach & Hobbs, 2008);
- There are differences among the soil conditions of the three restoration units. This could be an indication of these areas starting to separate into their own microhabitats, as desired by the Nature Conservancy of Canada;

- The species present at Lake Erie Farms are mostly indicative of early-successional species composition for similar ecosystems, however some later-successional species are present when compared to similar studies (Pickett, 1982; Blatt et al., 2005), suggesting some temporal advancement resulting from the directed succession (i.e. sculptured seeding),
    - This study is one of the first of its kind in the Carolinian Canada ecozone in terms of studying its early post-restoration temporal scale as well as some of the variables such as seed viability;
- The diversity in the control areas were significantly lower than each of the treated areas in terms of the Shannon Diversity Index for the data collected by the Nature Conservancy of Canada from 2006-2008 (Table 8), though there were no statistically significant differences for the data collected by Katelyn Inlow in 2008
    - Although the power analysis showed 54 quadrats to be a robust enough sample size, the lack of statically significantly outcomes from the data collected by Katelyn Inlow may be due to the fewer quadrats sampled (54 quadrats versus the 163 quadrats sampled by the Nature Conservancy of Canada).

4.6 Recommendations

It would be advisable to compare the findings of Lake Erie Farms with future projects of this nature in the Carolinian Canada ecozone, particularly if it is on the Plainfield Sands. It would also be useful for the advancement of knowledge on how to restore this area to monitoring the variables used in this study in the future to find long-term patterns. These patterns may even suggest early post-restoration indicators of failures that are currently unknown. Making changes to restoration treatments in the early stages is easier and thus more cost effective than making significant alterations to ecosystems once they are established.

More specifically, it would be interesting to investigate whether the seed viability rates found at Lake Erie Farms in 2008 are comparable to other viability rates of the same species. The results from this study could be used as a basis for comparisons with other viability studies on the same species. With this information, irregular viability rates could be used as predictors of the abundances of future generations.

Another recommendation is the addition and monitoring of soil microbes. The soil microbial community establishes symbiotic relationships with plants and provide nutrients to them, which increases the chances for success in establishing desired species compositions (Young et al., 2005; Greipsson & DiTommaso, 2006). Studies on the relationship between restoration success and soil microbe additions may provide missing pieces to our understanding of the mechanisms of community development (Young et al., 2005). Monitoring, and potentially restoring, the nutrient levels at Lake Erie Farms could provide feedback regarding species composition and successional rates (i.e. if there are low nutrient levels, the species composition will not successionally progress), particularly if the former agricultural practices would have decreased the nutrient content (e.g. nitrogen) of the soils at Lake Erie Farms (Compton et al., 1998; Compton & Boone, 2000; Fraterrigo et al., 2009).

Controlled burns are often used to maintain and/or restore oak savannas and sand barrens (Bowles & McBride, 1998; MacDougall & Turkington, 2007; Brudvig & Asbjornsen, 2009; Harrington & Kathol, 2009; Kittelson et al., 2009; Brudvig, 2010). As the trees become established, it may be practical to implement a fire regime into the management strategy to control the development of a closed canopy and mid-storey, shade-tolerant shrubs within the oak woodland restoration units (Bowles & McBride, 1998; Harrington & Kathol, 2009; Kittelson et al., 2009). Alternative methods could include managed grazing or physical removal of undesired mid-story species and girdling some of the over-storey trees to maintain an open canopy in both the oak woodland and sand barren units (Brudvig & Asbjornsen, 2009; Harrington & Kathol, 2009; Kittelson et al., 2009; Brudvig, 2010). The sand barren and the oak savanna are part of the same ecosystem. The sand barren is in the early successional stages while the oak savanna is in the later successional stages of the temporal development of this ecosystem (Leicht-Young et al., 2009). To maintain the sand barren unit, it may be necessary to shape the area with more undulations to mimic the dune-shape, which is less moist because of the mounding of sand and will thus be more barren than the oak woodland unit (Leicht-Young et al., 2009). There could be a risk with this, however, as dunes tend to disturbed by the wind, which could cause the sand barrens to roll over the other restoration units (Leicht-Young et al., 2009). To prevent this from happening, perhaps a balance between the dune-shape and the stabilization of the soil should be to be preserved (Leicht-Young et al., 2009). More research into the benefits of this maintenance strategy should be conducted before implementation. Planting has already occurred at Lake Erie Farms, thus preventing any reshaping of the sand barrens in the future. This could be a recommendation for other projects that are looking to restore sand barrens.

There are some sampling methods that I would recommend the Nature Conservancy of Canada to change. One would be adding a second seasonal abundance and identification count which could show if some species are emerging but possibly failing to reach maturity, or as I found, some species are only visible in early spring. Another would be to add measuring a 1x1 m quadrat within the 2x2 m quadrats. It is difficult to compare the findings at Lake Erie Farms with other sites because of the size difference. I would not stop measuring the 2x2 m quadrats, however, to keep having robust statistical outcomes.

