Assessing Progress of Urban Ecological Restoration Using Transplants

by

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Author's 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

Forests are the dominant terrestrial ecosystem in Ontario and were once widespread throughout much of eastern North America; in southern Ontario, forest cover has declined from 90% to approximately 17% in the past 200 years. Conversion of the surrounding landscape to urban and agricultural land uses creates edge effects which increase the impact of forest cover loss and threaten ecological integrity. Other impacts on forest ecological integrity include incompatible human activities, a generally inhospitable landscape matrix, and microenvironment changes, including those from chemical pollution and heat island effects. Coupled with direct degradation from human activities like free-riding, the altered forest microenvironment can facilitate increased invasion by opportunistic and competitive plant species, e.g. *Alliaria petiolata* (Brassicaceae). This tends to be worse in urban areas because of greater visitation that creates soil compaction, trampling, and more vectors to introduce the seeds of invasive species.

Active management is often necessary in areas which have been degraded. Ecological restoration represents an important active management tool for urban forest protected area managers. The ecological restoration of the forest understory in altered and threatened habitats represents an important aspect of forest restoration.

For this study, I tested the effectiveness of an ecological restoration using a multi-species assemblage approach with three native understory species to restore the herbaceous layer in Natchez Hills, an Environmentally Sensitive Policy Area in Kitchener, Ontario. The ecological restoration site was degraded and this reduced native plant species in the forest understory. The species selected, *Caulophyllum giganteum* (Farwell) Loconte & W.H. Blackwell (Berberidaceae), *Podophyllum peltatum* Linnaeus (Berberidaceae) and *Maianthemum racemosum* subsp. *racemosum* (Linneaus) Link (Asparagaceae), are widespread throughout eastern North America, and were selected for use in this study because of their persistence and relative abundance in Natchez Hills ESPA. Adult transplants were planted into $4m^2$ quadrats in different densities and shoot populations were monitored for two field seasons to determine if density played a role in early ecological restoration success. The methodology I designed for the ecological restoration was as practical as possible, given the real world financial and logistical constraints faced by many urban forest managers.

Statistical analysis using repeated measures multivariate analysis of variance showed aerial shoot density did not have a significant effect on early restoration success rates ($p > 0.05$). However, there was a significant increase in restored species. *Caulophyllum giganteum* and *Podophyllum peltatum* shoot populations increased by over 97% at the ecological restoration site one year after transplanting. *Maianthemum racemosum* subsp. *racemosum* shoot populations increased by 51% at the site. For comparison, another study in the same forest reported early ecological restoration rates of 92% to 100% for herbaceous understory species, including *C. giganteum* and *P. peltatum*. Other North American studies using transplanted perennial herbaceous material reported average survival rates of 16.6% two years after planting (varying from 1.25% to 51.4% using eight herbaceous species in the Boston area), to 91% one year after restoration (varying from 42.1% to 100% using 19 herbaceous species of the deciduous forest in the Midwest).

In instances where protected areas are degraded, active management and ecological restoration are often necessary tools. This ecological restoration demonstrated that the native herbaceous species used in this study are suitable for use in early ecological restorations in maple beech upland forests in the City of Kitchener. Using a practical and cost-effective methodology this ecological restoration experiment showed that it is possible to implement urban restoration of the forest understory in degraded areas, and to achieve success in early ecological restoration using native herbaceous forest understory species.

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Table of Contents

Author's Declaration ..... **ii**
Abstract ..... **iii**
Acknowledgements ..... **iv**
Table of Contents ..... **v**
List of Figures ..... **vi**
List of Tables ..... **vii**
Chapter 1 Introduction **1**
Background 1
Research Design 2
Chapter 2 Literature Review **5**
Restoration Ecology 6
Community Ecology 10
Monitoring 11
Active Management and Adaptive Ecosystem Management 12
Forest Ecology and Forest Restoration 15
Chapter 3 Methodology **24**
Study Site 24
Suitable Sources and Species for Ecological Restoration 32
Ecological Restoration Experiment Design 36
Ecological Restoration Experiment Methods 43
Monitoring 48
Interpretation of Field Results 48
Statistical Analyses 51
Chapter 4 Results **52**
Results of Field Season One 52
Results of Field Season Two 53
Statistical Analysis of Survivorship Through Each Growing Season 55
Herbivory of Restoration Species 61
Dispersal of Restoration Species 62
Donor Sites 62
Chapter 5 Discussion and Conclusion **64**
Discussion of Statistical Analysis 64
Discussion of Specific Species 67
Comparison of Ecological Restoration Experiment Results to Other Studies 81
Suitability of Restoration Species for Future Ecological Restorations 82
Opportunities for Future Research 82
Broad Scale Landscape Theories and Their Relevance to Ecological Restoration 88
Conclusion 92
References **95**
Appendix Ecological Restoration Planting Listing **107**

List of Figures

Figure 2.1 The relationship between ecological integrity and protected area size 15
Figure 3.1 Location of the City of Kitchener in southern Ontario.....25
Figure 3.2 Environmentally Sensitive Policy Areas in the City of Kitchener: Eastern Portion 27
Figure 3.3 Aerial View of ESPA #24 Natchez Hills 28
Figure 3.4 Photo of invasive species patches in Natchez Hills ESPA.....29
Figure 3.5 Restoration Site Location in Natchez Hills ESPA 32
Figure 3.6 Photo of the study species as naturally occurring in Natchez Hills ESPA.....35
Figure 3.7 Ecological Restoration Experiment Block and Quadrat Layout 37
Figure 3.8 Example of Density Treatments in Quadrats.....39
Figure 3.9 Ecological Restoration Block Locations in Natchez Hills ESPA 44
Figure 3.10 *Podophyllum peltatum* transplant in soil clod 47
Figure 4.1 2007 Shoot Population Comparison.....52
Figure 4.2 2008 Shoot Population Comparison.....54
Figure 4.3 a-c Shoot Population Survivorship by Species in 2007 and 2008.....56
Figure 4.4 a-b Shoot Population Survivorship by Block in 2007 and 2008 58
Figure 4.5 a-b Shoot Population Survivorship by Density in 2007 and 2008 60
Figure 5.1 Photo of *Podophyllum peltatum* 69
Figure 5.2 Dispersal of *Podophyllum peltatum* in 2008 72
Figure 5.3 Photo of *Maianthemum racemosum* subsp. *racemosum* 74
Figure 5.4 Dispersal of *Maianthemum racemosum* subsp. *racemosum* in 2008 77
Figure 5.5 Photo of *Caulophyllum giganteum* 79
Figure 5.6 Browsed *Maianthemum racemosum* subsp. *racemosum* shoot population 85

List of Tables

Table 2.1 Broad Scale Concepts Relevant to Restoration Ecology 5
Table 3.1 Randomly Assigned Treatment for Each Quadrat 40
Table 4.1 Monthly total precipitation 2006 to 2008 & Canadian Climate Normals 1971-2000 54
Table 4.2 Control Quadrat Results.....56
Table 4.3 MANOVAR testing responses of shoot populations in 2007 to block 56
Table 4.4 MANOVAR testing responses of shoot populations in 2008 to block 56
Table 4.5 MANOVAR testing responses of shoot populations in 2007 to density 60
Table 4.6 MANOVAR testing responses of shoot populations in 2008 to density 60
Table 4.7 Donor Site Statistics for *C. giganteum* 63
Table 4.8 Donor Site Statistics for *P. peltatum* 64
Table 4.9 Donor Site Statistics for *M. racemosum* subsp. *racemosum*.....64

Chapter 1 Introduction

Background

Many forests in eastern North America represent remnant patches of previously widespread forests; existing forested areas are often surrounded by, or are in close proximity to urban areas. Urban forests represent the natural environment many Canadians interact with most frequently, and provide environmental, economic and social benefits (GRCA 2004). Yet urban forest ecosystems are some of the most highly altered and threatened forest habitats. Relatively little is known about the effects of urbanization on natural communities (Friesen 1998). Despite protected area designations and jurisdictional regulations, the effects of human activities on urban protected areas may require active management strategies, including ecological restoration, to maintain species compositions, ecosystem functions, and ultimately ecological integrity.

In terms of biomass and annual energy fixation, the herbaceous layer of the forest understory may seem to contribute minimally to forest ecosystem processes (Muller 2003; Whigham 2004). Yet the forest understory herbaceous species of the temperate zone deciduous forests represent a wide diversity of species (Whigham 2004), and are crucial components to forest conservation. The herbaceous layer plays a complex role in forest ecosystem nutrient cycling (Muller 2003), absorbing nutrients from throughfall and influencing the amount of nutrients returned to the forest floor. The herbaceous layer also absorbs nutrients from forest soils during snowmelt, acting as a "vernal dam" by preventing the excess loss of nutrients from forest ecosystems. These nutrients would otherwise be lost during this time of low biotic activity and high soil moisture movement. Senesced material from the herbaceous understory layer also contributes an important source of nutrients to the forest floor annually, in the form of readily decomposable materials.

These herbaceous forest understory species are declining due to fragmentation of existing forests, habitat destruction, competition from invasive species, and degradation from human activities (McLachlan and Bazely 2003; Whigham 2004; Honnay et al. 2005; Murphy 2005); these effects are often exacerbated in an urban forest. In addition to decreasing local and regional biodiversity, the loss of forest understory species may have widespread implications for the functioning of the forest ecosystem. Despite the importance of herbaceous forest understory species to forest ecosystem processes, there is a lack of information about how to conserve and restore these species, especially in altered and threatened habitats (Whigham 2004). Forest understory restoration represents an important component of forest conservation and forest restoration. An essential area of research is the practical application of restoration ecology theories in the restoration of urban forests and the urban forest understory. Implementing, monitoring and

reporting on progress in early ecological restorations are the first steps in developing useful restoration strategies for the urban forest understory.

Research Design

For this thesis I completed an ecological restoration using three native herbaceous forest understory species in an upland hardwood forest in an urban forest protected area. The urban forest protected area for this ecological restoration experiment is located in the City of Kitchener, Ontario, and has been studied previously at both the local (site specific) and regional (landscape) level. As an Environmentally Sensitive Policy Area (ESPA) in the Regional Municipality of Waterloo, the study area forms part of an unofficial network of natural areas, in the highly urbanized and agricultural landscape of southern Ontario. This protected area also contributes to the natural area corridor and watershed forest cover along the Grand River, a Canadian Heritage River (GRCA 2004).

This ecological restoration experiment was designed to restore an area of the ESPA subjected to human-caused degradation, and to provide initial monitoring of restoration results. The three native forest understory species used in this ecological restoration, *Caulophyllum giganteum* (Berberidaceae), *Podophyllum peltatum* (Berberidaceae) and *Maianthemum racemosum* subsp. *racemosum* (Asparagaceae), occur together naturally within the protected area study site, as well as widely throughout eastern North America. These three species were used to create assemblages of native herbaceous forest understory species in different densities, and to measure the success with which they were restored in the urban forest protected area.

Integral to this study was the use of a practical and replicable methodology for restoring herbaceous forest understory species. The methodology was designed to be cost effective and practical, to increase its value and applicability for other protected and natural areas in southern Ontario. The research undertaken in this study also provides municipal and city staff, and ecological restoration researchers and practitioners with an experimental design that is tractable, easily replicated, and meets the sampling design, monitoring and reporting needs of the City of Kitchener and the Regional Municipality of Waterloo.

Research Questions

The specific research question examined in this study is:
Do different density combinations of three native forest understory species (*Caulophyllum giganteum*, *Podophyllum peltatum* and *Maianthemum racemosum* subsp. *racemosum)* differ in their early restoration

responses in an upland hardwood forest urban protected area in Kitchener, Ontario that has been subject to severe human induced degradation?

Further to this research question, the methodology used in this study set up initial monitoring for the following related subquestion:
Do the three native forest understory species (*Caulophyllum giganteum*, *Podophyllum peltatum* and *Maianthemum racemosum* subsp. *racemosum)* differ in their early restoration responses in an upland hardwood forest urban protected area in Kitchener, Ontario that has been subject to severe human induced degradation?

Research Contributions

The research contributions of this study include:

1) The immediate restoration of the native herbaceous forest understory in an area of human-caused disturbance in an Environmentally Sensitive Policy Area in the City of Kitchener.

2) The monitoring of early restoration progress, to test the effect of density combinations on native herbaceous forest understory ecological restoration success.

3) The monitoring of early restoration progress, to test the suitability of each of the three native herbaceous forest species used in this study in forest understory ecological restoration success.

4) To provide the City of Kitchener with a forest understory ecological restoration design that is both practical and cost-effective, to help guide future restoration activities

5) The transferability of results of this ecological restoration through the use of replications

6) To provide information on an ecological restoration of the forest understory herbaceous layer guided by the theories of restoration ecology, and to contribute to the literature on the field of restoration ecology

Thesis Outline

If restoration ecology has one major challenge, it is that there needs to be more sharing of results of practical and replicable experiments among practitioners. This thesis presents the results of an ecological restoration of the forest understory in an urban forest protected area. I begin with an exploration of the literature from the fields of restoration ecology, community ecology, and forest ecology, while acknowledging broader scale concepts that helped guide research and design, such as

landscape ecology. Through these fields I identify the context for forest understory restoration. I then outline the methodology implemented in this study, for the early ecological restoration experiment in an urban forest protected area previously degraded by human activities. Specific details relating to the case study in Kitchener, Ontario are presented. Following the methodology, I present the study results. The implications of the findings at both a local and larger scale are considered in the subsequent discussion, as well as the generalizability of the study to other areas. Recommendations for future research are summarized in the conclusion.

Chapter 2 Literature Review

Literature Review Overview

The central concept for this thesis was restoration ecology. The ecological restoration experiment focused on a specific part of the forest ecosystem, the forest understory. A review of community ecology, with focus on the herbaceous species of the forest understory helped define the scope of the study. A thorough review of forest ecology in the urban forest also defined the parameters of the site specific study. However, no site specific study could be effectively completed without considering the larger scale factors impacting the site. To gain this perspective and inform the site specific study, theories of landscape ecology and associated broader scale concepts were reviewed. Since this ecological restoration experiment focused specifically on an urban protected area, theories related to protected areas were also reviewed, especially theories relating to protected areas planning within a heterogeneous human altered landscape. A brief description of several of the broad themes that provided the context for this study are outlined in Table 2.1. Further discussion of these topics can be found in Chapter 5.

Table 2.1 Broad Scale Concepts Relevant to Restoration Ecology

Theory	Important Concepts	References
Landscape Ecology	Spatial heterogeneity and ecological processes, and understanding how these elements interact; How we understand natural and human dominated landscapes, including the causes and consequences of spatial heterogeneity	Turner 2005; IALE 2011
Island Biogeography Theory	Smaller islands will support less species richness and smaller populations, islands farther from the mainland will have less species richness; The loss of one species may trigger chain effects to other species	MacArthur and Wilson 1967; Wilcox and Murphy 1985; Daigle and Havinga 1996
Ecosystem Fragmentation	Isolated habitat patches in a landscape can be regarded as terrestrial islands in some aspects, as some species and processes are disconnected through fragmentation; Fragmented natural areas in contrast to the surrounding landscape are called patches and corridors	Daigle and Havinga 1996; SER 2008; Theberge and Theberge 2009
Metapopulation Theory	A subdivided population; Metapopulation persistence depends on connections with other populations and exchange of members or genetic material; Spatial discreteness of metapopulation depends on the degree to which the landscape matrix is inhospitable or limiting	Wilcox and Murphy 1985; McCullough 1996; Theberge and Theberge 2009

Limitations to Island Biogeography Theory as Applied to Habitat Fragmentation	Fragmentation effects are too context specific to be treated as unitary phenomena; Habitat fragmentation should be regarded as distinct from habitat loss; Natural habitats are fragmented to some degree; Evolutionary histories of islands and terrestrial habitats are not readily comparable	Haila 2002
Protected Area Theory and Design	Protected area planning must consider the landscape, regional and ecosystem levels; Good design must consider the protected area size and shape, landscape context, and level of connectivity	Franklin 1993; IUCN 1994; Dearden and Rollins 2009; IUCN 2011
Ecoregional Planning	Protected areas are too small and isolated to continue with hands-off preservation planning model; Conflicting pressures originate within and outside of protected area boundaries; Planning across ecoregional boundaries addresses actual flows of species, ecosystems processes as they occur in the landscape	Woodley 2009
Single Large or Several Small	A single large reserve will support more species than several small reserves with the same total area; Optimal system design will have blocks closer, contiguous blocks, and interconnected blocks	Wilcox and Murphy 1985; Noss et al. 1997; Theberge and Theberge 2009
Creating Landscape Level Linkages	Increased patch area and connectivity are important in a landscape context; Tools to increase landscape linkages are buffer zones, corridors, patches and bioregional planning; Opportunities for dispersal are needed	Noss 1987; SER 2008

Shifting from Conservation to Restoration

Many now recognize that while habitat conservation is the first priority for maintaining species populations, conservation alone is insufficient due to the widespread and varying impacts humans have on landscapes. Conservation must be used as a tool in conjunction with restoration (Drayton and Primack 2000; Environment Canada 2004a). Suding et al. (2004) point to the fact that in the past decade, conservation efforts have been shifting from the preservation and protection of intact systems, to the restoration of degraded system, and Holl et al. (2003, p 491) identify ecological restoration as "among the most expensive and extensive conservation actions worldwide."

Restoration Ecology

Restoration ecology is a relatively new field; it is an "emergent subdiscipline of ecology" that describes the set of scientific practices used to advance the field; this includes hypotheses, experiments, field observations, publications, and so on (Higgs 2005, p 159). The science of restoration ecology still requires much development and testing to contribute to theories about how to restore damaged

ecosystems (Palmer et al. 1997). However, the arguments that demonstrate the importance of this field are compelling. Hobbs and Harris (2001) argue that in the coming century restoration ecology will be one of the most important fields, due to the extent of human-induced change and damage. In their own words, ecosystem repair is "an essential part of our future survival strategy, and this demands that restoration ecology provide effective conceptual and practical tools for this task" (Hobbs and Harris 2001, p 239).

Ecological Restoration

Ecological restoration is the practice of carrying out the theories of restoration ecology. The Society for Ecological Restoration (SER) formally defines ecological restoration as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004, p 3). In the simplest cases, ecological restoration would involve removing or modifying a disturbance, and allowing an independent recovery to occur through ecological processes (SER 2004). This is commonly referred to as passive restoration, and it allows for the natural regeneration of the ecosystem to restore the site. If the situation is more complex, and a certain threshold beyond which the ecosystem will not regenerate itself has been surpassed, then deliberate reintroduction of native species that have been lost and/or the elimination or control of harmful invasive species may be required (SER 2004). This is referred to as active restoration, and it often entails planting and management activities.