Another indicator that I considered for my study was measuring the Leaf-area index. I decided not to measure this indicator because there is little canopy to measure at this stage post-restoration. Perhaps at later successional stages, the leaf-area index would be an appriopriate measurement to asses incoming radiation and by extension community productivity (i.e. energy cycling) as recommended by Chen et al. (1997) and Bréda (2003). It could also be an indicator of the divergence of the restoration units. For instance, there should be more incoming radiation in the sand barrens than in the mesic forest.

It is also recommended to measure the leaf litter depth, which is indicative of decomposition rates (Borders et al., 2006). This variable measures a direct interaction between the biotic and abiotic factors.

Because of the positive outcomes from the soil moisture analysis, I would recommend that the Nature Conservancy of Canada continue monitoring this in the coming years to see if the trends found by this study continue beyond one year. Soil moisture could be a simple indication of whether the abiotic processes are diverging between the restoration units.

The use of publicly assessable data on all types of ecosystem studies, including monitoring, would be useful for comparative purposes. This database should be simple to add data to as well. This would require, however, a common monitoring practice and recording style, which would require using common, accepted field monitoring, data collection and data entry methods (Halle & Fattorini, 2004). This would also be a useful tool for connecting practictioners and academics, which is typically a problem in most research fields (Clewell & Rieger, 1997).

Finally, it is important that an educational program be implemented to teach owners of former agricultural lands the importance of their properties as ecological refugia in areas that have intensive agricultural and urban pressures on the environment. These land owners need to know what management tools and resources are available to them to ensure they can use a straightforward process to protect or restore their properties.

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Appendix

Maps

Figure 21 - The Carolinian Canada Boundary: The Carolinian Canada ecozone is approximately located below the black line, as indicated by typical species found within this zone. The red star is approximately where LEF is located.

LEF SOIL MAP

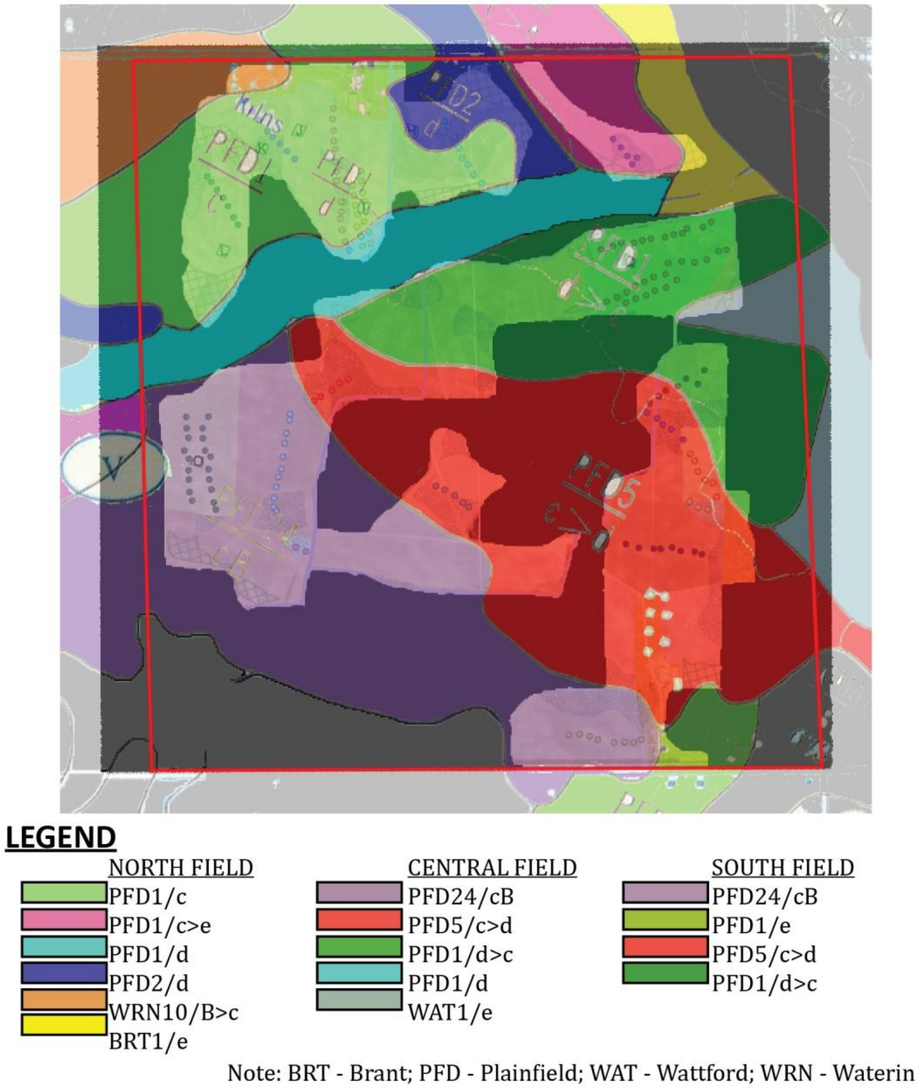

Figure 22 - LEF Soil Map: A Norfolk County Soils Map is overlaying a map of the restoration units and quadrat locations. The darker areas represent the natural areas surrounding the abandoned agricultural fields within LEF.