Frequently human activities are the direct or indirect cause of the degradation or damage that has occurred to the ecosystem (SER 2004). Determining the historic trajectory of the ecosystem before the disturbance, and returning the ecosystem to that trajectory is often defined as one goal of restoration (SER 2004), although the reference ecosystem need not be a past condition. Current conditions at an unaltered (intact) reference site or anticipated future conditions at the site (especially if climate change is expected to alter local conditions) may also be used to help define restoration goals (White and Walker 1997). A recovered and restored ecosystem is one which contains "sufficient biotic and abiotic resources to continue its development without further assistance or subsidy" (SER 2004, p 3). It will structurally and functionally sustain itself, and will demonstrate resilience to environmental stress and disturbance within a normal range (SER 2004).

Although ecological restoration is often used interchangeably with the term restoration ecology, some authors argue that ecological restoration is a more encompassing term (e.g. Higgs 2005). Ecological restoration includes the science of restoration ecology, but also includes other areas which influence the restoration field and restoration success, including human and natural sciences, politics, technology, and economics. Ecological restoration is an integrated approach, one which aims to garner long-term support and positive legacies, to help ensure the success of comprehensive restoration projects (Higgs 2005).

Ecological Restoration in Urban Areas

The concise definition for ecological restoration does not begin to hint at the complexities inherent in practicing ecological restoration, especially in urban ecology. The Society for Ecological Restoration states "[t]he preeminent objective of ecological restoration is for humans to assist with the self-regeneration of healthy, self-sustaining, and resilient ecosystems that have some degree of landscape connectivity" (SER 2008, p 2). Urban areas, by definition are often so damaged, subject to such continual degradation, and so isolated within the landscape, that ecological restoration in the above sense is not feasible (except for perhaps extreme cases, for example, where the urban area has been abandoned).

It could be argued that much of what constitutes ecological restoration in urban areas (and in fact most places) is either rehabilitation ecology or reconciliation ecology. Ecological rehabilitation has been defined as "a management strategy used to improve a degraded area by partially or fully establishing a self-sustaining ecosystem, but not necessarily the historical ecosystem that once existed" (Quon et al. 2001, p 421). Reconciliation ecology examines how to "modify and diversify anthropogenic habitats so that they harbour a wide variety of wild species" (Rosenzweig 2003, p 201). These definitions, which can be used to create realistic goals for ecological restoration in urban areas, depend on the extent to which we can recreate or salvage a site (Allaby 2004). What is feasible in urban areas is ecological restoration of urban protected areas, and perhaps creating other protected areas that are physically connected to enhance metapopulations and ecosystem function.

In ecological restoration it is improbable that the same long-term events leading up to community composition and structure could ever be recreated, so in many cases the most important features or species are used, to guide efforts. For the purposes of this thesis the term ecological restoration is used, but with the recognition that most urban ecological restorations are in fact ecological rehabilitations or reconciliations.

Linking Ecological Restoration Theory and Practice

Although both the theory and the practice are expanding rapidly, there is often a gap between what ecologists argue is necessary for successful restoration ecology experiments, and how practitioners undertake ecological restoration activities on the ground. Clewell and Rieger (1997, p 350) identify "the mutually satisfactory engagement of practice and theory" as the "ultimate challenge" in restoration ecology. Several authors (e.g. Choi 2004, p 75; Manning et al. 2006, p 487) point to the "ad hoc" approach that is taken in many restoration experiments as problematic, with Choi citing this method as an example of gardening. Bradshaw (1996, p 3) identifies the tendency for "individual groups involved in the practical aspects of restoration to pay little attention to what others are doing, or have done". Lake

(2001, p 111) identifies "[t]he lack of reporting on the progress and outcomes of restoration projects" as a major obstacle to the development of restoration ecology.

Part of the difficulty in carrying out restoration experiments in the field is that some restorations may be inherently messy and multivariate, defying the conditions required for a replicated experimental design (Clewell and Rieger 1997). Restoration practice in any scenario is also "new, multidisciplinary, and complex" (Clewell and Rieger 1997, p 350), and when restoration projects are repeated, inadequate monitoring often means the protocols used cannot be examined for effectiveness.

Many advocate that to advance restoration ecology as a scientific discipline, research approaches must be expanded to include many types of studies. Michener (1997, p 324) cites the need for "long-term studies, large-scale comparative studies, space for time substitutions, modeling, and focused implementation", and the use of a range of analytical tools, including "observational, spatial, and temporal statistics". Practitioners require strong ecological foundations to design and implement restoration projects, and ecologists need well designed restoration experiments to obtain useful results and to address complex ecological dynamics (Clewell and Rieger 1997). Linking collaboration between practitioners and ecologists could address both short-term management questions and more broad ecological questions. Increased collaboration could also help determine if restoration approaches are suitable to expand to broader restoration applications (e.g. other ecosystems). This collaboration would benefit both the science and management areas of restoration ecology (Michener 1997).

Restoration Goals

Restoration ecology has a diverse lineage that draws from the theories of conservation biology and geography; practical applications in everything from wetland restoration to mitigating the impacts of resource extraction have fallen under the umbrella of restoration ecology. These varied roots and applications have led to diversity in defining the goals for restoration efforts (Ehrenfeld 2000). Most practitioners agree that restoration goals need to be specified before a project begins. Goals set expectations for the restoration project, guide the plans, and determine the extent to which post-project monitoring will be undertaken.

Restoration approaches and goals are generally defined at the species, ecosystem function, or ecosystem services level; all of these approaches have their advantages and disadvantages (see Ehrenfeld 2000). The general consensus is that restoration projects which recognize ecosystem and landscape-level interactions and processes will be more likely to succeed, and that there are numerous examples of species specific restorations that have failed, due to unforeseen connections between habitat variables and ecosystem or landscape processes (Ehrenfeld 2000). In order to overcome the barriers of site-specific or

species-specific restorations, many authors (e.g. Franklin 1993; Bradshaw 1996; Bell et al. 1997; Ehrenfeld 2000; Holl et al. 2003; Choi 2004; Manning et al. 2006) advocate for restoration projects that take an expanded focus of the larger landscape, rather than viewing restorations as small degraded areas. Goals which are developed appropriately relative to the scope and level of restoration effort are important, as they will help manage expectations for what a restoration can realistically accomplish (Ehrenfeld 2000).

Community Ecology

The changing pattern of community structure, as physical and biological structure change through space and time, is the focus of community ecology (Smith and Smith 1998). Defining the boundaries between communities can be difficult, especially if patterns and species compositions shift gradually. Often it is the similarities or differences between two areas that help to describe the objectives of a community level study (Smith and Smith 1998). While what explicitly constitutes a community has been debated, it may be fairest to state that ecologists arbitrarily set the boundaries of communities at the level of ecological organization in which they are interested (Palmer et al. 1997).

It is difficult to dissociate theories of community ecology from theories of landscape ecology or ecosystem ecology, as these theories are also highly relevant to understanding how patterns and processes at the community level operate (Palmer et al. 1997). However, even though landscape ecology and ecosystem ecology influence community functioning, in some cases it is only practicable to conduct an ecological restoration experiment at the community level. Studies focused at the community ecology level often examine how a number of co-existing species respond to a treatment (Scheiner 2001). Although the study of community ecology has been criticized as "experimental, highly reductionist and intensely local in focus" (Lawton 2000, p 15, cited in Simberloff 2004), others present compelling arguments for studying communities. Simberloff (2004, p 787) states that understanding communities is "crucial to dealing with many key conservation and environmental issues", and that contrary to abandoning this field, it is imperative that we study at this level of organization.

Ecological Restoration at the Community Ecology Level

The field of community ecology provides many opportunities to further the field of restoration ecology. Palmer et al. (1997, p 292) describe the field of community ecological theory as "extremely relevant to restoration ecology because restoration efforts so often involve a focus on multi-species assemblages". Multi-species assemblages must be understood in terms of the interactions of the populations of co-occurring species, but also in terms of the population processes, habitat dynamics, resource dynamics, and disturbances which operate within the community. Additionally, successional

processes related to the roles of community assembly theory, dispersal, and colonization are incredibly significant to restoration efforts (Palmer et al. 1997). In understanding how fragmentation and isolation in the landscape affects species composition, it has been demonstrated that the disposition of one species may affect the survival of others, and single-species population phenomena ignore interspecific interactions and community-level phenomena (Wilcox and Murphy 1985). Restoration efforts which focus at the community level are better suited to inform landscape level decisions than those focused solely on one species.

Ecological restorations similarly contribute to knowledge about community ecology by providing information about links between community function and biodiversity. With a better understanding of which species affect ecosystem functioning, we can increase our knowledge on the relationship between the structure and function of natural systems (Palmer et al. 1997). Additionally, and most importantly, "we may be able to target particular species or functional groups in order to restore a system to a self-sustaining level of functioning" (Palmer et al. 1997, p 292). The mutual benefits to restoration ecology and community ecology from ecological restorations conducted at the community level further demonstrate the importance of study at this level.

Monitoring

Restoration progress can only be evaluated if monitoring is implemented. Monitoring may be necessary to assess "(i) the state of the inputs, (ii) the restoration manipulation, and (iii) the ecological responses" (Lake 2001, p 112). Restoration must be seen as a program in which monitoring is a built-in component, not a discrete event with a beginning and an end, in order to effectively adapt the restoration project and to inform future restoration efforts. As Manning et al. (2006, p 488) state, "there is often no definable static end point for restoration. Long-term planning and funding is needed […]". In the past, most restoration projects have been inadequately monitored, or not monitored at all, providing little opportunity to evaluate the methods used (Clewell and Rieger 1997).

The need for long-term monitoring in addition to initial monitoring cannot be understated. In fact, many of the questions ecologists ask are in need of long-term studies for adequate responses. Long-term in the context of ecological structure and function often means decades to centuries, a timeframe that is rarely supported politically and financially (Michener 1997). However, feedback between species composition and ecosystem processes mean that some ecosystem processes will develop at different times. Therefore, the actual practice of restoration "may involve the setting of sequential, multi-step goals" (Palmer et al. 1997, p 292). An example of the sequence of multi-step goals would be to restore

desired species richness (community structure), monitor the development of community structure, then verify that linkages between community structure and function have been established (Palmer et al. 1997).

For monitoring protocols to be useful there are several considerations that must be taken into account. Associated with any longer term study design and monitoring are "consideration of issues related to data and sampling consistency, quality assurance and data management, and tradeoffs among temporal, spatial and thematic […] resolution" (Michener 1997, p 325). Leading protected areas agencies such as Parks Canada identify a need for monitoring protocols which must be "easily repeatable with different observers, quick and easy (to minimize costs), implemented with low measurement error with the expertise of our staff (for quality control), and be precise "*enough*" to inform management decisions" (Zorn 2006, p 69). Other protected area agencies should similarly identify specific agency needs in their monitoring protocols.

Monitoring is an essential part of assessing the state of restoration projects, protected areas, and ecosystems. Assessing and adapting management programs, are important components of good ecological restoration design. Adapting management programs based on monitoring is part of active and adaptive ecosystem management.

Active Management and Adaptive Ecosystem Management

Active management is the direct involvement of protected area managers in some form of action to maintain or restore ecological integrity, and can be used to sustain or even reintroduce species or ecosystems. Active management can be defined as "any prescribed course of action directed towards maintaining or changing the condition of cultural, physical, or biological resources", and is directed by the protected area agency's objectives (Woodley 2009, p 115). It is used to ensure species or ecosystems which may otherwise be lost are maintained, especially if loss from the protected area would mean a larger scale regional loss. Active management can also be used to repair ecosystem functions which are fluctuating more than the expected variation for that system (Woodley 2009). Ecosystem restoration is a form of active management, and the two terms are often used interchangeably.

Active management is often used in conjunction with adaptive ecosystem management. Adaptive ecosystem management is a process for improving management policies, by continually adapting policies and practices on the basis of new information gained through observing outcomes (Woodley 2009). It is a cyclical and ongoing process, where prescriptive active management plans and actions are designed and implemented, monitored and evaluated, actions are adjusted, and impacts assessed. The concept of adaptive management has been recognized as important to parks and protected areas for decades (Gray 2006). In the complex context of restoration experiments, adaptive management is especially necessary

(Manning et al. 2006). The concept of adaptive ecosystem management reinforces the importance of restoration experiment adaptability, as the restoration site changes in response to the treatments and other variables in the ecosystem. Adaptive ecosystem management provides the feedback tool, derived through inventory, research and monitoring (Paleczny 2006) by which restorations can and should be adapted to changing conditions.

Two concepts used to assess and adapt management and restoration plans are ecological integrity and ecosystem health. Ecological integrity and ecosystem health (also referred to as ecological health) are commonly used interchangeably when referring to the state or "well-being" of a natural system (Daigle and Havinga 1996, p 31). Ecological integrity and ecosystem health provide conceptual measures to assess the integrity and resilience of ecosystems, and to provide management direction for ecological restoration activities. Although the concepts are similar in that they both assess ecosystem status, they are distinct ideas.

Ecosystem Health

Ecosystems are sometimes compared to organisms, and characterizations of nature in terms of health have been applied for centuries. Ecosystem health is a metaphor commonly adopted in both science and policy. It is a useful and politically powerful metaphor, as it inherently helps humans relate to the 'health' or state of the natural environment, evoking personal experiences regarding human health. In an ecosystem context this term also expresses environmental concern, and can even prompt action (Ross et al. 1997). Daigle and Havinga (1996) describe ecological health as a requisite of ecological integrity, along with resilience. They state that ecological health indicates a system's ability to maintain function and ecological processes. Ecological health does not require that all of the parts of the system are present; if the system is able to continue functioning it is ecologically healthy.

Ecological Integrity

Noss (1995) suggests that ecological integrity is a concept which encompasses many related ideas; in his view an ecosystem with integrity also possesses ecosystem health, biodiversity, stability, and sustainability, among other attributes. Karr and Dudley (1981) describe ecological integrity as equivalent to the sum of chemical, physical and biological integrity. The Canada National Parks Act (2000, Section 2) defines ecological integrity for a park as: "a condition that is determined to be characteristic of its natural region and likely to persist, including abiotic components and the composition and abundance of native species and biological communities, rates of change and supporting processes". Ecosystems possessing ecological integrity have their native components and processes intact, an idea strongly related to in-situ conservation (Environment Canada 2006). Ecological integrity requires ecological health, but

also implies that a system is whole, from the level of the gene to the landscape, and implies that all parts are functioning and interconnected, evolving with the abiotic environment (Noss 1995; Daigle and Havinga 1996; Karr 1996).

Karr (1996, p 101) contends that integrity implies correspondence with an original state, one which is complete or undivided, whereas he describes health as a state of "flourishing" or well-being, a healthy organism being one which is capable of performing its vital functions normally and properly. He describes ecological health as the goal for a human dominated landscape (e.g. a cultivated or managed landscape), and one where integrity, in the evolutionary sense, cannot be the goal. Noss (1995) presents a useful example to illustrate the distinction between ecological health and ecological integrity; a tree farm which grows rapidly and accumulates biomass could be classified as ecologically healthy, but since it is certainly absent of some species it does not have integrity[1].

Ecological integrity is a complementary goal to the goal of conservation, through representation of enduring features and biological elements in protected areas, and ecological integrity addresses issues of persistence that representation alone does not deal with (Noss 1995). Developing an appropriate concept of ecological integrity for a highly disturbed urban protected area may require further research, especially given Karr's (1996) assertion that human dominated landscapes simply cannot be managed for ecological integrity. However, since the concept of ecological integrity relates to a system which is whole, is more encompassing than the concept of ecological health, and has been widely used in recent years by protected area agencies, the concept of ecological integrity is used in this study.

Managing Protected Areas for Ecological Integrity

Recent studies suggest that Canada's protected areas network is less effective at conserving biodiversity hotspots and habitats for species at risk, particularly in settled regions in southern Canada (Environment Canada 2006). Ecological integrity identifies the possibility that active management in protected areas may be necessary to maintain or restore ecological integrity (Woodley 2009). The relationship between ecological integrity, protected area size, and intensity of anthropogenic stress has been used by Woodley (1994) to demonstrate how national parks of varying sizes subjected to anthropogenic stress respond in terms of ecological integrity, and how the loss of ecological integrity increases the need for active management (Figure 2.1). This relationship is especially relevant and can be extrapolated to protected areas in southern Ontario, where the landscape is highly fragmented and there is a large human population, increasing the amount of anthropogenic stress exerted on protected areas.

[1] For further discussion on how ecological health is a necessary component of ecological integrity, but not alone sufficient see Noss (1995), Karr (1996), and Daigle and Havinga (1996).

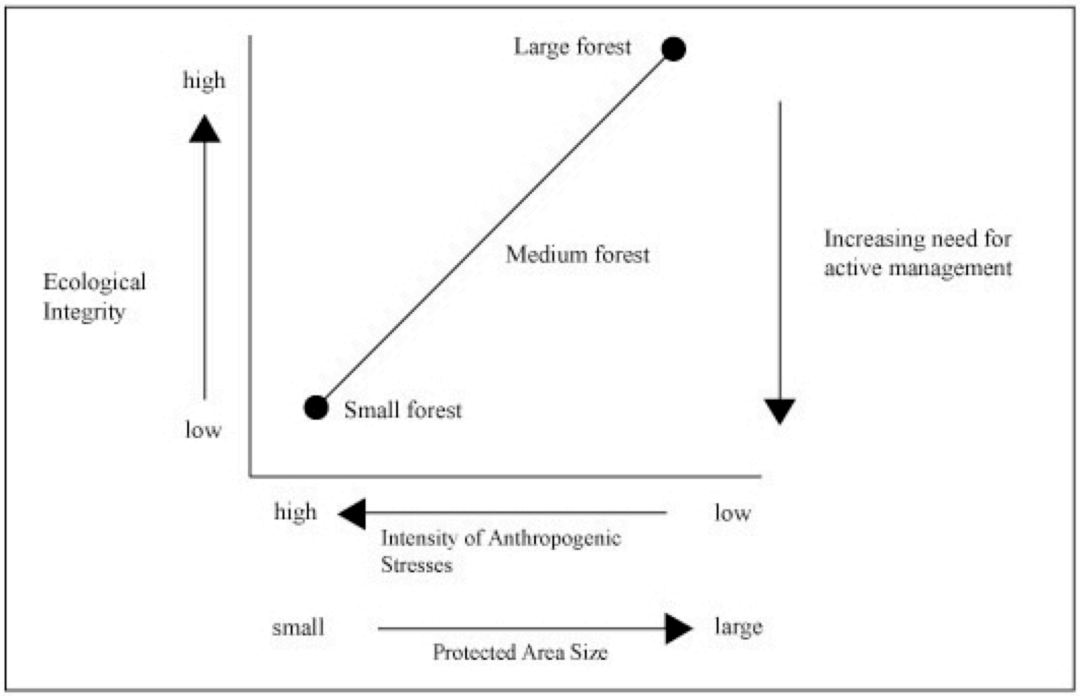

Figure 2.1 The relationship of ecological integrity to protected areas management.
Source: Adapted from Woodley 1994

Forest Ecology and Forest Restoration

The Current State of Forests

Worldwide, forests cover approximately 40 million km^2 (Natural Resources Canada 2008). Forests provide a variety of services, including acting as places for recreation, providing protection for groundwater, and providing habitat to sustain biodiversity (Hüttl et al. 2000). For thousands of years forests have provided humans with food, habitat, and essential services such as clean water and maintaining the atmospheric balance of O_2 and CO_2 (Wickham et al. 1999; Gilliam and Roberts 2003a). However, the amount, pattern, and composition of global vegetation has been altered dramatically by human activities (Riitters et al. 2000). A net loss of 7 to 11 million km^2 of forest has occurred over the past 300 years, largely due to timber extraction and agricultural expansion (Foley et al. 2005). The loss of forest area and fragmentation of remaining forests threatens biodiversity, as well as the ecological goods and services derived from forests (Riitters et al. 2000). Even land-use activities which do not necessarily decrease forest area (e.g. road expansion, collection of fuel-wood) can degrade forest ecosystems, and change species composition, stand structure, biomass, and productivity (Foley et al. 2005).