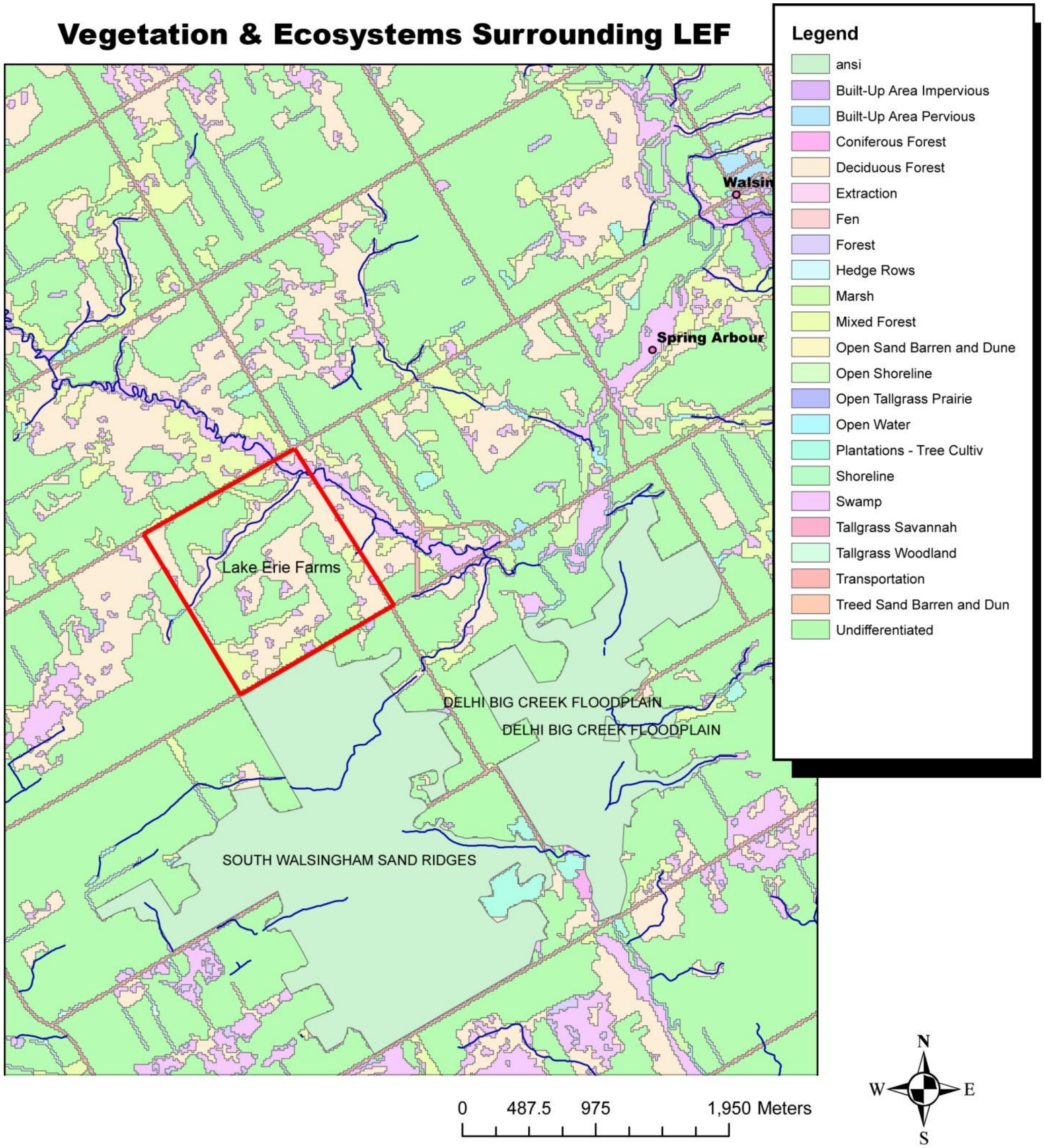

Figure 23 - Vegetation and Ecosystems Surrounding LEF: There are two provincially or regionally significant ecological areas identified by the Ontario Ministry of Natural Resources adjacent and near to LEF: the South Walsingham Sand Ridges and the Delhi Big Creek Floodplain.