Temperate forests cover more than 20 million km^2 worldwide; the temperate forest includes the boreal conifer forests, the evergreen rain forests, and the mixed deciduous forests of eastern North America (Newton and Featherstone 2005). The temperate zone is the world region most extensively altered by human activities (Franklin 1995), so it is not surprising that the forests of the temperate zone have been highly impacted. The temperate forests of eastern North America were once widely distributed throughout the landscape; in the two centuries following European settlement clearing for agriculture and lumber altered the landscape towards one that was largely agricultural (Roberts and Gilliam 2003). Temperate forests often have slow recovery rates from anthropogenic disturbances (Newton and Featherstone 2005).

In Canada the area covered by forests and other wooded land amounts to 4 million km^2, representing 10% of global forest cover (Natural Resources Canada 2008). In Ontario forests are the dominant terrestrial ecosystem (Daigle and Havinga 1996), and much of southern Ontario (the region south and west of the Canadian Shield) was covered in forests, subject mainly to natural disturbance before widespread human settlement (Elliott 1998). By 1920, 90% of the original woodlands of southern Ontario had been removed (Larson et al. 1999). The remaining forests of southern Ontario have been extensively altered by humans, and urban development has played a large role in changing the character and distribution of these forests. Forest cover in some parts of southern Ontario decreased as low as 2% or 3%, with most of the landscape being dominated by urban and agricultural land-uses (Elliott 1998).

Forest and Forest Understory Ecology

Forests are defined as a plant community with tree canopy cover of over 60% (Lee et al. 1998). Trees make up the largest visual component of many forest ecosystems, as well as the largest contribution to biomass, but forests are comprised of a diversity of plant species, including trees, herbaceous plants and woody shrubs. In many forests this flora forms vegetative layers at several different heights, and makes up an important part of the forest ecosystem, along with mammals, birds, microorganisms, and other fauna (Daigle and Havinga 1996; Gilliam and Roberts 2003b).

The distinct vegetation layers in forest communities are structured vertically according to their adaptation to various light levels. The canopy comprises the dominant tree species and is the highest layer. The canopy shapes the layers beneath it by influencing the microclimate and available amount of sunlight (Daigle and Havinga 1996). The next layers are the understory and the herb layer. Understory is often used to describe the shrubs and seedlings of shade tolerant species growing beneath the canopies of hardwood and mixed forests. This layer may be subdivided into the sub-canopy and the shrub and seedling layer, depending on the type, species richness, and maturity of the community. The herb layer of

the deciduous forest comprises the wildflowers, ferns, grasses and other vascular plants adapted to the dark, cool, moist and sheltered microenvironment of the forest floor (Daigle and Havinga 1996). For the purposes of this study, the herb layer is referred to as part of the forest understory.

In the deciduous forest the litter, humus and topsoil layers form the most biologically diverse and important layer (Daigle and Havinga 1996). The diverse community in this layer, including the microscopic soil bacteria, fungi, nematodes, insects, earthworms, moles and other organisms, break down the organic debris rapidly, cycling material back into the soil and enriching the soil with nutrients and organic compounds. Organisms in the soil also contribute to soil aeration, facilitating water movement and gas exchange, and the mycelium of the soil fungi (e.g. mushrooms, puffballs, etc.) often form mycorrhizal relationships with plants, aiding in water and nutrient uptake. Decomposing organic debris (such as leaves, bark, and dead plants and animals) does not accumulate significantly in the mesic forests of southern Ontario (Daigle and Havinga 1996).

The Herbaceous Layer of the Forest Understory

The diversity of the herbaceous layer is described as "one of the most striking features of deciduous forests in the temperate zone" (Whigham 2004, p 583). The herbaceous layer in deciduous forests of North America accounts for most of the non-tree vascular plant species diversity (Ricketts et al. 1999; Whigham 2004). The herbaceous layer also correlates with the species richness of several taxa including birds, butterflies and mammals, more strongly than any of these taxa correlate with the richness of tree species (Gilliam and Roberts 2003a).

The variety of herbaceous plant species found in the forest understory is a result of the changeable environment on the forest floor, where the varied microclimates and rich soil layer contribute to wide-ranging availability of essential resources for plant growth through space and time. Light, nutrients and moisture are the primary resources necessary for herbaceous plant growth, and which are available in limited supply in the forest understory (Muller 2003). Deciduous forests are particularly changeable in terms of light availability and intensity, which contribute to physiological and developmental plant processes and control photosynthetic rate and respiration, and ultimately growth (Constable et al. 2007). Consequently, herbaceous species of the deciduous forest have evolved considerable physiological and morphological adaptations for the light habitats of the forest understory; these adaptations may be grouped into six common phenological strategies (Neufeld and Young 2003). Spring ephemerals, for example, are shade avoiders, they leaf out before canopy development and their leaves senesce as the canopy closes. Summer-green species leaf out during or after canopy closure and senesce later in the year (usually in late summer or in the fall), these species are termed shade tolerators.

Both guilds play important roles in temperate deciduous forests, and the majority of understory species (69%) are summer-green species (Neufeld and Young 2003).

Contributions of the Herbaceous Layer to the Forest Ecosystem

The contributions of deciduous forest herbs to processes occurring at the ecosystem level are frequently ignored, owing largely to the small contribution of this layer to overall forest biomass (Gilliam and Roberts 2003a). The herbaceous layer in deciduous forests is vitally important to the nutrient cycling that takes place in forest ecosystems, especially in spring when other species are dormant (Anderson 2003; Muller 2003; Murphy 2005). The vernal dam properties of the herbaceous layer retain vital nutrients in the ecosystem during spring run-off, a time when snowmelt may account for up to thirty percent of annual streamflow (Likens et al. 1977, cited in Muller 2003). Significant nutrient loss, due to excess nutrient removal in the large volume of water leaving forest systems, is offset by the uptake and fixation of nutrients by spring ephemeral species. This temporary biotic dam slows the loss of nutrients and thus enhances the internal biogeochemical cycling in forest ecosystems. The herbaceous layer can also significantly modify throughfall characteristics, especially in dense, well-developed understory layers (Muller 2003).

Concentrations of some foliar nutrients are often higher for forest herbs than woody overstory species. For some nutrients, foliar concentrations are consistently two to three times higher in herbaceous species than in woody overstory species. Although foliar nutrient concentrations do not vary considerably from site to site, they do vary through seasons, indicating that herbaceous species consume nutrients seasonally in a manner that supports growth when nutrients are limited in availability (Muller 2003). With increased reliance on alternative nutrient mechanisms to maintain leaf structure (as opposed to structural compounds such as lignin and cellulose), the foliage of herbaceous forest understory species are also expected to decompose more rapidly than other organic materials, returning nutrients to the forest ecosystem. Most decomposition studies for herbaceous species indicate that all herbaceous materials are decomposed within six months of leaf senescence (Muller 2003).

The herbaceous layer plays an important role in maintaining the functional integrity of forest ecosystems (Gilliam and Roberts 2003b). Understanding interactions between vegetation layers in forest ecosystems is an important step in understanding the entire forest community. Managing for biodiversity and conservation of forest ecosystems requires less focus on commercially important tree species, and more focus on non-commercial tree and herbaceous species (Gilliam and Roberts 2003c). Studies which focus on the herbaceous forest understory layer as an independent component of the forest ecosystem, and a component which influences community and ecosystem processes, are far less common than studies

which view forest understory herbs as dependent variables, responding to the availability of light, soil nutrients, and topographic conditions (Muller 2003). However, in recent years awareness of the role of the herbaceous layer in maintaining the functional integrity of forest ecosystems has grown, thanks to research in eastern forests concerning the ecological dynamics of the herbaceous layer (Gilliam and Roberts 2003a, 2003b).

The Impacts of Fragmentation on Forests

The conversion of forests to other land uses, and competition from invasive species has led to species losses and declines in many forests. The size and shape of remnant forest patches affect their ecology greatly. Small forests (those without a 100 to 200 m buffer around the forest interior) are more susceptible to predation, parasitism, desiccation by wind, and have a higher susceptibility to catastrophic events, including human disturbances such as adjacent development (Environment Canada 2004a). Fragmentation effects in forests can also include increased spread of invasive plant species, increased relative length of edges, and associated rapid microenvironment changes which are often unfavourable to native species.

Given the highly urban and agricultural landscape of southern Ontario (Daigle and Havinga 1996; Elliott 1998), and the high degree of fragmentation and persistence of edge effects in existing forests, it is not surprising that the ecological integrity of the existing forests in southern Ontario is threatened. Ecological integrity in forests refers to forests which contain all of the components (both biotic and abiotic) and processes expected for their forest ecosystem type, from the forest floor to the canopy. In the deciduous forests of southern Ontario this includes the understory perennial herbaceous layer.

The Impacts of Fragmentation on the Herbaceous Forest Understory

The loss of overstory vegetation and the associated edge effects of habitat fragmentation detrimentally affect many forest understory species. Many of the eastern deciduous forests present today, are relatively young second-growth stands, often 100 to 150 years old, and markedly different from those that existed before widespread land clearing (Franklin 1995). Yet even with higher light levels than more mature stands, forest herbs are in reduced abundance. The recovery of this layer from past disturbances such as logging, and particularly conversion to agriculture, may take decades to centuries (Roberts and Gilliam 2003; Newton and Featherstone 2005). Disturbance from agriculture affects the herbaceous layer greatly; annual cultivation removes vegetative propagules and seed banks, and the microenvironment is changed dramatically (Singleton 1998; Roberts and Gilliam 2003). This is particularly problematic for many native forest understory species, since they generally exhibit slow growth rates, with a long phase prior to reproduction, a low reproductive output, and low dispersal capabilities (Middleton and Merriam

1983; Jacquemyn et al. 2003). In some forest herbs the dispersal rates are so low that they are still expanding their range in response to glacial retreat (e.g. *Erythronium americanum* in Nova Scotia) (Holland 1980, cited in Middleton and Merriam 1983).

Forest understory herbs are vulnerable from habitat loss, habitat fragmentation, non-native species introductions, secondary effects (e.g. loss of pollinators, browsing by artificially high densities of herbivores), and global change (Jolls 2003). Populations of some understory herbaceous species are threatened from harvesting from the wild for food, for medicinal purposes, or for ornamental garden plantings (Small and Catling 1999). Forest understory herbs are also highly vulnerable to physical destruction due to incompatible forms of recreation, including hiking and riding off-trail and other destructive activities.

In landscapes where forest habitat has been fragmented and other disturbances are present, the persistence of native herbaceous plant populations in small forests may be due to long generation times and prolonged clonal growth. Current distributions may reflect a historical landscape more than the present one, potentially masking an extinction debt (Honnay et al. 2004). Habitat fragmentation reduces the native species richness in the forests of eastern North America, and even increases in forest cover translate to slow recovery of understory herbaceous species (McLachlan and Bazely 2001).

The Urban Forest

Urban forests are defined as the trees that are cultivated and managed in areas influenced and used by the urban population; this definition includes street trees, trees in parks, trees on private property and the trees found throughout watersheds (GRCA 2004). The urban forest is now common, as the urbanization of formerly rural and agricultural land converts the landscape, leaving remnant forests and protected natural areas amidst developed areas. By virtue of their very setting, urban forests are fragmented from the larger forested landscape.

The effects of forest fragmentation in an urban setting share some similarities with fragmentation in non-urban forests, but there are additional impacts in this environment. The edge effects from human activities on suburban forests are greater in severity and spatial extent than natural edge effects (Friesen 1998). Factors that modify the urban forest environment include chemical pollutants such as acid deposition and ozone, as well as physical modifications such as soil compaction, hydrophobic soil conditions, and the urban heat island effect (Watmough et al. 1998). More than most places, the urban forest ecosystem is dynamic and subject to sudden changes like housing developments or new roads that bring more people and pets into the system, along with the cumulative impacts of development, pollution, and the overarching global climate changes.

The changes in environmental conditions in urban areas are not negligible; the "speed, persistence, and ubiquity of anthropogenic habitat alterations may suddenly put even previously well-adapted native species at a competitive disadvantage with non-native species" (Byers 2002, p 449). Compared with the much slower evolutionary responses of most native species, the rapid pace of urban ecosystem change often means that few species are equipped to adapt to this change (Byers 2002). Opportunistic species, (whether native or non-native), can outcompete the complex assemblage of species present in the forest understory, lowering biodiversity and setting the stage for future invasion (e.g. as with *Alliaria petiolata*, an alleopathic invasive perennial herb of the forest understory) (Murphy 2005)[2].

Despite the impacts experienced in the urban forest, urban forests provide areas of habitat and biodiversity. Given that habitat loss resulting from urbanization and agriculture is the most significant threat to species at risk in Canada (Venter et al. 2006), urban forests provide an important refuge in an altered landscape. The urban forest contributes a number of additional benefits, including reduced storm water run-off and improved water quality, energy conservation through the cooling effects of evaporation of water from leaves, noise buffering, increased property values, and improved psychological well-being, among others (GRCA 2004).

The benefits associated with urban forests illustrate that urban forests are indeed important, but they clearly represents a type of protected area which requires modified management strategies. It may be most helpful to think of an urban forest as one type of area species may use, within a complex landscape mosaic of human altered and semi-natural environments. Due to the relatively small size of most urban protected areas, and the increased stress from anthropogenic activities and the intervening landscape matrix, urban forest and protected areas often require increased active management. Ecological restoration is one form of active management that can be used in urban forests and protected areas.

Ecological Restoration in Urban Forest Areas

Increasingly ecological restoration is being implemented in fragmented forests, to halt or reverse the negative effects of fragmentation, and restore ecological integrity. In urban forests degradation from human impacts often intensifies the changes (Mottl et al. 2006), requiring increased active management. Combined with the limited financial and human resources available for urban protected area managers (Strobl 1998; Schmitt 2004; Schmitt and Suffling 2006), the result is a need for pragmatic protected areas management and restoration in forest ecosystems. In southern Ontario, where there are already concerns

[2] See Murphy 2005 for a detailed literature listing on *Alliaria petiolata*.

about forest species diversity and where human population pressures are highest, these needs are especially great (Strobl 1998).

Along with the complexities described and the minimal resources available are additional issues associated with urban forested areas. Restorations are inherently complex, and may involve any number of factors, including the restoration of species' populations, vegetation structure, and ecological processes (Menges 2008). Issues arise regarding where to obtain sufficient locally adapted and provident genotypes and phenotypes to do ecological restoration. The lack of ability to control access (to protected restored sites from people and other animals), or to constantly "care" for the site, provide additional challenges, especially considering that ecological restoration projects must be self-sufficient after a short period of time if they are to be considered successful (Quon et al. 2001; Murphy 2005; Murphy et al. 2007). Restoration of the forest understory may involve consideration of all of these factors, as well as others.

Ecological Restoration in the Forest Understory Community

Previous work in the field of restoration ecology has focussed on a variety of ecosystems and species. While much attention has been directed to the ecological restoration of tree species, less attention has been focused on restoring the native herbaceous forest understory layer. Although these types of ecological restorations are less represented in the literature, they are no less important to advancing the field of forest restoration. The lack of information about forest understory restoration using herbaceous species has not gone unnoticed (Whigham 2004; Mottl et al. 2006). Whigham (2004) notes that although we understand the basic biology of forest understory herbs and are able to make some broad generalities, few species have been studied in detail. Additionally he notes that "[w]e are especially lacking in information needed to conserve and restore species in altered and threatened habitats" (Whigham 2004, p 583). Ferguson (2007) notes the underrepresentation of several areas in both the practice of ecological restoration and the literature on restoration ecology. These include the lack of focus on spatially and temporally small-scale restoration efforts, the lack of use of combinations of vegetation to test for synergisms or antagonisms, and the aforementioned focus of most forest restorations on tree species as opposed to the herbaceous understory vegetation (Ferguson 2007).

Due to time and other constraints, restoration experiments in forest ecosystems are often designed and implemented at the community ecology level, even if the broader landscape level implications are considered (e.g. Murphy 2005; Ferguson 2007). The practical nature of small scale studies in terms of labour and materials means that these studies merit further study and focus in the restoration ecology literature and practice, as Ferguson (2007) argues.

A key component of furthering forest understory restoration is to develop workable methodologies for the species in question (Drayton and Primack 2000). Efforts to restore ecosystem function must take into account the role individual species play in the system, particularly if some species are especially important to ecosystem functioning (Palmer et al. 1997). Understanding the role of various forest understory species can help to define goals for community-based restorations. Testing the effectiveness of potential species at improving the establishment of desired vegetation is one way Clewell and Rieger (1997) advocate for accelerating restoration, and is one example of a community based ecological restoration with a specific goal. Ecological restorations help contribute to knowledge about community ecology by providing information about links between community function and biodiversity. The function of native herbaceous forest understory species in forest ecosystems, and their impact on forest ecological integrity is a key reason why ecological restorations involving this group should be carried out, with focus at the community level.

Chapter 3 Methodology

Methodological Overview and Goals

The study site for this ecological restoration experiment is a remnant upland hardwood forest and urban protected area, where human-caused disturbances have degraded the forest understory layer to a great degree. The goal of the experiment was to test early restoration progress in an urban forest protected area using different densities of native herbaceous species, to restore forest understory species to the site. The potential of early ecological restoration in areas of human-caused disturbance was tested using species of the summer-green perennial guild of forest understory herbs. Ecological restoration projects are experiments where different kinds and numbers of species are introduced, and are experiments in which we can test "the effects of species richness and species roles on community recovery and functioning" (Palmer et al. 1997, p 293). This ecological restoration experiment was designed to contribute to knowledge on early ecological restoration using species assemblages, and to be both academically rigorous and practicable. The experimental design sets the groundwork for the ecological restoration to be monitored long-term, to serve as a long-term ecological model for advising future restoration and management plans in Natchez Hills ESPA, and the City of Kitchener. The practical design of this experiment was carefully considered so that it could easily be recreated, given the financial and logistical realities that constrain many ecological restorations.

Study Site

Location, Climate, Geology and Vegetation

The study site for the ecological restoration is within the City of Kitchener, in southern Ontario, Canada. The City of Kitchener is located in the Regional Municipality of Waterloo, in the Manitoulin-Lake Simcoe Ecoregion of the Mixedwood Plains Terrestrial Ecozone (Ecological Framework of Canada [date unknown]). Warm summers and mild winters characterize this ecoregion; the mean annual temperature is 6°C, with a mean summer temperature of 15.6°C and a mean winter temperature of -4.5°C. The mean annual precipitation is between 750mm and 1000mm, with precipitation evenly distributed throughout the year (Ecological Framework of Canada [date unknown]).

Characteristic climax vegetation includes sugar maple, beech, eastern hemlock, red oak and basswood, with pioneer species of white pine, paper birch and trembling aspen. The ecoregion is in the West St. Lawrence Lowland, where the geology is carbonate-rich Palaeozoic bedrock, and there are a wide variety of deep glacial deposits. The historical soils in the area are predominantly Loamy Melanic Brunisolic and Gray Brown Luvisolic (Ecological Framework of Canada [date unknown]). Agriculture is the most extensive land use in this ecoregion (56%); other land uses include urban development,

recreation and tourism. The Kitchener-Waterloo area is a major community of this ecoregion (Ecological Framework of Canada [date unknown]).

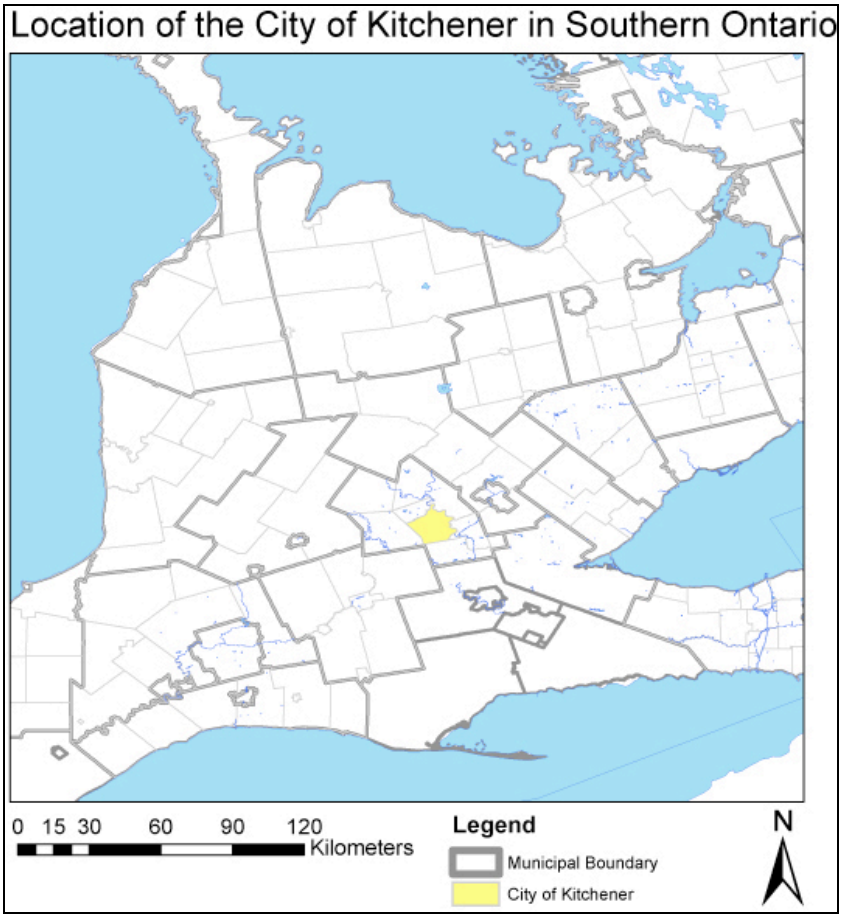

Figure 3.1 Location of the City of Kitchener in southern Ontario. The City of Kitchener is located in the Regional Municipality of Waterloo. Source: Municipal Boundary Data and location of the City of Kitchener from the Ontario Ministry of Natural Resources (2008).

Protected Areas in the Regional Municipality of Waterloo

In the Mixedwood Plains Terrestrial Ecozone only 1.59% of the area is protected (Environment Canada 2010a). This figure highlights the importance of not only maintaining existing protected areas in this ecozone, but also in restoring degraded protected areas. Protection and restoration of natural areas in southern Ontario has been approached undertaken by a number of groups. National and provincial parks, conservation areas, and nature or ecological reserves funded by land trusts, citizen groups and other agencies have all helped to create a network of protected areas throughout the province of Ontario (Elliott 1998). Many municipalities now also take active roles in the protection of natural areas.

The Regional Municipality of Waterloo is a highly developed area. In highly settled landscapes, regionally and municipally protected landscapes form an important component of a natural areas network

for conservation. The Regional Municipality of Waterloo designates several types of protected areas, including Environmentally Sensitive Landscapes (ESL), Environmentally Sensitive Policy Areas (ESPA), and Regional Agreement Forests. Combined with regional parks, private woodlots and watercourses, these areas form the "natural forest network" that serves as the core of the ecosystem region (Slocombe 1998, p 34). Individual ESPAs protect local species specific populations and ecosystem communities; across the landscape ESLs, ESPAs and other areas create a regional network of protected areas. The location of the ecological restoration for this study is in Natchez Hills, as ESPA located in the City of Kitchener.

Environmentally Sensitive Policy Areas

The Environmentally Sensitive Policy Area (ESPA) designation is for natural areas which meet selected criteria (Schmitt 1995). ESPAs are designated in the Regional Official Policies Plan (Regional Municipality of Waterloo 2011), and are meant to address two objectives. As outlined in the City of Kitchener Draft Municipal Plan the objectives are:

> "1) To provide for the long-term protection and preservation of naturally significant areas.
>
> 2) To ensure that development within or contiguous to any designated environmental areas is cognizant of the natural functions of the area and makes use of environmental planning principles." (Schmitt 1995)

Currently there are 80 ESPAs in the Regional Municipality of Waterloo, 14 of these are in the City of Kitchener and Natchez Hills is ESPA #24 (Schmitt 1995; Regional Municipality of Waterloo 2011). The ESPAs in the City of Kitchener are important areas for conservation both locally and regionally; ESPAs are viewed by the Region of Waterloo as "crucial nodes" in the greenspace network (Schmitt 1995).

Despite the designation of numerous natural areas and parks as ESPAs, little management and protection of these areas have been undertaken, owing largely to the sentiment that they would be self-sustaining (Schmitt 1995). Although part of a greenspace network, many ESPAs are isolated from one another and are described as islands in the landscape (see Figure 3.2). There is also some concern that the species for which the ESPAs were noted are no longer present (Schmitt 1995). Along with the issues of isolation and fragmentation, there has been evidence of degradation caused by humans in some ESPAs. The conservation and restoration of native vegetation within urban natural areas is one of the general management objectives the City of Kitchener has identified to maintain biodiversity and ecological integrity in urban natural areas; ecological restoration in specific ESPAs is the active management strategy identified to achieve this objective (City of Kitchener 2006).

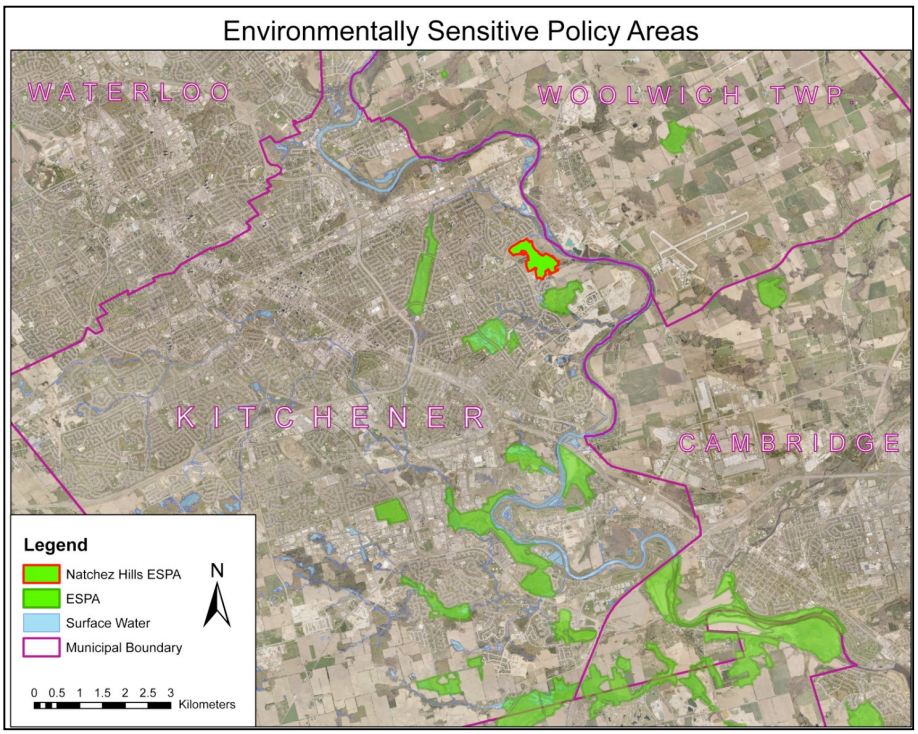

Figure 3.2 Environmentally Sensitive Policy Areas (ESPAs) in the eastern portion of the City of Kitchener. Natchez Hills ESPA is outlined in red. Other cities and townships in the Regional Municipality of Waterloo are also shown. Source: ESPA data from Regional Municipality of Waterloo (2009), Orthoimagery base layer data from First Base Solutions (2006), Surface Water and Municipal Boundary Data from GRCA (2003).

Natchez Hills Environmentally Sensitive Policy Area

Natchez Hills ESPA is located in the eastern portion of the City of Kitchener, in Ward 6. Natchez Hills ESPA is approximately 8.4 hectares in size (J. Shea, email communication, Apr 6 2011), however it is often reported along with the nearby Lackner Woods ESPA, for a combined area of 38.4 hectares (94.9 acres). Twenty hectares of the Natchez Hills area were designated part of the City of Kitchener's parkland in February 1995. Natchez Hills ESPA is also designated as a Life Science Area of Natural and Scientific Interest (ANSI-LS) by the Ontario Ministry of Natural Resources, and as an ecologically significant open space area in the Kitchener Official Plan (Schmitt 1995; NHIC 2011).

The rolling topography of Natchez Hills ESPA is the product of till moraines, with common soils being sands, gravels, and fine sandy loams. The area is comprised of steep wooded hills, which slope to the east to the Grand River floodplain (Schmitt 1995). The second-growth forest is predominantly sugar maple-beech, white ash and basswood. Wetland sections, including numerous springs, and small permanent wet pools exist within the boundaries of the Natchez Hills ESPA, and two stream systems flow

through and alongside the area (Schmitt 1995). Natchez Hills ESPA is upland from the Grand River, and groundwater seepage along the slopes facing the Grand River supports a number of species, including some uncommon wetland species (The Walter Bean Community Trails Foundation [date unknown]).

Natchez Hills ESPA is viewed as a significant local natural resource in the Grand River South Community Plan, and as part of the Community's Natural Heritage System (City of Kitchener 2000). It is an important forested area for watershed cover along the Grand River. Forest cover of 30% is recommended for a healthy watershed (Environment Canada 2004b); approximately 19% of the Grand River watershed is forested (GRCA 2004). Natchez Hills ESPA is part of the watershed natural area protected by municipal designation from development and is especially important for this reason, the protected land base may be as much as 10% in the Grand River watershed, but the actual amount is not known (GRCA 2004).

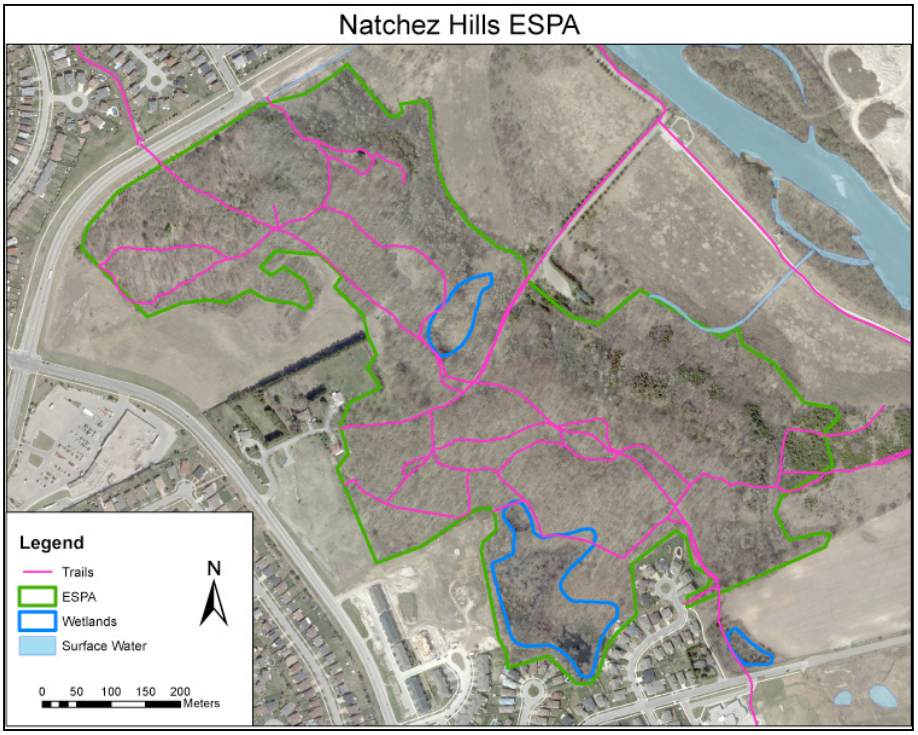

Figure 3.3 Aerial View of Natchez Hills ESPA #24. Natchez Hills ESPA showing ESPA boundary, all trails (official trails as well as trails created through unpermitted uses), wetlands, and nearby surface water (the Grand River). Source: Trail Data from The City of Kitchener Municipal Data (2008), ESPA Data from Regional Municipality of Waterloo (2009), Wetlands and Surface Water Data from GRCA (2003), Orthoimagery Data from Regional Municipality of Waterloo (2006).

Natchez Hills ESPA provides a significant contribution to landscape level corridors (through the Grand River corridor and associated flood plains and valley lands) and the network of protected and natural areas within the City of Kitchener and the Region of Waterloo. It is in very close proximity to

Lackner Woods ESPA (ESPA #25) (Schmitt 1995); Lackner Woods is approximately 0.5 km to the southeast and is separated from Natchez Hills ESPA by residential development. Natchez Hills ESPA is also located near to an upland woodlot referred to as the Fung Woodlot (City of Kitchener 2000).

Natchez Hills ESPA has remained mostly unchanged in size and shape for the past 50 years; what has changed is the surrounding land-use. Once a matrix of agricultural fields, the landscape surrounding Natchez Hills ESPA is now urbanized, as a result of increasing residential development. Over the two field seasons during which this ecological restoration experiment took place there was considerable development adjacent to Natchez Hills ESPA. The area to the south of the study site was under construction, and several new housing complexes were erected in this time. These complexes are not shown, but the area cleared for their construction is visible in Figures 3.3 and 3.5.

Natchez Hills ESPA is subject to pressures of urbanization, fragmentation, edge effects, and invasion by exotic plant species, including *Alliaria petiolata* (garlic mustard) and *Chelidonium majus* (greater celandine) (Ferguson 2007). Recreational activities have also caused significant damage to the forest ecosystem in Natchez Hills ESPA.

Figure 3.4 Invasive Species in Natchez Hills ESPA. *Alliaria petiolata* and *Chelidonium majus* patches in Natchez Hills ESPA, to the north of ecological restoration site. *Alliaria petiolata* patches are visible in the background of the photo, while *C. majus* patches are visible in the foreground.

The Grand Valley Trail runs through Natchez Hills ESPA and is used for recreational hiking (Schmitt 1995). Further recreational uses of Natchez Hills ESPA include mountain biking, although this

activity has not been limited to the trails and has caused severe degradation (Lefler 2006). The hilly topography of Natchez Hills ESPA provided an off trail riding course for mountain bikers, which was augmented through the construction of jumps, bridges, and numerous other built elements. The resulting degradation in areas heavily used by mountain bikers included the complete destruction of the forest understory, soil disruption and compaction, erosion, and the introduction of invasive species, among other effects (Lefler 2006).

Restoration efforts were undertaken in the most seriously degraded areas of Natchez Hills ESPA beginning in 2003, using a passive restoration approach to allow the damaged ecosystem to recover. Active restoration efforts included the removal of built structures for mountain biking, the relocation of dead trees and coarse woody debris, soil excavations to mimic the natural topography and habitat of the site, and planting of native herbaceous plantings and saplings to encourage forest understory regeneration (Lefler 2006; Ferguson 2007). Lefler (2006) found that the restoration efforts undertaken in Natchez Hills ESPA in 2003 will require more time to recover, based on the extensive damage caused by mountain biking. The restoration efforts undertaken in 2003 were not observed to be thriving or expanding, and there was minimal woody regeneration. Therefore a more intensive restoration strategy, including a more intensive planting regime was recommended (Lefler 2006).

Following Lefler's (2006) recommendations, Ferguson (2007) used populations of three native herbaceous species in an early restoration experiment in the same general area, to determine if the speed of restoration could be accelerated (Heritage Resources Centre 2006). The ecological restoration experiment completed during this study builds on the restoration work and monitoring recommendations in both of these studies, and focuses on a nearby area in Natchez Hills ESPA where little forest understory vegetation remained following extensive human caused degradation.