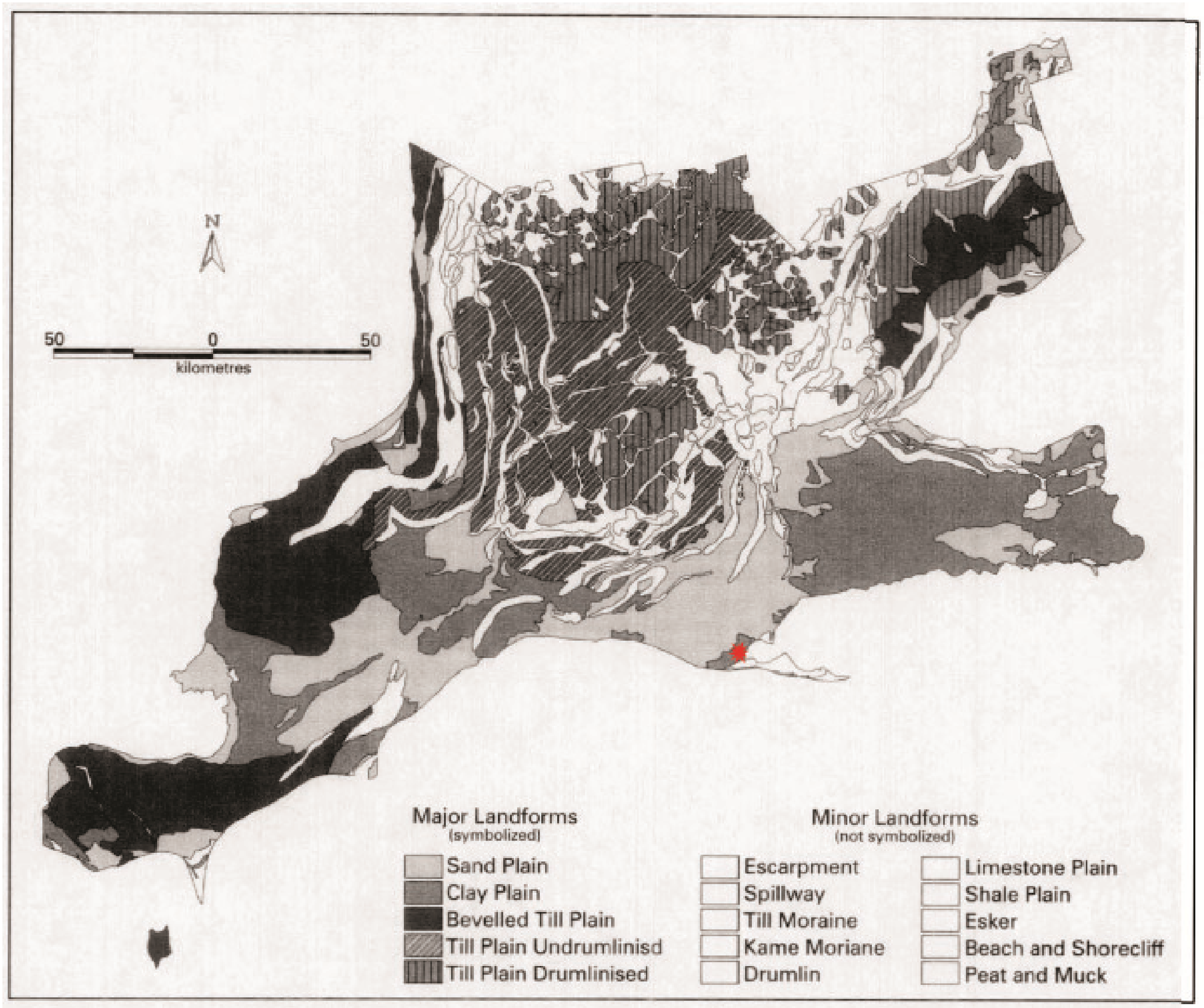

Figure 24 – Landforms of the Carolinian Canada Zone according to Klinkenberg (2002). Only the top five landforms are shown on this map. The red star is the approximate location of Lake Erie Farms.