Considerations for Site Selection

The area in Natchez Hills ESPA identified as suitable for an ecological restoration experiment was first visited in late April 2007. The general area suitable was recommended based on a previous visit to Natchez Hills ESPA by Chris Gosselin (Regional Municipality of Waterloo) and Stephen Murphy (University of Waterloo) (S.D. Murphy, personal communication, April 2007). Several criteria, including proximity to trails, location of previous disturbance, and location of previous restoration activities were used to narrow the general area suitable to the specific ecological restoration location.

The Grand Valley Trail traverses Natchez Hills ESPA to the northeast of the ecological restoration site. To decrease proximity to the trail, the ecological restoration site was located as far southwest of the Grand Valley Trail as possible given the ESPA boundaries. Maintaining a large distance

between existing trails and the ecological restoration site was designed to decrease visibility of the site and discourage people from leaving trails to investigate the site, and was also done at the request of the City of Kitchener (S.D. Murphy, personal communication, April 2007). Although ecological restoration experiments present opportunities to help people to understand local ecosystems, not every ecological restoration site is suitable for education. This site was not intended to have a demonstration role, therefore any educational benefits that arise from this study will not be by way of physical visitation to the site. The site for the ecological restoration was located well out of view of the main trails; however some of the additional sites used during the ecological restoration were located nearer to the trails. Certain sites needed to be situated where the restoration species were growing in large populations, and locations for these sites were inherently less flexible.

Previous disturbance history at the site was relevant for multiple reasons. Serious degradation due to off trail mountain bike use has degraded areas of Natchez Hills ESPA significantly. While the destructive activities were mostly halted, the damage is still evident, most notably in the absence of a forest understory layer. Earlier ecological restoration experiments have focussed on this area of Natchez Hills ESPA in recent years; these restorations were concentrated mainly on the steep slopes used extensively during mountain biking. Areas of previous ecological restoration were avoided, to not interfere with their progress.

The ecological restoration site for this experiment is located at the base of the hill used for mountain biking, in an area that is a local depression (see Figures 3.5). The ecological restoration site was selected for its lack of native herbaceous understory vegetation. This is not typical for this type of forest, and is thought to be a result of the degradation caused by mountain biking activities. Large populations of non-native herbaceous species such as *Alliaria petiolata*, *Chelidonium majus* and *Hesperis matronalis* are also found near this location. The combined factors of a lack of native understory species and nearby invasive species made this site a good candidate for early ecological restoration, and to test the effects of different densities of native herbaceous species in ecological restoration.

To the south and west of the general site area are the actual boundaries of Natchez Hills ESPA, these boundaries are reflected in the adjacent landscape. To the south is a newly constructed residential development, and to the west is a meadow (see Figure 3.5).

Figure 3.5 Study Site location in Natchez Hills ESPA. The approximate location of the restoration site is identified by the black box[3]. The ecological restoration site was north of the residential development, east of the meadow, south of the steep slope (visible in the contour lines) previously used for off-trail mountain biking and the site of previous ecological restorations, and southwest of the Grand Valley Trail. Source: ESPA data from the Regional Municipality of Waterloo (2009), Wetland data from GRCA (2003), Trails and Contour Interval data from the City of Kitchener (2008), Orthoimagery data from the Regional Municipality of Waterloo (2006).

Suitable Sources and Species for Ecological Restoration

Restoring native forest herbaceous species presents a number of challenges. One of the most significant practical challenges in the ecological restoration of understory vegetation is the acquisition of plant materials for restoration (Mottl et al. 2007). This can be an especially limiting factor if budgets for purchasing plant materials are minimal. When plant materials of suitable species are available as seedlings or seeds there are still questions about the local adaptive capacity of that material for the restoration site, and even the genetic variability within suitable species.

[3] The decimal latitude and longitude for Natchez Hills ESPA are 43.456 and -80.425 (NHIC 2008). The approximate location of the study site within Natchez Hills ESPA is 43.455 and -80.425 (D Schmitt, personal communication, November 2007).

Transplanting Within Natchez Hills ESPA

In the case of Natchez Hills ESPA, the ecological restoration experiment was designed to restore native forest understory herbaceous species to the site. When determining the most effective method for an ecological restoration experiment with a measurable impact in the time frame for this study, transplants were the most appropriate plant material. Transplanting from existing populations with high densities within Natchez Hills ESPA was the method for obtaining plant material for this ecological restoration experiment. A similar methodology was followed by Mottl et al. (2006), where species selection was dictated by which local source populations were large enough to withstand a harvest. To ensure the removal of transplant individuals did not negatively impact the source populations, donor plots from which the restoration species were sourced were monitored for population and vegetation changes after harvesting (see the section on donor plots in this chapter).

In this ecological restoration experiment, the restoration species were not removed from the ESPA, but were relocated within the boundaries of Natchez Hills ESPA. The City of Kitchener (2006, p 29) specifies that "all vegetation planted within and adjacent to the City of Kitchener's natural areas must be native to the area as defined by the Region of Waterloo, common to the existing or historical habitat, and of a local or nearby genotype whenever possible". Transplanting plant material within Natchez Hills met these planting specifications.

For future ecological restoration experiments, a preferred low cost source of native species transplants may be the rescue of plants from an area near the restoration site and similar in ecosystem characteristics, but slated for development. Murphy (2006) has previously used this technique in the Regional Municipality of Waterloo. This strategy requires several years of advance planning of restoration activities to allow for identification of suitable sites for plant rescue, and may also require an area where plants can be housed and cared for until they can be used in an appropriate ecological restoration project.

Considerations for Species Selection

Before the ecological restoration experiment could begin, the three native forest understory herbaceous species to be used had to be selected. Several species present in Natchez Hills ESPA were considered for use in this study. These included *Trillium grandiflorum* (Michaux) Salisbury (white trillium), *Uvularia sessilifolia* Linnaeus (sessile-leaved bellwort), *Maianthemum racemosum* subsp. *racemosum* (Linnaeus) Link. (large false Solomon's seal), *Caulophyllum giganteum* (Farwell) Loconte & W.H. Blackwell (giant blue cohosh), *Sanguinaria canadensis* Linnaeus (bloodroot), *Dicentra cucullaria*

(Linnaeus) Bernhardi (Dutchman's breeches), *Erythronium americanum* Ker Gawler (yellow trout lily) and *Podophyllum peltatum* Linnaeus (May-apple).

Several factors in the forest understory affect plants' distributions in various patches, including total available light, seasonal and daily distributional changes in light availability, nutrient availability, soil depth, and soil moisture (Neufeld and Young 2003). However, measures such as these were not included in the selection of restoration species. Mottl et al. (2006) address the association of forest perennial herbs with light, nutrients, and other specific environmental factors, and how restorations of these species often emphasize matching species to specific microenvironment conditions. However, the study conducted by Mottl et al. (2006) suggested that detailed knowledge of variation in light and other abiotic factors may not be necessary to undertake herbaceous forest understory perennial restoration. Following this rationale, and evaluating the constraints measurement of these factors would involve, specific microenvironment factors were not collected. The time and financial constraints involved in collecting this data from both the donor site and the restoration site would not create a practical methodology for other practitioners.

Rather, the selection of species was based on which species were already established in various patches near the restoration site in Natchez Hills ESPA. Although microenvironment conditions may have differed from the donor site to the restoration site, species suitability based on local colonization in Natchez Hills ESPA was the primary factor considered. Forest understory species, and especially the understory species of deciduous forests, have adapted to the resource limitations imposed by their location in the forest understory. The ability of perennial forest understory herbaceous species to store carbohydrates in their underground storage organs, and to mobilize these carbohydrates to sustain leaf emergence despite any number of stresses, (including prolonged periods of cloud cover and herbivory) (Neufeld and Young 2003) makes them an ideal forest guild to use in ecological restoration experiments where transplant stresses are involved.

The ultimate criteria dictating species suitability were local availability of species in numbers suitable for transplanting. Balancing the need for locally adapted species and local genetic sources with the native herbaceous species present identified a specific set of species for consideration. Timing of emergence and senescence further reduced the list of possible species, only those that had emerged within the transplanting time frame but had not yet begun to senesce were considered. Species growing within Natchez Hills ESPA in smaller populations, or in large populations in only one area were omitted from consideration for use in the ecological restoration. There was no justification for moving an entire local

population of a species (or a large proportion of a local population) from one area of Natchez Hills ESPA to the ecological restoration site in Natchez Hills ESPA.

Therefore, the species used in this ecological restoration were those found to be in greatest abundance, and in large populations in at least two different areas, to ensure genetic variation. Many herbaceous forest species are perennial, reproduce vegetatively, and are found in patches of clones (Whitford 1949; Whigham 2004). Patches of clonal species may contain few genets, possibly only one or two (e.g. as in *Clintonia borealis* and *Podophyllum peltatum*), or they may contain numerous genotypes (e.g. *Asarum canadense*) (Whigham 2004). To allow for the introduction of more than one genotype of each species to the ecological restoration site, transplants were selected from at least two patches. In the area of Natchez Hills ESPA scouted for source populations, the native herbaceous forest understory species with populations large enough to withstand a harvest from at least two distinct areas were *Podophyllum peltatum, Caulophyllum giganteum* and *Maianthemum racemosum* subsp. *racemosum.*

Furthermore, *Podophyllum peltatum, Caulophyllum giganteum* and *Maianthemum racemosum* subsp. *racemosum* were observed growing together in existing populations in Natchez Hills ESPA. Therefore an ecological restoration experiment using these three species together in combination would create similar species assemblages to those existing in non degraded areas of Natchez Hills ESPA.

Figure 3.6 *Caulophyllum giganteum*, *Podophyllum peltatum* and *Maianthemum racemosum* subsp. *racemosum* growing together in natural populations in Natchez Hills ESPA.

Ecological Restoration Experiment Design

Experimental design encompasses the selection of experiment objectives, the number and physical arrangement of experimental units (replicates), and the manner through which treatments are assigned to replicates (Dutilleul 1993). Good experimental design "requires the avoidance of systematic errors that would produce a bias and would render an experiment invalid or inconclusive", and takes spatial heterogeneity into account in an aim to minimize, "the variance of the random error in the statistical model of analysis" (Dutilleul 1993, p 1647). Overestimating the variance in the random error will affect the detectable differences in the experiment, and the interpretation of the results. Methods to reduce confusion effects in experimental design are control, replication, randomization and interspersion (Hurlbert 1984, cited by Dutilleul 1993).

In a randomized complete block design, a similar group of experimental units (blocks) are divided into subblocks of equal sizes (referred to as quadrats in this study). Treatments are then randomly assigned to the quadrats within each block. Each block must contain all of the treatments, and there is generally one quadrat with each treatment per block (Dutilleul 1993), although multiple replicates of each treatment can be applied, as long as each treatment is applied equally throughout each block, and throughout the entire experiment. When experimental units (quadrats) are grouped into blocks with a relatively constant environment, differences between the quadrats within a block "provide a measure of the treatment effect, whereas repetition of the blocks provides replication of the treatments" (Potvin 2001, p 67).

In this ecological restoration experiment a randomized block design was used to test the experimental effects of the treatments. The experimental block design used four 10 m × 10 m blocks, each containing six 2 m × 2 m quadrats (Figure 3.7). One of three treatments was randomly assigned and applied to each of the quadrats.

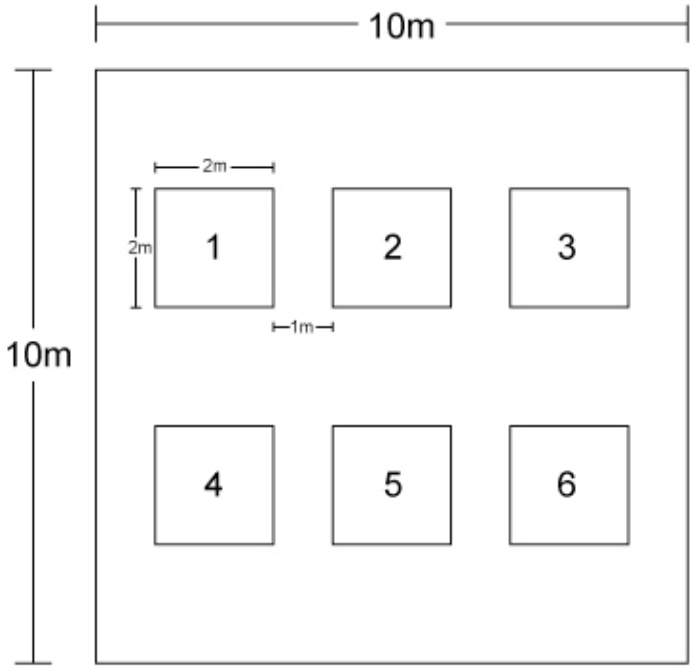

Figure 3.7 Ecological Restoration Experiment Block and Quadrat Layout

Randomization and Replication

Good experimental design requires "randomization of the levels of treatment factors over the experimental units and treatment replication", this randomization provides "a warranty against confounding sources of variation" (Potvin 2001, p 66). Replication is the second tenet of good experimental design, following randomization (Potvin 2001). The two main purposes of replication are to increase the accuracy of the experimental comparison, and to provide an estimate of the error "by which the significance of these comparisons is to be judged" (Fisher 1971, cited in Potvin 2001). Classical randomized block design uses one replication per block, but randomized block design with multiple replications per block has an advantage if there is moderate mortality, or if experimental units are lost for other reasons (Potvin 2001).

A constraint to the design of this experiment was the unpredictability of seasonal changes and weather conditions, and how this would affect the timing of plant emergence, and ultimately the planting schedule. The initial study was designed to have four replications of each treatment, or one replication in each block. Because the conditions were favourable (time and weather permitting), an additional four replications were completed (for a total of two replications of each treatment in each block). This extra

replication provided insurance against the loss of plants through mortality or other events. Quadrats 1 to 3 in all four blocks were completed first, followed by quadrats 4 to 6.

Restoration Treatments

Two of the treatments in the ecological restoration involved transplanting the selected native herbaceous species into the quadrats in two different densities. The third treatment involved no planting at all, and was a control treatment. No existing vegetation was removed from any of the restoration quadrats.

Density Treatments

The two density treatments used in this experiment were 18 individuals per quadrat and 9 individuals per quadrat. These densities were chosen for the 2 m × 2 m quadrats for several reasons. Based on the sapling like structure of *C. giganteum*, Ferguson (2007) used a density of two *C. giganteum* shoot populations per $1m^2$. In this study, similar consideration of the space requirements for *C. giganteum* and the species planted in assemblages with it dictated that the plantings should not be overcrowded. In the density treatment of nine individuals per $4m^2$ quadrat the approximate density is 2.25 individuals per $1m^2$; for the density treatment of 18 individuals per $4m^2$ the density is approximately 4.5 individuals per $1m^2$.

While previous studies have indicated ideal densities of native herbaceous species for outcompeting invasive species (e.g. Murphy 2005), an important part of this study was survivorship of transplants. (Transplants are referred to as shoot populations, as that was the portion of the transplant visible for monitoring). To answer the question of survivorship it was important to be able to conclusively determine that shoot populations observed in 2008 were from the same rhizome as the individual transplanted in 2007. This required that the transplants were adequately spaced, so the 2008 shoot populations could be accurately identified. A grid pattern was adopted in each quadrat, to aid with future identification (Figure 3.8). Transplants were placed in this grid formation as much as possible, based on obstructions within the quadrats, such as trees and existing vegetation.

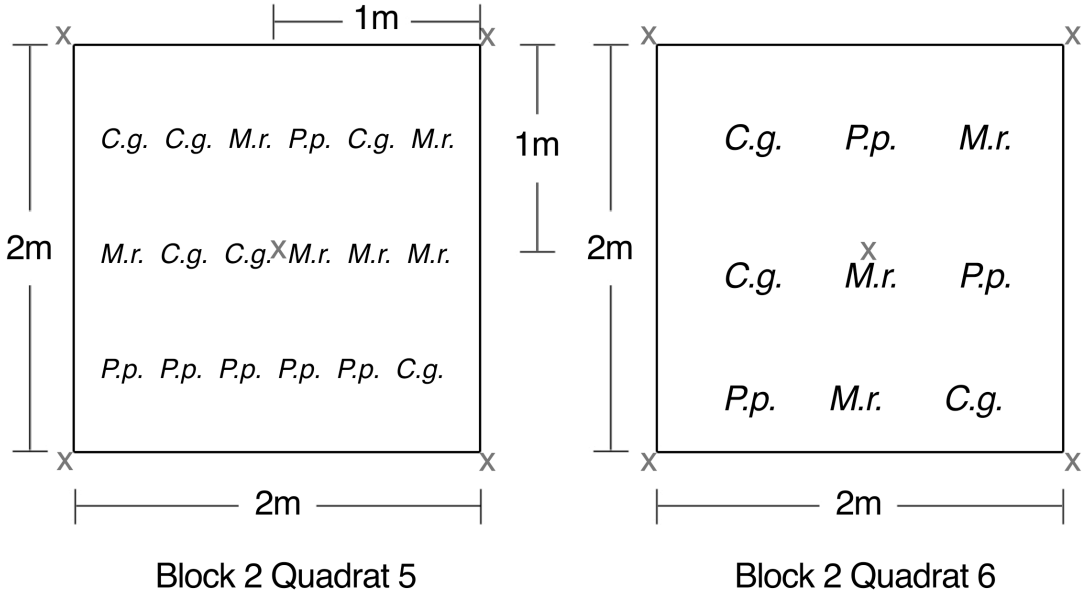

Figure 3.8 Transplant positions and spacing in two quadrats of Block 2, as determined by random assignment of transplant species to a predetermined grid layout. The corners and centre of each 2m × 2m quadrat were marked, as indicated by ×. *C.g.* is *Caulophyllum giganteum*, *M.r.* is *Maianthemum racemosum* subsp. *racemosum,* and *P.p.* is *Podophyllum peltatum*. A full list of species planting locations in each block can be found in the Appendix.

Two factors in this study, the restoration treatment for each quadrat and the planting locations for each species within each quadrat were randomly assigned in order to strengthen the experimental design. The random selection of the restoration species' locations within the quadrats meant that in some blocks several transplants of the same species were planted near each other; this would complicate transplant identification if the grid pattern for planting did not enable verification of plant position. Figure 3.8 presents an example this in Block 2, Quadrat 5. The random assignment of five *P. peltatum* plantings adjacent to one another highlights the need for certainty with respect to identification of shoot populations.

Further supporting the selected densities was the fact that all of the species used in the restoration are clonal, and are therefore likely to reproduce vegetatively. Vegetative reproduction causes the pioneer plant (in this case the transplant) to be closely surrounded by any offspring it bears vegetatively (Clapham 1936, cited in Whitford 1949). Since an ongoing goal of this ecological restoration is for the transplants of *C. giganteum*, *P. peltatum* and *M. racemosum* subsp. *racemosum* to disperse into an area of Natchez Hills ESPA lacking native understory vegetation, allowing the transplants sufficient room for dispersal was a necessary consideration in the ecological restoration experiment design.