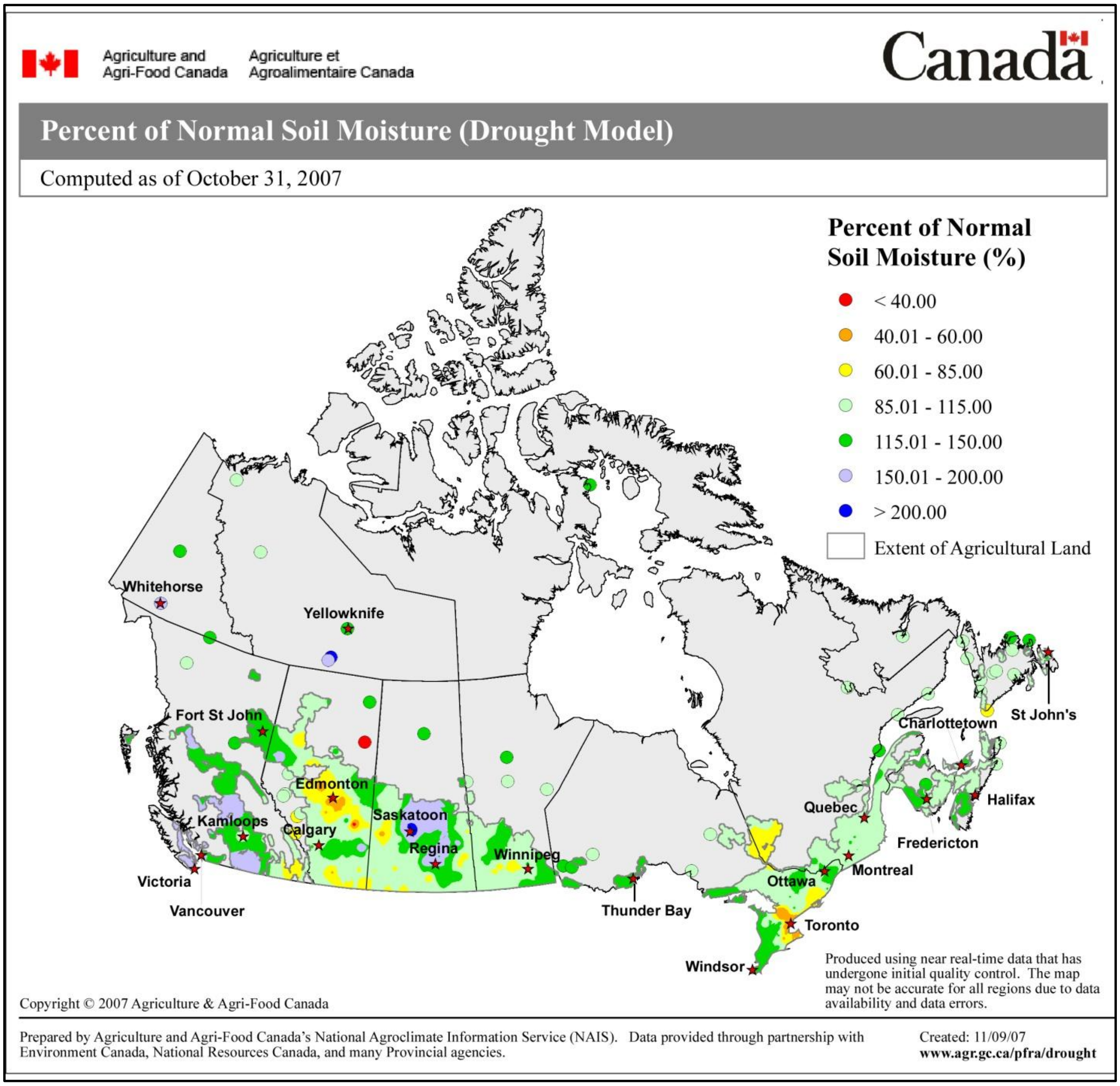

Figure 25 - Percent of Normal Soil Moisture (Drought Model) by Agriculture and Agri-Food Canada (2007).

Figure 26 – LEF's Restoration Plan: shows the restoration units and the quadrat locations within LEF's abandoned agricultural fields. The grey areas represent the natural forested areas within LEF.

Pooled Plant List

LEGEND

KJI Vegetation Identification

KJI Seed Identification

R - Restored Native Species

NN - Native Non-Planted Species

NW - Native Weedy Species

EW - Exotic Weedy Species

Scientific Name	Common Name	Species
Herbaceous Species		**Category**
Amaranthus albus L.	Pigweed	ew
Amaranthus L. *spp.*	Pigweed	ew
Ambrosia artemisiifolia L.	Common ragweed	nw
Ambrosia trifida L.	Great ragweed	nw
Anthemis arvensis L.	Mayweed	ew
Anthemis cotula L.	Stinking chamomile	ew
Apera spica-venti (L.) P. Beauv.	Silky bent grass	ew
Arenaria serpyllifolia L.	Thyme-leaved sandwort	ew
Artemisia campestris L.	Wormwood	r
Artemisia campestris L. ssp. caudata (Michx.) H.M. Hall & Clem.	Tall Wormwood	nn
Asclepias syriaca L.	Common milkweed	r

Asclepias tuberosa L.	Butterfly milkweed	r
Aster L. *spp. 1*	Aster	r
Aster L. *spp. 2*	Aster	r
Berteroa incana (L.) DC.	Hoary alyssum	ew
Brassica L. *spp.*	Mustard	ew
Brassica nigra (L.) W.D.J. Koch	Black mustard	ew
Brickellia eupatorioides (L.) Shinners var. *eupatorioides*	False boneset	nw
Campanula rapunculoides L.	Creeping bellflower	ew
Carex L. *spp.*	Sedge	r
Cenchrus longispinus (Hack.) Fernald	Mat sandbur	nn
Cerastium arvense L.	Chickweed	nn
Cerastium fontanum Baumg. ssp. *vulgare* (Hartm.) Greuter & Burdet	Big chickweed	ew (invasive)
Chenopodium album L.	Lamb's quarter	ew (invasive)
Chenopodium album L. var. *album*	Giant lamb's quarter	ew (invasive)
Cirsium Mill. *spp.*	Thistle	nn
Convolvulus arvensis L.	Field Bindweed	ew
Conyza canadensis (L.) Cronquist	Canadian horseweed	nw
Crepis capillaris (L.) Wallr.	Smooth hawksbeard	ew
Cyperus esculentus L.	Yellow nutsedge	ew
Desmodium canadense (L.) DC.	Showy ticktrefoil	r
Digitaria cognata (Schult.) Pilg.	Fall witchgrass	nn
Digitaria sanguinalis (L.) Scop.	Hairy crabgrass	ew
Echinochloa crus-galli (L.) P. Beauv.	Barnyardgrass	ew (invasive)
Elymus repens (L.) Gould	Quackgrass	ew (invasive)