Control Treatment

In the ecological restoration experiment, the control treatment consisted of the planting of no transplant species. This control treatment was included to provide an indication of how natural recruitment (Ehrlen and Eriksson 2000) played a role at the ecological restoration site. The control treatment quadrats were used to demonstrate if shoot populations present in the experiment in the second field season were a result of natural regeneration from seeds present in the soil seed bank, or as emerged plants from rhizomes which did not produce ramets during the first field season. The control treatment was replicated in eight quadrats, the same number of replications as each of the density treatments.

Table 3.1 Randomly assigned treatments for each quadrat.

	Block 1	*Block 2*	*Block 3*	*Block 4*
Quadrat 1	DENSITY 18	DENSITY 18	DENSITY 18	CONTROL
Quadrat 2	DENSITY 9	DENSITY 9	DENSITY 9	DENSITY 18
Quadrat 3	CONTROL	CONTROL	CONTROL	DENSITY 9
Quadrat 4	DENSITY 18	CONTROL	DENSITY 9	CONTROL
Quadrat 5	DENSITY 9	DENSITY 18	DENSITY 18	DENSITY 18
Quadrat 6	CONTROL	DENSITY 9	CONTROL	DENSITY 9

Control and Reference Sites

Reference information is an important aspect of the practice of restoration ecology, and is used to "define restoration goals, determine the restoration potential of sites, and evaluate the success of restoration efforts" (White and Walker 1997, p 338). Reference sites provide benchmarks by which to measure ecological restoration success. Most ecological restoration experiments are compared against reference information that is either based on historical data from the site to be restored, or from reference sites that provide contemporary data (White and Walker 1997). Both of these commonly used measures have some degree of difficulty associated with their use. Interpretations of historical data can be confounded from unmeasured factors (White and Walker 1997), and historical data may become biased through the views of original compilers and subsequent generations (Suffling et al. 2003). Finding a contemporary analog which is an ecologically relevant 'match' for a restoration site is also not without problems (White and Walker 1997). Other sources of reference information can include projections of future site characteristics, especially in areas where climate change is expected to have a direct effect on the ecosystem.

The control treatment in the ecological restoration site was designed to determine the amount of natural regeneration occurring at the ecological restoration site. Given the degraded state of the ecological restoration site a non-degraded reference site was needed, to serve as an undisturbed ecologically relevant match and as a benchmark for ecological restoration progress. In choosing a reference site for this

ecological restoration, several criteria including distance from the restoration site, species composition, and ecosystem type were considered.

A Non-Degraded Reference Site

The absence of the restoration species in the control quadrats meant that the control quadrats were not suitable for use in the statistical analyses of the ecological restoration experiment. The absence of shoot populations would skew any statistical tests and not provide useful results. A suitable reference site was needed, where normal phenology and natural variations in the species shoot populations could be observed and compared with the shoot populations in the ecological restoration.

Another study that began in 2007 in the City of Kitchener provided this reference information. A nearby site in Natchez Hills ESPA, 230 m away from this study site was the site of a phenological study on herbaceous forest understory species. The reference site was located in a central interior forest, and surveyed the phenological characteristics of all species present; this included *C. giganteum, P. peltatum* and *M. racemosum* subsp. *racemosum* (S.D. Murphy, personal communication, 2007). The information from this reference site was extremely suitable for comparison due to the presence of the same species and the similar site characteristics, and was therefore used as the unimpaired reference condition for the statistical analyses of the ecological restoration treatments.

Additional Methodological Considerations in the Ecological Restoration Experiment

Most studies examining the physiology of forest understory herbs tend to focus on light (Anderson 2003), as it is the primary limiting resource for growth; however nutrients, moisture, mychorrizal associations, herbivory, pollinator availability and a number of other factors all influence forest understory herbaceous species and their persistence and distributions (Neufeld and Young 2003; Whigham 2004). Ecological restoration experiments involving forest understory herbaceous species usually involve some consideration of these factors in the methodological design. The decisions made with respect to several of these factors, and the rationale for these decisions are discussed below.

Recreating the Soil Community at the Ecological Restoration Site

The functioning of soil fauna and microbial communities is still largely unknown, however restoration plans must consider these communities, as they are likely critically important to ecosystem function (Newton and Featherstone 2005). The association between forest herbs and mycorrhizae activity is widespread, every forest herbaceous species that has been examined has been shown to develop these associations, and the primary benefit to the herbaceous species is increased nutrient uptake (Whigham 2004). Mychorrizal associations may also help some species to alleviate water stress in spring (e.g.

Erythronium americanum) (Lapointe and Molard 1997). Enabling the existing mycorrhizal associations in the ecological restoration transplants was therefore an important consideration.

The most practical and cost effective method to establish these relationships at the ecological restoration site was with soil clods. The ecological restoration shoot populations and their rhizomes were transplanted within the same soil clod from which they originated (as per Murphy 2006). This was done to maintain already established plant mycorrhizae relationships, as well as to facilitate survival by reducing transplant shock. This method also reduced the need for expensive soil testing at the ecological restoration site, as the soil the transplants were already growing in was clearly adequate for growth.

A Non-Watering Methodology

The temperate deciduous forest is not subject to a pronounced drought season, however in the summer (and winter when soils are frozen) periods of insufficient water for forest understory species do occur (Neufeld and Young 2003). Understory herbaceous species have most of their roots concentrated in the surficial soils (between five and fifteen centimetres) (Plasilová 1970, cited in Neufeld and Young 2003), a depth at which soil water deficits can develop rapidly (Neufeld and Young 2003). With limited access to soil water supplies, mychorrizal associations may help alleviate water stress during the spring growing season for some species (Lapointe and Molard 1997). More research on the water relations of the different forest understory phenological groups is needed, but it has been suggested that as a group, forest understory herbs have greater maximum hydraulic conductivities than woody seedlings (Neufeld and Young 2003), facilitating water transport within the plant despite limited access to water supplies.

Since the selection of restoration species was based on their local abundance, and not on predicted species responses to precipitation levels, a non-watering methodology was applied for all three restoration species. This was also done in keeping with the practical methodology that could easily be replicated by the City of Kitchener. There are no facilities to provide access to water at Natchez Hills ESPA, similar to many other ESPAs.

Herbivory

Herbivory of the forest understory by invertebrates and mammals can have significant impacts on both individuals and populations of herbaceous species (Anderson 2003). Dramatic increases in populations of herbivores such as white-tailed deer (*Odocoileus virginianus* Zimmerman) can directly and indirectly alter plant species richness and community dynamics, if some species are browsed to extinction and others are avoided (Anderson 2003). Exclosures are one method commonly used to protect restoration plantings from herbivory by mammals. No instances of herbivory were recorded in Ferguson (2007) or Lefler's (2006) observations of herbaceous species in Natchez Hills ESPA, despite the fact that

some species, including *Erythronium americanum* Ker Gawler (which was used in an ecological restoration by Ferguson [2007]), are commonly and even preferentially browsed. Additionally, exclosures may have attracted people to the ecological restoration sites. Therefore, no exclosures were erected.

Herbivory of forest understory species is complex and species dependent, some species produce volatile chemicals that attract or repel herbivores, and the handling of plants by experimenters can magnify these chemicals. Visitation by experimenters has been shown to influence the intensity of herbivory on plants (e.g. Cahill et al. 2001, 2002), however the variability and unpredictability of this effect has resulted in it being termed "the herbivory uncertainty principle" (Cahill et al. 2001). Other authors have shown that as a general rule, visitation does not impact estimates of herbivory (Bradley et al. 2003). In this ecological restoration experiment gloves were not used to transplant species, however the restoration transplants were moved in soil clods, not transplanted bare root, meaning the plants themselves were not touched. While in initial visits information on plant characteristics such as height were measured, subsequent monitoring was on a presence/absence basis, and did not require the restoration species to be handled.

Donor Plots

Since transplanting within Natchez Hills ESPA was the source of plants for this ecological restoration experiment, it was determined that the study design should include monitoring of the sites from which the restoration species were removed; these were referred to as donor plots. Donor plots were 1 m × 1 m plots set up in areas where the restoration species were abundant. For *P. peltatum* two general areas in Natchez Hills ESPA were used, for *C. giganteum* two general areas were used, and for *M. racemosum* subsp. *racemosum* three general areas were used. All donor plot areas were within 150 m of the ecological restoration site and were within the boundaries of Natchez Hills ESPA. Monitoring provided information on any immediate impacts in the donor plots in the first two years of the ecological restoration.

Ecological Restoration Experiment Methods

Block Locations

The ecological restoration site was scouted and set up in preparation for planting on May 2 2007. Four 10 m × 10 m blocks were randomly distributed within the site and numbered Blocks 1 to 4. The blocks were situated randomly throughout the site in similar alignments, in areas with comparable characteristics, including general absence of existing herbaceous understory vegetation and avoidance of large trees and other obstructions. We selected blocks that were as similar as possible, to minimize the

spatial effects of different blocks, and to strengthen the power of replications within the ecological restoration.

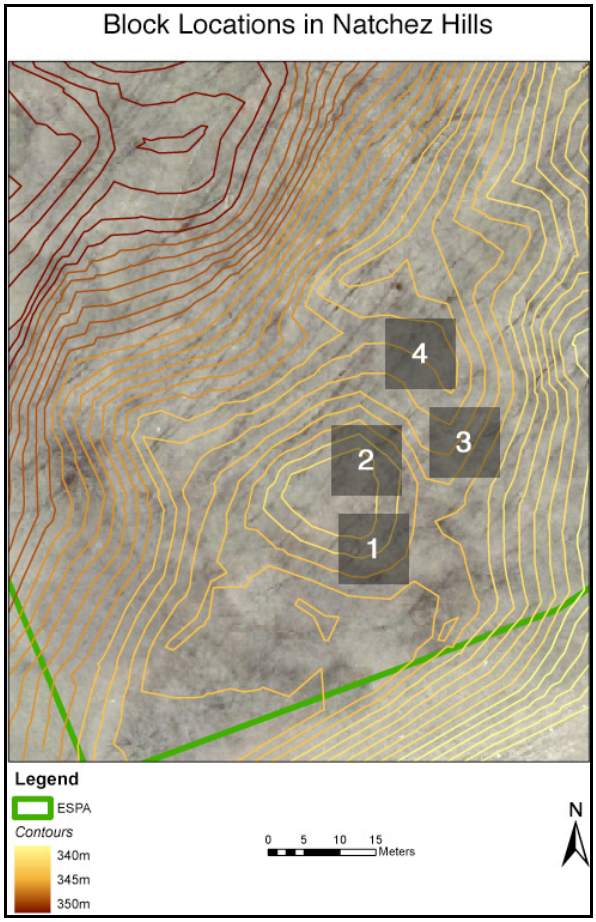

Figure 3.9 Block locations for the ecological restoration experiment in Natchez Hills ESPA. Block locations are approximate. Source: ESPA data from the Regional Municipality of Waterloo (2009), Contour Interval data from the City of Kitchener (2008), Orthoimagery data from the Regional Municipality of Waterloo (2006).

All of the blocks slope to some extent due to the general rolling topography of the terrain, but compared to the nearby steep slope (the focus of other ecological restoration experiments) and other nearby hills in Natchez Hills ESPA, the slopes in the ecological restoration site are moderate. Nonetheless, it is still worth reporting variation among the quadrats within each block, as follows (contour intervals demonstrating elevation changes are depicted in Figure 3.9). Within the ecological restoration site Block 1 is the southernmost block, and slopes northward into a local depression. Block 1 Quadrats 1 to 3 have moderate slopes, while quadrats 4 to 6 are slightly sloped. Block 2 is located north of Block 1, and also slopes into the local depression. Block 2 Quadrats 1 and 4 are located on the eastern edge of the depression and slope moderately to the west. Block 2 Quadrats 2 and 3, are flat and in the

centre of the depression, while quadrats 5 and 6 each slope slightly, quadrat 5 to the southwest and quadrat 6 to the south. Block 3 is located east of Block 2 on a slightly higher ridge area. Block 3 Quadrats 1, 2, 4 and 5 are relatively flat, whereas quadrats 3 and 6 have a slightly greater slope. Block 3 Quadrats 1 and 4 slope slightly to the east, quadrats 2 and 5 slope slightly to the west, and quadrats 3 and 6 slope moderately to the west. Block 4 is located northwest of Block 3 on a flat higher area. All quadrats in block 4 slope slightly to the southwest, but are relatively flat.

Site Set-Up and Initial Measurements

Within each of the four blocks, the six 2 m × 2 m quadrats were set up. Quadrats 1 through 3 were in the south end of the block, and quadrats 4 through 6 were located directly north of the first three quadrats. A minimum of 1 metre was left between adjacent quadrats, to allow access to each quadrat and each of the individual transplants during monitoring (see Figure 3.7). The general layout was altered slightly in Block 1 where quadrats 1 and 4 were moved 1 metre to the east to avoid a large tree (left in Figure 3.7), and in Block 2 where quadrat 3 was moved 2 metres south to avoid a fallen tree and debris (up in Figure 3.7). Quadrats were marked at all four corners, as well as in the centre using small markers; the northwest and southeast corners of each quadrat were also marked using survey flags.

Prior to transplanting, percent cover analysis of species composition in the ecological restoration quadrats was completed in all blocks. Percent cover analysis was recorded to provide an estimate for species composition in the restoration area in a quick and efficient manner, and was recorded on numerous dates to provide records of changes experienced in the quadrats over the course of the ecological restoration. Percent cover was recorded initially on May 2 and May 3 2007 before any restoration planting occurred, and again on May 28 2007, after transplanting of the restoration species was complete. Percent cover was recorded in 2008 on May 12, May 28, June 10 and June 26. Percent cover analysis was completed in $1m^2$ blocks (4 times for each $4m^2$ quadrat) to simplify approximation and allow the use of the $1m^2$ quadrat frame as a visual aid.

Donor Plot Set-up

A 1 m^2 quadrat frame was used to define the donor plot area. The quadrat frame was placed selectively in areas where the restoration species were abundant. All four corners were marked with small wooden numbered markers. Two opposite corners were marked with numbered survey flags to enable easier resurveying of the donor plots in the second field season. A total of 83 donor plots were used to harvest the three species used in the ecological restoration (29 for *P. peltatum*, 24 for *M. racemosum* subsp. *racemosum* and 30 for *C. giganteum*). Once donor plots were identified several measures of plant composition in the plot were recorded before harvesting of transplants began. For the restoration species a

stem count was recorded, to monitor specific data on changes in these species through time. Percent cover analysis for the entire donor plot was also recorded, to provide a baseline of overall species composition.

Restoration Species Planting Locations

To ensure the random location of each species' transplants in the quadrat, the position numbers for each density treatment (one to eighteen and one to nine) were randomly selected (see Appendix). In the first four replications of each treatment (quadrats 1 through 3 in all blocks) the transplant species were randomly assigned positions within the quadrats in the order of *P. peltatum, M. racemosum* subsp. *racemosum, C. giganteum*, since this was the order of donor site preparation. For the next four replications (quadrats 4 to 6 in all blocks), all donor sites for all three species were set-up, so there was no constraint on the order in which the species were planted. Therefore, in replications five to eight, the order of the transplant species was randomly selected. A transplant species was randomly selected, and six (in the density eighteen treatment) or three (in the density nine treatment) planting locations within the quadrat were randomly selected and assigned; this was then repeated for the second transplant species. The final six or three positions were assigned to the third transplant species. For a review of the density and control treatments for each block see Table 3.1.

Transplant Methods

Transplanting started as donor plot locations were set-up and their statistics were recorded. (Due to a limited planting timeframe these activities were done concurrently by me and a field assistant). From within each donor plot two to three individual transplants were removed. For *C. giganteum* the number of stems transplanted for each individual transplant varied from one to six stems, and the average stem count was 1.92 (standard deviation ±0.88). All *P. peltatum* shoots transplanted were single vegetative stems. All *M. racemosum* subsp. *racemosum* shoots transplanted were single stems. For *C. giganteum* and *M. racemosum* subsp. *racemosum* it was not possible to determine if the individuals were vegetative or sexual at the time of transplant.

Figure 3.10 *Podophyllum peltatum* transplant from this ecological restoration experiment, in soil clod with leaf litter and other vegetation. Photo Credit: K. Ferguson

Each transplant was removed from the donor plot in a soil clod approximately six inches deep and six to eight inches in diameter, with soil clod variance depending on the size of the transplant itself. In most cases this soil clod included only the transplant species, but some soil clods contained other vegetation as well, as seen in Figure 3.10. Three to four transplants were moved at a time from the donor plots to the restoration site, where they were planted in the predetermined randomly assigned positions. Some leaf litter was transplanted along with the soil clods, and leaf litter from the ecological restoration quadrat was also placed around the base of the transplant after planting was complete. This was done to prevent desiccation of the soil due to the fairly open tree canopy, as tree leaves had not yet fully emerged and a large quantity of light was reaching the forest floor. The placement of leaf litter was the only step taken to improve restoration success after transplanting. No watering was undertaken at any point in the restoration, to ensure the methodology followed was one that would be replicable by other ecological restoration practitioners facing practical constraints.

Monitoring

Monitoring in Field Season One

Following transplanting, the ecological restoration quadrats were monitored on 14 dates in 2007. (The first four replications had two additional monitoring dates due to their earlier planting, for a total of 16 monitoring dates). Immediately following planting the ecological restoration quadrats were monitored on a weekly basis. As the season progressed monitoring was reduced to biweekly visits.

Monitoring in Field Season Two

Survivorship was monitored on 17 dates in 2008. From the end of April, when shoot populations were emerging, through May monitoring visits were conducted every few days. Survivorship was continually monitored during weekly visits in June and biweekly visits in July. The final sampling date of the second field season was July 24 2008. The final sampling date for the emergence of new shoot populations was May 16 2008, however the ecological restoration site was monitored for an additional two months to observe individual plant phenology, as well as to monitor other impacts at the site.

Monitoring of Donor Plots

The donor plots from which the transplant species were harvested were monitored in both field seasons to provide some information on any immediate effects of harvesting, as well as to provide baseline data which could be used in future years. Stem densities for *P. peltatum* clones are known to be fairly constant from year to year (Rust and Roth 1981), and resurveying the donor plots in the second field season monitored the effects of removal of aerial shoots (and rhizomes portions) on *P. peltatum* stem densities. Fruit and seed production of clones is more variable than stem densities (Rust and Roth 1981), therefore although phenological characteristics were also monitored, the change in total number of aerial shoots was a better determinant of impact of harvesting. The same methodology was used to determine the impact of harvesting on both *C. giganteum* and *M. racemosum* subsp. *racemosum*.