Elymus trachycaulus (Link) Gould ex Shinners ssp. *trachycaulus*	Slender wheatgrass	r
Equisetum arvense L.	Field horsetail	nw
Equisetum L. *spp*	Fleabane	nn
Erigeron annuus (L.) Pers.	Daisy fleabane	nn
Erigeron hyssopifolius Michx.	Hyssopleaf fleabane	nn
Euphorbia corollata L.	Flowering spurge	r
Eurybia macrophylla (L.) Cass.	Bigleaf aster	nn
Fragaria vesca L.	Wood strawberry	nn
Fragaria virginiana Duchesne	Virginia strawberry	nw
Glycine Willd. *Spp*	Soy bean	ew
Hordeum jubatum L.	Skunk grass	nn
Houstonia L. *spp*	Bluet	nn
Houstonia longifolia Gaertn.	Longleaf summer bluet	nn
Hypericum L. *spp.*	St. Johnswort	nn
Hypericum mutilum L.	Dwarf St. Johnswort	nn
Hypericum perforatum L.	Common St. Johnswort	ew
Hypericum pyramidatum L.	Great St. Johnswort	nn
Lactuca serriola L.	Prickly lettuce	ew
Laportea canadensis (L.) Weddell	Canadian woodnettle	nn
Lepidium campestre (L.) W.T. Aiton	Field pepperweed	ew
Lepidium L. *spp*	Pepperweed	ew
Lespedeza capitata Michx.	Roundhead lespedeza	r
Lespedeza frutescens (L.) Hornem.	Shrubby lespedeza	r
Lespedeza hirta (L.) Hornem.	Hairy lespedeza	r

Lolium persicum Boiss. & Hohen. ex Boiss.	Persian darnel	ew
Lupinus perennis L.	Sundial lupine	r
Medicago lupulina L.	Black medick	ew
Monarda fistulosa L.	Wild beragmot	r
Oenothera biennis L.	Common evening primrose	nn
Oenothera laciniata Hill	Cutleaf evening primrose	ew
Oligoneuron album (Nutt.) G.L. Nesom	Upland white aster	nn
Oxalis stricta L.	Yellow wood sorrel	nw
P unknown 5		
Panicum miliaceum L.	Millet	ew
Parthenocissus Planch. *spp*	Creeper	r
Penstemon digitalis Nutt. ex Sims	Talus slope penstemon	nn
Phytolacca americana L.	American pokeweed	r
Plantago major L.	Common plantain	ew
Polygonum convolvulus L.	Black bindweed	ew
Polygonum lapathifolium L.	Curlytop knotweed	nn
Polygonum pensylvanicum L.	Pennsylvania Smartweed	nw
Polygonum persicaria L.	Lady's Thumb	ew
Portulaca oleracea L.	Little hogweed	ew (invasive)
Potentilla norvegica L.	Rough cinquefoil	nn
Pseudognaphalium obtusifolium (L.) Hilliard & B.L. Burtt ssp. *obtusifolium*	Sweet everlasting	r
Pycnanthemum virginianum (L.) T. Dur. & B.D. Jacks. ex B.L. Rob. & Fernald	Virginia mountainmint	r
Rudbeckia hirta L.	Black eyed Susan	r
Rumex crispus L.	Curly dock	ew (invasive)

Rumex L. *spp*	Dock	ew
Salix bebbiana Sarg.	Bebb willow	nn
Schizachyrium scoparium (Michx.) Nash	Little bluestem	r
Scrophularia marilandica L.	Carpenter's square	nn
Securigera varia (L.) Lassen	Crown Vetch	ew (invasive)
Setaria P. Beauv.	Bristlegrass	ew
Silene antirrhina L.	Sleepy catchfly	nn
Silene vulgaris (Moench) Garcke	Maidenstears	ew
Sisymbrium altissimum L.	Tall tumblemustard	ew
Solanum L. *spp*	Nightshade	ew
Solidago canadensis L.	Canadian goldenrod	nn
Solidago L. *spp*	Goldenrod	r
Sporobolus cryptandrus (Torr.) A. Gray	Sand dropseed	r
Symphyotrichum laeve (L.) A. Löve & D. Löve var. Laeve	Smooth Aster	r
Symphyotrichum lanceolatum (Willd.) G.L. Nesom ssp. *Lanceolatum*	White panicle aster	nn
Symphyotrichum novae-angliae (L.) G.L. Nesom	New england aster	r
Symphyotrichum oolentangiense (Riddell) G.L. Nesom	Skyblue aster	r
Symphyotrichum urophyllum (Lindl.) G.L. Nesom	Arrow-leaved aster	r
Taraxacum officinale F.H. Wigg.	Common dandelion	ew (invasive)
Thlaspi arvense L.	Field pennycress	ew
Trifolium campestre Schreb.	Low hop clover	ew
Triodanis perfoliata (L.) Nieuwl.	Clasping venus' looking glass	nn
unknown		
Unknown dicot		