Interpretation of Field Results

Shoot Populations

In this study the populations of transplanted species present in the ecological restoration blocks or donor plots are referred to as shoot populations. Shoot populations (aerial shoots or ramets) are an easily observable measure of plant survival and health, and one which can be assessed from aboveground. The state of the transplanted rhizomes (and overall plant health) is impossible to determine without digging up or digging extensively in the soil around the rhizome, which would likely harm and could kill the transplanted individuals. Since this experiment is designed to continue indefinitely (similar to Murphy

2005 and other ongoing restoration experiments) no destructive plant or soil sampling methods were used. The only measurements recorded were those available using the aboveground biomass (e.g. presence, flowering) and without harming or collecting the plant tissues or rhizomes.

In the short-term, shoot populations provide a suitable measure of survivorship, and are readily used as a measure of initial restoration success (Menges 2008). To move beyond initial rates and draw conclusions on overall plant health, further benchmarks such as fruiting, flowering and dispersal are used, and these measures generally require a longer-term monitoring protocol than the length of this study.

Stem Counts in Clonal Species

All of the species used in this ecological restoration experiment are clonal species; they reproduce vegetatively as well as sexually (Sohn and Policansky 1977; Brett 1981; Whigham 2004). For many perennial forest understory species, clonal reproduction may be as important or more important for dispersal than sexual reproduction (Harper 1977, cited in Brett 1981; Jolls 2003).

Given the clonal growth forms of these species, using stems counts of shoot populations as a monitoring measurement requires consideration of whether individual shoots represent individual plants, or if there are underground rhizome connections. Whitford (1949) and others have dealt with this in herbaceous forest understory species by counting all aerial stems as individuals, regardless of growth forms. Stem counts recorded in this study are consistent with this methodology for *P. peltatum* and *M. racemosum* subsp. *racemosum,* where one stem was recorded as one individual. *Caulophyllum giganteum* shoot populations were often present in close growing groups of stems, but individuals stem counts were still recorded. In donor plots stems of *C. giganteum* were recorded individually and as groupings, since in areas of high abundance multiple stems were often growing so closely together that individual stem counts were difficult.

Although this stem count method does not provide information on clonal relationships, it is difficult to distinguish the member units of a true clonal individual in the field. This may be the only non-destructive or disturbance inducing method which is useful in a field quadrat study, for many clonal species (Whitford 1949).

Interpretation of Plant Health

Qualitative field observations for herbaceous vegetation are not widely or systematically interpreted into quantitative results. Some authors have used multiple categories to describe plant health, for example, de Kroon et al. (1991) used four categories to describe *P. peltatum* shoot state, including

completely senesced, partly senesced and completely green. However even within the literature on *P. peltatum* this is not a consistently applied method of categorization.

In 2007 and 2008 shoot populations experienced dieback as a result of natural senescence, as well as other factors. Catastrophic events such as windstorms and browsing had immediate effects on the appearance of the plant, but did not necessarily contribute to an immediate decline in plant health. Shoot populations that remained after plant damage sometimes remained green and healthy in appearance for many observation dates, before becoming desiccated and/or dying off. Natural senescence generally occurred over a prolonged period, and individual shoot populations showed evidence of this (e.g. desiccation, discolouration, loss of rigidity) to varying degrees. This was evident in the shoot populations in the ecological restoration, as well as in the nearby natural populations of the study species in Natchez Hills ESPA, including natural populations in the reference site (S. D. Murphy, personal communication, 2008).

In many cases it was impossible to determine if a decline in plant health was a result of a catastrophic event, transplant shock (in the first field season), or natural senescence. In some cases it was likely a combination of these factors, as well as a result of natural variation within the population. To ensure consistency in classification of the shoot population irrespective of the contributing factors, each shoot was classified as present until the aboveground biomass visible during monitoring was entirely dead, at which point it was then termed absent. In the event a plant could not be located it was also classified as absent. This is similar to the classification used by Watson and Lu (1999) for populations of *P. peltatum*, and was used to classify all three transplant species. This classification does not involve any characterization of the health of the rhizome, and is not a comment on overall plant health. To measure overall plant health without digging up the rhizome the ecological restoration will require further years of monitoring, beyond the initial two years in this study.

Interpreting Qualitative Field Observations as Quantitative Values

Vegetative monitoring techniques that require low personal judgement have higher accuracy and precision and reduced observer effects (Korb and Fulé 2008). In order to simplify the monitoring methodology, and allow for consistent interpretation in the field in future years (without the primary researcher and potentially with inexperienced observers), only two possibilities for the state of the shoot populations were identified, present (1) or absent (0). Each qualitative field observation was either present (alive, some green present in the stem or leaves of the plant) or absent (dead, no green remains in the stem or leaves of the plant, or, unable to locate). This classification was used for every sampling date, and facilitates interpreting qualitative field results into quantitative values. This categorization allowed for the

transformation of qualitative descriptive data into percent of shoot populations present, which could then be used to analyze for trends within and between years, and in the statistical models.

Statistical Analyses

The randomized block design used allows the variance to be parsed into the treatment factor, the experimental error, and the block effect (Potvin 2001). Repeated measurements are collected when characteristics or factors of the same individual or site are measured at several different times (Von Ende 2001). Repeated measures examine questions about the "patterns or shape of the response over time" (Von Ende 2001, p 134). The establishment of permanent quadrats allows temporal changes to be separated from spatial variation (McCune and Grace 2002). Since this ecological restoration experiment used quadrats in a randomized block design (permanent for the duration of the experiment), a repeated measures design was used.

Data from successive dates in permanent sample units are usually correlated, and cannot be analyzed as though they are independent replicates of a treatment (McCune and Grace 2002). In measuring the same shoot population through time, each measurement is not independent of the measurements taken on other dates. A basic assumption of analysis of variance (ANOVA) is independence of replicates, so a repeated measures analysis must be used to take into account the correlation among dates (Von Ende 2001, p 135). Repeated measures ANOVA tests allow for natural variation and other correlated effects to be separated from restoration effects. Since the responses of multiple variables were tested, repeated measures multivariate analysis of variance (MANOVA) was the appropriate statistical method to use for analysis (Scheiner 2001).

The statistical analyses of the results were carried out using R (Version 2.8.1), a free software system for statistical computing and graphics (Hornik 2011). Arcsine square root transformations were carried out on the data because the data were not homoscedastic. Statistical analysis was carried out for each year, as well as between years. To conduct statistical analysis between 2007 and 2008, the peak number of shoot populations present in 2007 were compared to the cumulative peak number of shoot populations present for each species in 2008. The data from the ecological restoration shoot populations were also analysed against the phenological information from the reference site. The statistics for tests of significant differences among groups reported by statistical software often include Wilk's lambda, Pillai's trace, Hotelling-Lawley trace and Roy's greatest root. The statistical conclusions about whether groups differ will usually be the same for all four measures. Pillai's trace has been shown to be the most robust to violations of assumptions (Scheiner 2001) and was used as the test of significant differences.

Chapter 4 Results

Results of Field Season One

In 2007 overall shoot population survivorship was above 85% in the six weeks following the completion of transplanting (Figure 4.1). The decline in survivorship was more rapid after this date, but coincides with the expected time of senescence (late summer to early fall) for species of the summer-green guild. The more rapid decline of *C. giganteum* and *P. peltatum* as compared with the dieback seen by Ferguson (2007) is likely a result of the non-watering methodology combined with the lower than normal precipitation in 2007 (Table 4.1). Conditions throughout southern Ontario were described as dry to drought-like in 2007. Precipitation was lower than normal from April to October 2007, with September 2007 reported as the driest September in Waterloo Wellington since 1971 (Environment Canada 2007a). A severe windstorm on May 15 2007 brought a new record for a maximum recorded wind gust (113 km/h; old record was 106 km/h in May 1967); this event downed woody debris that killed two transplants of *M. racemosum* subsp. *racemosum.*

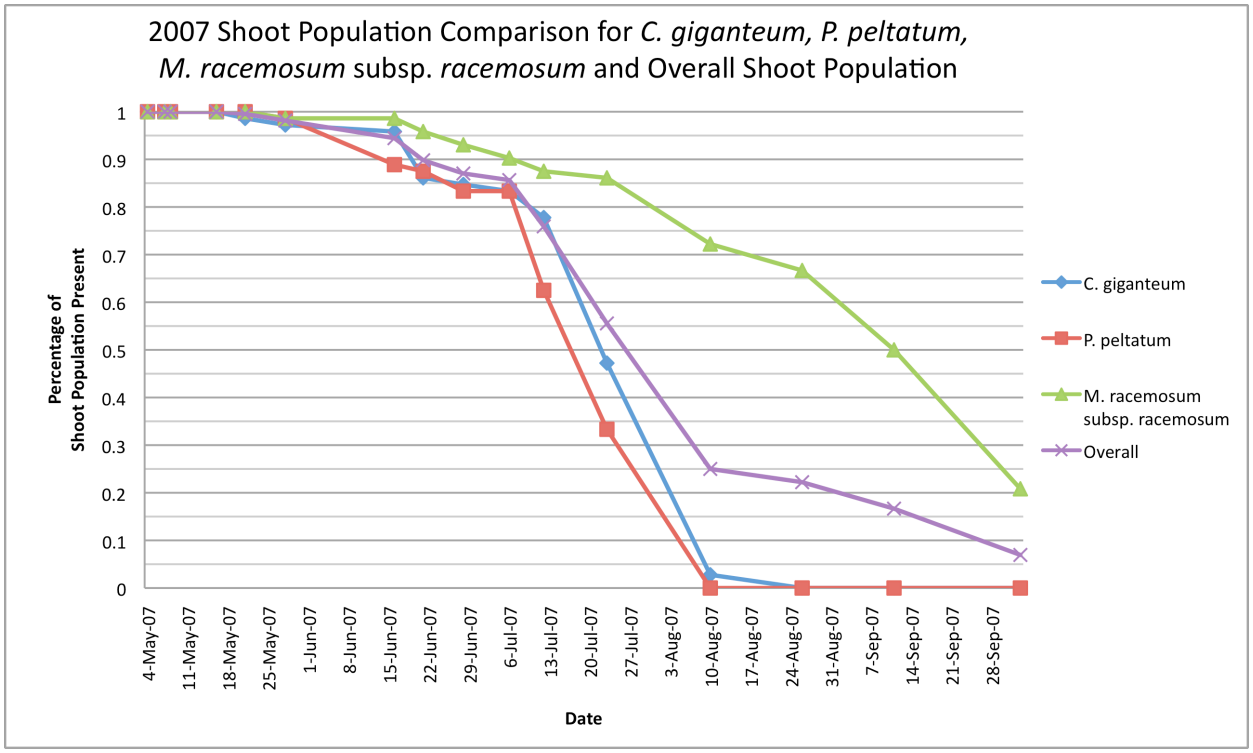

Figure 4.1 Percentage of Shoot Population Present in 2007 for all three transplant species, as well as the overall shoot population.

Table 4.1 Monthly Total Precipitation Comparison for April to September of 2006, 2007, 2008, and Canadian Climate Normals 1971-2000. Percent of normal precipitation (from Canadian Climate Normals) is also shown. Data for 2006 to 2008 is from Region of Waterloo International Airport weather station. Data for Canadian Climate Normals from 1971-2000 is from Waterloo Wellington A weather station[4]. Source: Environment Canada 2010b, 2010c.

Month	**Precipitation (mm)**			
	2006 (Percent of Normal)	**2007 (Percent of Normal)**	**2008 (Percent of Normal)**	**Waterloo Region Climate Normals (1971-2000)**
April	69.5 (90%)	48 (62%)	46 (60%)	76.9
May	93.5 (119%)	46 (59%)	62.5 (80%)	78.3
June	17.5 (22%)	33 (41%)	81.5 (100%)	81.3
July	182.5 (199%)	28 (31%)	203.5 (222%)	91.8
August	38.0 (44%)	13.5 (16%)	84.5 (98%)	86.3
September	141.0 (164%)	23.5 (27%)	112 (131%)	85.8
Total Precipitation April to September (mm)	**542 (108%)**	**192 (38%)**	**590 (118%)**	**500.4**

Results of Field Season Two

In 2008 (Figure 4.2), the peak total shoot survivorship (relative to 2007) was highest (79.6%) on May 16. The cumulative total shoot population for the season was 82.41%, as not all shoots emerged concurrently. *Caulophyllum giganteum* and *P. peltatum* had cumulative shoot population presences above 97%; this was significantly lower ($p < 0.01$; Pearson product moment test) for *M. racemosum* subsp. *racemosum* where the cumulative total was approximately 50%.

Monthly total precipitation in 2008 was much more variable than the consistently lower than average precipitation received in 2007 (Table 4.1). By the end of September 2008, total precipitation for the period April to September 2008 was more than three times greater than the total precipitation received for this same period 2007, and nearly 20% higher than the normal for April to September.

[4] The Region of Waterloo International Airport weather station is located at 43.46 and -80.38 and an elevation of 321.3 metres (Environment Canada 2010b). Waterloo Wellington A weather station is located at 43.35 and -80.38 and an elevation of 317.0 metres (Environment Canada 2010c). The approximate location of the study site in Natchez Hills ESPA is 43.455 and -80.425.

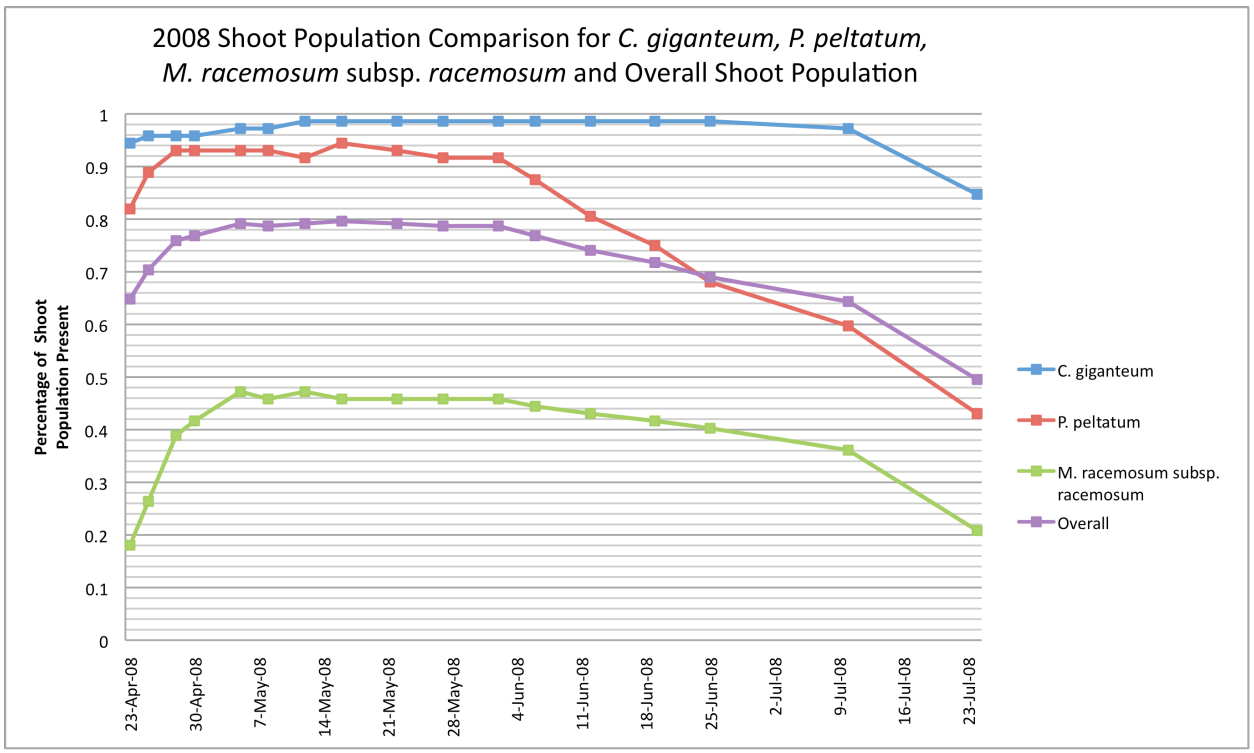

Figure 4.2 Percentage of Shoot Population Present in 2008 for all three transplant species, as well as the overall shoot population.

Control Quadrats

Seven of the eight control quadrats did not contain any of the transplant species used in this ecological restoration in either field season. One control quadrat (Block 1, Quadrat 3) did contain *M. racemosum* subsp. *racemosum* shoot populations in both 2007 and 2008. No change in the approximate number of shoot populations of *M. racemosum* subsp. *racemosum* in this quadrat was observed from the 2007 field season to the 2008 field season. The percent cover data for *M. racemosum* subsp. *racemosum* in Block 1 Quadrat 3 was highest on May 28 2007 at 2.5%, and highest on June 10 2008 at 3%, but the variation in percent cover for all six percent cover sampling dates (two sampling dates in 2007 and four sampling dates in 2008) was very low.

Table 4.2 Control Quadrat Results. Summary table indicating presence or absence of each of the study species in the control quadrats in 2007 and 2008.

Block	**Control Quadrat**	**Species**					
		C. giganteum		***P. peltatum***		***M. racemosum* subsp. *racemosum***	
		2007	**2008**	**2007**	**2008**	**2007**	**2008**
1	3	Absent	Absent	Absent	Absent	Present	Present
1	6	Absent	Absent	Absent	Absent	Absent	Absent
2	3	Absent	Absent	Absent	Absent	Absent	Absent
2	4	Absent	Absent	Absent	Absent	Absent	Absent
3	3	Absent	Absent	Absent	Absent	Absent	Absent
3	6	Absent	Absent	Absent	Absent	Absent	Absent
4	1	Absent	Absent	Absent	Absent	Absent	Absent
4	4	Absent	Absent	Absent	Absent	Absent	Absent

Statistical Analysis of Survivorship Through Each Growing Season

Statistical Analysis of Block Effects

As the season progressed the shoot populations showed a significant decline as indicted by Pillai's trace (Tables 4.3 and 4.4; Figures 4.3 a-c). The spatial effect, as represented by block, had little influence except for block 3 in 2008 (Table 4.5 vs. Table 4.6; Figure 4.4b). There was a significant interaction between blocks and time in 2008, in other words the effect of block diminished over time (Table 4.6 and Figure 4.4b).