Unknown grass		
unknown opposite blue funnel flower		
Unknown shrub Photo 2008		
Unknown T8 Sample		
unknwn collected downy taperin lvs		
unkwn ylw asterlike flwrs		
Verbascum thapsus L.	Common mullein	ew
Verbena stricta Vent.	Hoary verbena	nn
Veronica arvensis L.	Corn speedwell	ew
Vicia cracca L.	Bird vetch	ew
Vicia villosa Roth	Winter vetch	ew
vine sp collected		
Viola bicolor Pursh	Field pansy	ew (invasive)
Zea mays L.	Corn	ew
Scientific Name	**Common Name**	**Species**
Woody Species		
Acer × freemanii E. Murray [*rubrum × saccharinum*]	Freeman maple	nn
Acer rubrum L.	Red maple	r
Acer saccharum Marsh.	Sugar maple	nn
Amelanchier arborea (Michx. f.) Fernald	Common serviceberry	r
Betula allegheniensis Britton	Yellow birch	r
Betula papyrifera Marsh.	Paper birch	r
Carya cordiformis (Wangenh.) K. Koch	Bitternut hickory	r

Ceanothus americanus L.	New Jersey tea	r
Corylus americana Walter	American Hazelnut	r
Populus grandidentata Michx.	Bigtooth Aspen	nn
Prunus americana Marsh.	America Plum	r
Prunus serotina Ehrh.	Black cherry	r
Prunus virginiana L.	Choke cherry	r
Quercus alba L.	White oak	r
Quercus macrocarpa Michx.	Bur oak	r
Quercus prinoides Willd.	Dwarf chinquapin oak	r
Quercus rubra L.	Red oak	r
Quercus L. *spp*	Oak	r
Quercus velutina Lam.	Black oak	r
Rhus copallinum L.	Winged sumac	r
Rhus typhina L.	Staghorn sumac	r
Vitis aestivalis Michx.	Summer grape	r
Vitis riparia Michx.	Riverbank grape	r

__Nature Conservancy of Canada's Seed Mix Species List by Restoration Unit__

		Planting rate (kg/ha) **		
Common Name	**Scientific Name**	**Mesic Forest**	**Oak Woodland**	**Sand Barren**
Grass				
Broomsedge	*Andropogon virginicus*		0.01	0.01
Kalm's Brome	*Bromus kalmii*	0.1	0.1	
Sedge	*Carex ciccada*		0.01	0.01
Pennsylvania Sedge	*Carex pennsylvanica*	0.01	0.01	0.01
Sedge	*Cyperus lupulinus*		0.01	0.01
Poverty Grass	*Danthonia spicata*	0.01	0.01	0.01
Panic Grass	*Dicanthelium oligosanthes*		0.01	0.01
Panic Grass	*Dicanthelium spp*		0.01	0.01
Slender Wheatgrass	*Elymus trachycaulus*	0.1	0.1	
Little Bluestem	*Schizachyrium scoparium*	0.1	0.1	
Sand Dropseed	*Sporobolus cryptandrus*		0.01	0.01
Herbaceous				
Spreading Dogbane	*Apocynum androsaemifolium*	0.01	0.01	0.01
Wormwood	*Artemesia campestre*	0.1	0.1	0.1
Common Milkweed	*Asclepias syriaca*	0.1	0.1	0.1
Butterflyweed	*Asclepias tuberosa*	0.05	0.05	0.05
Smooth Aster	*Aster laevis*	0.1	0.1	
Lance-leaved Aster	*Aster lanceolatus*	0.05	0.05	
New England Aster	*Aster novae-angliae*	0.1	0.03	
Sky Blue Aster	*Aster oolentangiensis*	0.1	0.1	0.1
Frost Aster	*Aster pilosus*	0.05		
Flat-topped Aster	*Aster umbellatus*	0.1	0.1	
Arrow-leaved Aster	*Aster urophyllus (sagittifolius)*	0.03	0.03	
Showy Tick-trefoil	*Desmodium canadense*	0.1	0.1	
Panicled Tick-trefoil	*Desmodium paniculatum*	0.02	0.02	
Prostrate Tick-trefoil	*Desmodium rotundifolium*		0.01	0.01
Flowering Spurge	*Euphorbia corollata*		0.1	0.1
Clammy Cudweed	*Gnaphalium mccounii*	0.01	0.01	0.01
Sweet Everlasting	*Gnaphalium obtusifolium*	0.01	0.01	0.01
Long-leaved Bluets	*Hedyotis longifolia*		0.02	0.02
Bicknell's Rock Rose	*Helianthemum bicknellii*		0.02	0.02
Woodland Sunflower	*Helianthus divaricatus*	0.1	0.1	

Intermediate Pinweed	*Lechea intermedia*		0.02	0.02
Pinweed	*Lechea villosa*	0.01	0.01	0.01
Round-headed Bushclover	*Lespedeza capitata*	0.1	0.1	0.1
Hairy Bushclover	*Lespedeza hirta*	0.02	0.02	
Intermediate Bushclover	*Lespedeza intermedia*		0.01	0.01
Puccoon	*Lithospermum canescens*		0.01	0.01
Wild Lupine	*Lupinus perennis*	0.1	0.1	
Wild Bergamot	*Monarda fistulosa*	0.05	0.05	
Virginia Groundcherry	*Physalis virginianus*	0.01	0.01	0.01
Pokeweed	*Phytolacca americana*	0.02	0.02	0.02
Virginia Mountain Mint	*Pycnanthemum virginianum*	0.05	0.05	
Brown-eyed Susan	*Rudbeckia hirta*	0.1	0.1	0.1
Early Goldenrod	*Solidago juncea*	0.02	0.02	0.02
Grey Goldenrod	*Solidago nemoralis*	0.02	0.02	
Venus' Looking Glass	*Specularia perfoliata*	0.01	0.01	0.01
Arrow-leaved Violet	*Viola fimbriatula*		0.01	0.01

Woody Large

Bitternut Hickory	*Carya cordiformis*	3	0.5	
Shagbark Hickory	*Carya ovata*	1		
American Hazel	*Corylus americana*	3	5	1
American Plum	*Prunus americana*	0.5	0.5	
White Oak	*Quercus alba*	10	10	
Swamp White Oak	*Quercus bicolor*	1		
Bur Oak	*Quercus macrocarpa*	0.05		
Dwarf Chinquapin Oak	*Quercus prinoides*	1	1	1
Red Oak	*Quercus rubra*	15	1	
Black Oak	*Quercus velutina*	5	15	0.5

Woody Small

Red Maple	*Acer ruber*	0.02		
Downy Serviceberry	*Amelanchier arborea*	0.005	0.005	
Yellow Birch	*Betula allegheniensis*	0.04		
White Birch	*Betula papyrifera*	0.02		
New Jersey Tea	*Ceanothus americanus*	0.05	0.05	0.05
Alternate-leaved Dogwood	*Cornus alternifolia*	0.05		
Eastern Flowering Dogwood	*Cornus florida*	0.01	0.01	
Waxy-fruited Hawthorn	*Crataegus pruinosa*	0.04	0.04	
American Beech	*Fagus grandifolia*	0.07		
White Ash	*Fraxinus americanus*	0.05		

Witch-hazel	*Hamamelis virginiana*	0.02	0.02	0.02
Red Cedar	*Juniperus virginianus*	0.01	0.01	
Tulip-tree	*Liriodendron tulipifera*	0.05	0.03	
Virginia Creeper	*Parthenocissus quinquefolia*	0.02		
Virginia Creeper	*Parthenocissus vitacea*	0.02	0.01	
White Pine	*Pinus strobus*	0.02	0.01	
Pin Cherry	*Prunus pensylvanica*	0.02	0.02	
Sand Cherry	*Prunus pumila susquehana*			0.05
Black Cherry	*Prunus serotina*	0.05	0.03	
Choke Cherry	*Prunus virginiana*	0.03	0.02	
Wild Crabapple	*Pyrus coronaria*	0.02	0.02	
Winged Sumac	*Rhus copallina*	0.02	0.02	0.01
Staghorn Sumac	*Rhus typhina*	0.05	0.01	
Smooth Rose	*Rosa blanda*	0.01	0.01	
Carolina Rose	*Rosa carolina*	0.01	0.01	0.01
Blackberry	*Rubus allegheniensis*	0.01		
Dwarf Dewberry	*Rubus flagellaris*	0.01	0.01	
Red Elder	*Sambucus pubens*	0.01		
Sassafras	*Sassafras albidum*	0.05	0.05	
Spirea	*Spirea alba*	0.05		
American Basswood	*Tilia americana*	0.06		
Eastern Hemlock	*Tsuga canadensis*	0.01		
Slippery Elm	*Ulmus rubra*	0.01		
Nannyberry	*Viburnum lentago*	0.02		
Maple-leaved Viburnum	*Viburnum acerifolium*	0.01		
Downy Arrowwood	*Viburnum rafinesquianum*	0.01	0.01	
Summer Grape	*Vitis aestivalis*	0.01	0.01	
Riverbank Grape	*Vitis riparius*	0.01	0.01	