Table 4.3 MANOVAR testing responses of shoot populations in 2007

Variable	*Block*			*Time*			*Block × Time*		
	MS	*F*	*p*	*Pillai*	*F*	*p*	*Pillai*	*F*	*p*
C. giganteum	1.14	0.69	0.883	0.52	14.32	<0.01	1.83	0.95	0.811
P. peltatum	1.52	0.74	0.851	0.58	14.87	<0.01	2.01	1.10	0.799
M. racemosum subsp. *racemosum*	1.92	1.03	0.826	0.51	14.26	<0.01	1.74	0.91	0.833
All species	1.49	0.72	0.847	0.55	14.51	<0.01	1.86	0.99	0.817

Table 4.4 MANOVAR testing responses of shoot populations in 2008

Variable	*Block*			*Time*			*Block × Time*		
	MS	*F*	*p*	*Pillai*	*F*	*p*	*Pillai*	*F*	*p*
C. giganteum	9.61	5.92	<0.05	0.71	17.01	<0.01	0.73	17.48	<0.01
P. peltatum	9.28	5.68	<0.05	0.72	17.13	<0.01	0.72	17.15	<0.01
M. racemosum subsp. *racemosum*	9.97	6.14	<0.05	0.76	17.96	<0.01	0.78	18.37	<0.01
All species	9.47	5.77	<0.05	0.74	17.61	<0.01	0.75	18.09	<0.01

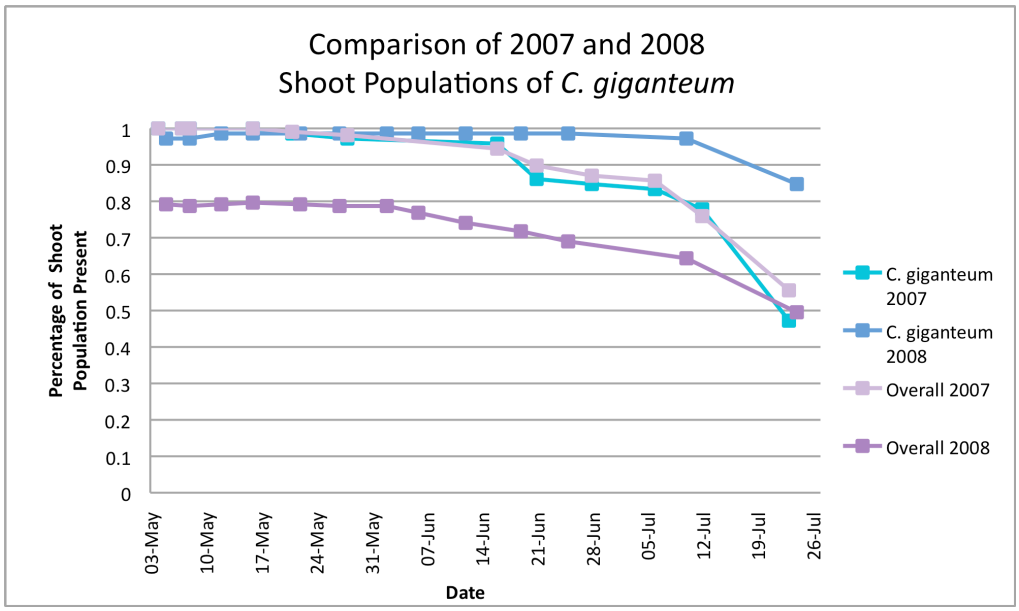

Figure 4.3a Comparison of 2007 and 2008 Shoot Populations of *Caulophyllum giganteum*

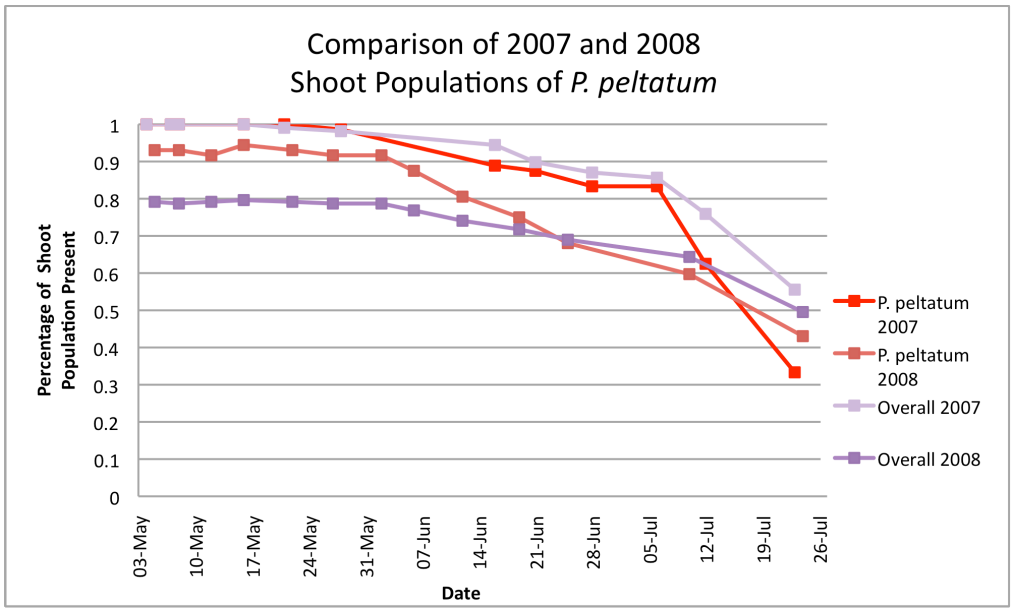

Figure 4.3b Comparison of 2007 and 2008 Shoot Populations of *Podophyllum peltatum*

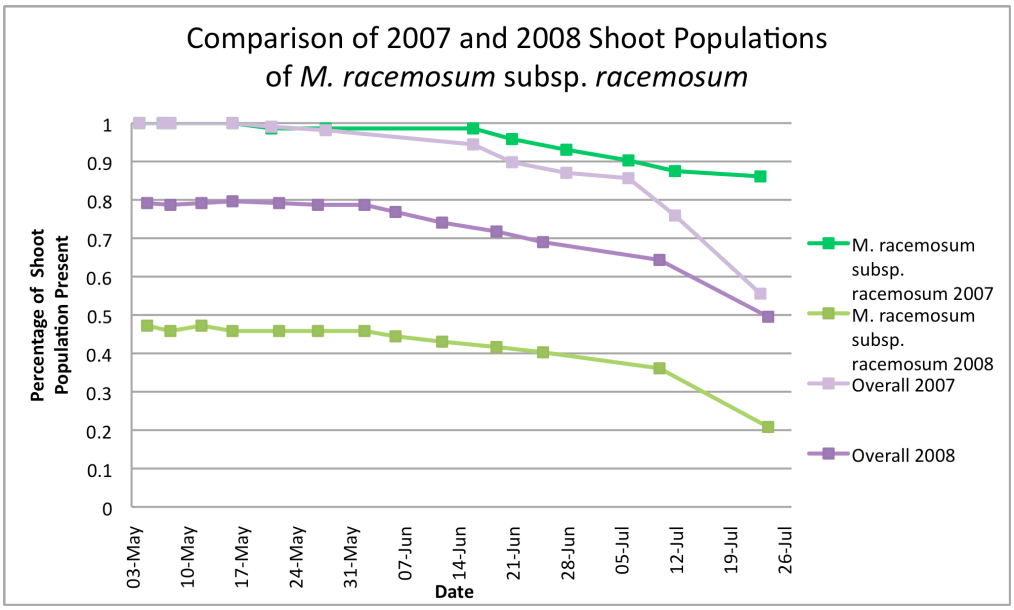

Figure 4.3c Comparison of 2007 and 2008 Shoot Populations of *Maianthemum racemosum* subsp. *racemosum*

Figures 4.3.a-c Comparisons of survivorship of each species transplanted. Overall shoot population survivorship curves representing the shoot population survivorship of all three restoration species are also shown.

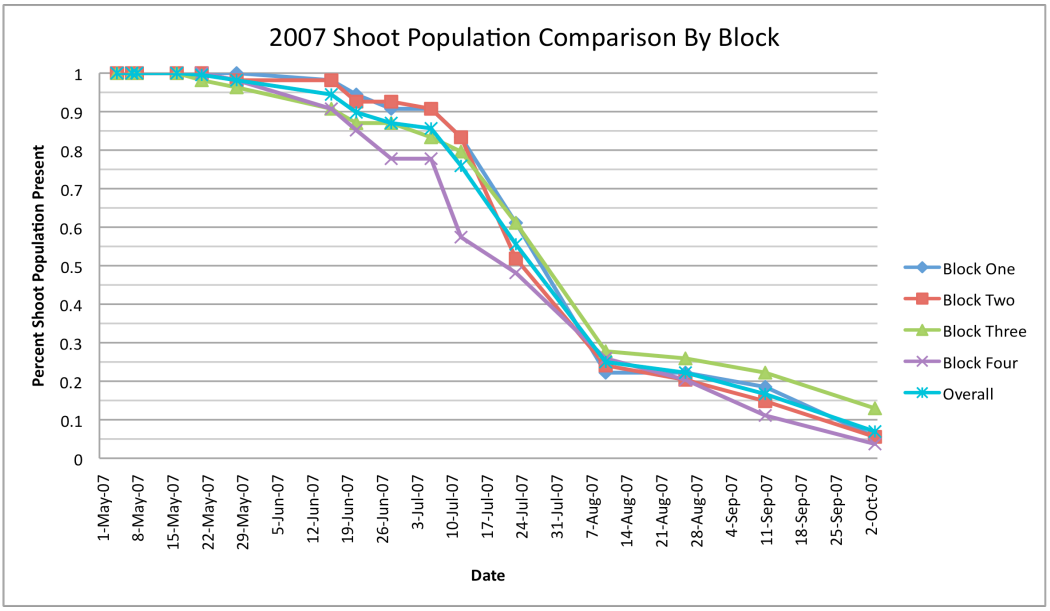

Figure 4.4a Shoot Population Survivorship by Block in 2007

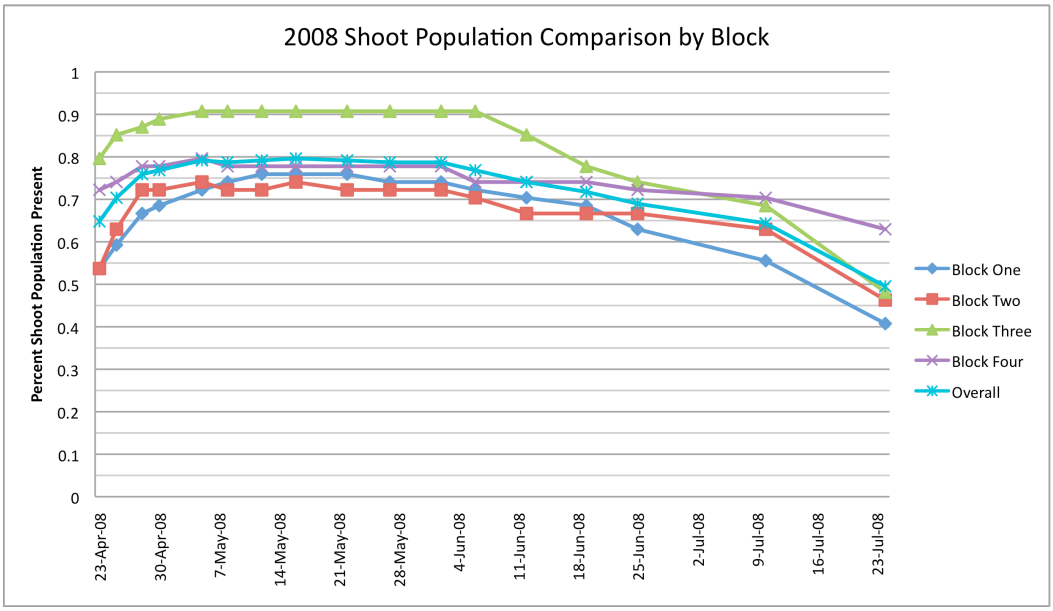

Figure 4.4b Shoot Population Survivorship by Block in 2008

Figure 4.4 a-b Shoot Population Survivorship by Block in 2007 and 2008

Statistical Analysis of Density Treatment Effects

We detected no effect of density treatment on the shoot population survival rates in the first two years of this ecological restoration experiment (Table 4.5 and 4.6, Figure 4.5 a-b). The results for time were, of course, the same as in the analysis related to block effects; similarly, the density × time effect was a separate test.

There were no significant effects of block × density (analysis not shown).

Table 4.5 MANOVAR testing responses of shoot populations to density in 2007

Variable	*Density*			*Time*			*Density × Time*		
	MS	*F*	*p*	*Pillai*	*F*	*p*	*Pillai*	*F*	*p*
C. giganteum	1.0	0.64	0.897	0.52	14.32	<0.01	1.17	0.66	0.891
P. peltatum	1.4	0.73	0.848	0.58	14.87	<0.01	1.80	0.94	0.834
M. racemosum subsp. *racemosum*	1.8	0.97	0.830	0.51	14.26	<0.01	1.49	0.75	0.841
All species	1.4	0.71	0.853	0.55	14.51	<0.01	1.44	0.72	0.850

Table 4.6 MANOVAR testing responses of shoot population to density in 2008

Variable	*Density*			*Time*			*Density × Time*		
	MS	*F*	*p*	*Pillai*	*F*	*p*	*Pillai*	*F*	*p*
C. giganteum	1.11	0.65	0.894	0.71	17.01	<0.01	1.53	0.77	0.829
P. peltatum	1.19	0.66	0.892	0.72	17.13	<0.01	1.59	0.80	0.817
M. racemosum subsp. *racemosum*	1.77	0.94	0.836	0.76	17.96	<0.01	1.38	0.70	0.849
All species	1.33	0.68	0.871	0.74	17.61	<0.01	1.45	0.73	0.851

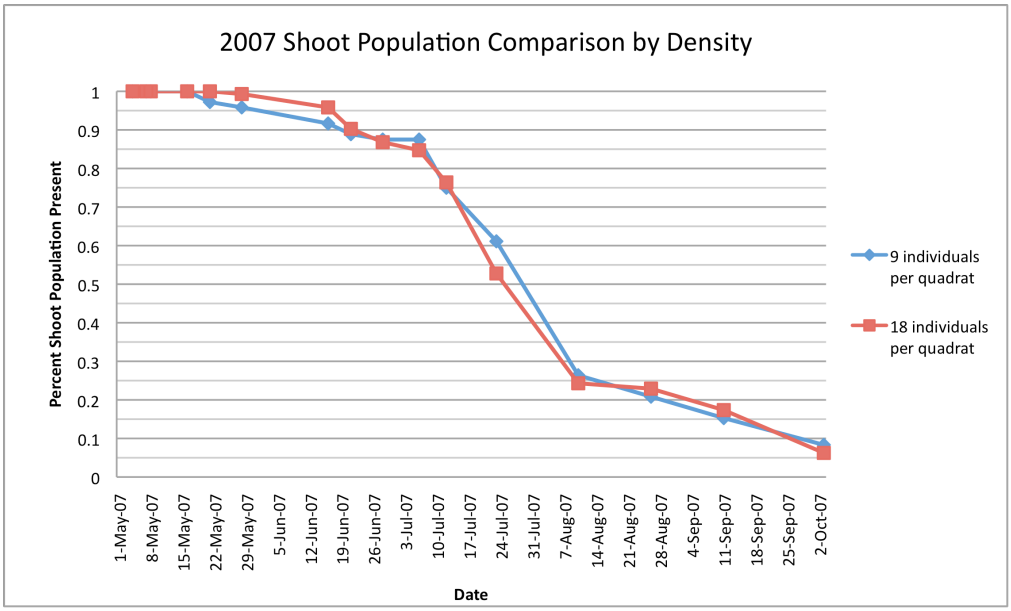

Figure 4.5a Shoot Population Survivorship by Density in 2007

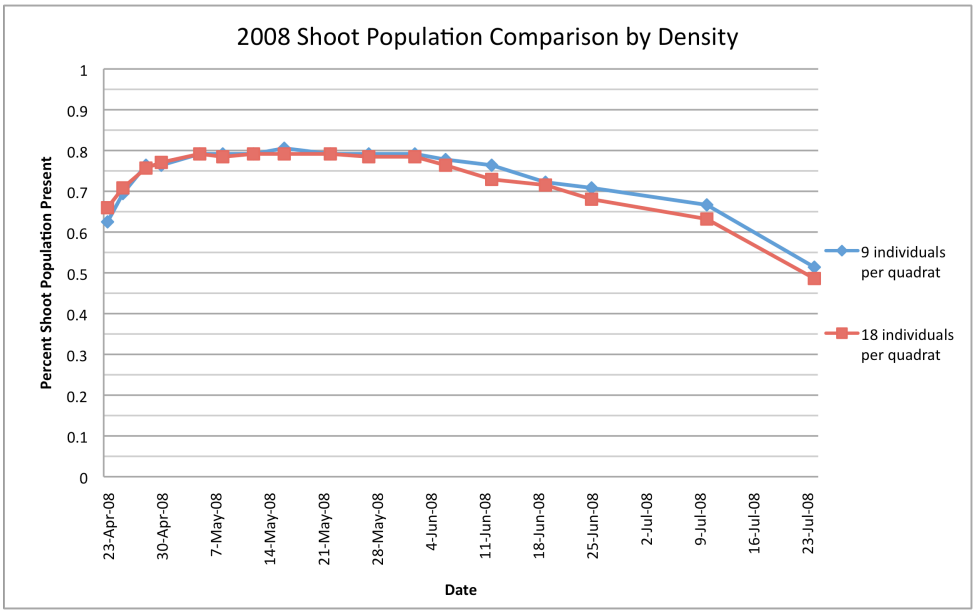

Figure 4.5b Shoot Population Survivorship by Density in 2008

Figure 4.5a-b Shoot Population Survivorship by Density in 2007 and 2008

Thus far, the analyses indicate that there were few significant effects of treatments on survivorship overall, except for block 3 in 2008 and the expected phenological responses as plants senesced through a growing season, as expressed by Pillai's trace testing of the effects of time.

As a final set of tests, Kruskal-Wallis tests were used to compare peak survivorship (thereby eliminating the artefact of senescence). This was done comparing survivorship in treatment quadrats versus a control plot (all species present in similar densities to the treated quadrats; no manipulation performed; control plot was within 230 m of treated quadrats).

Peak survivorship between species in treated quadrats did not vary in 2007 (Kruskal Wallis $p = 0.764$) but did vary in 2008 where *M. racemosum* subsp. *racemosum* was significantly lower (Kruskal Wallis $p < 0.01$). Similarly, comparisons of peak survivorship within species but between years (2007 versus 2008) in treated quadrats indicated no significant differences for *C. giganteum* (Kruskal Wallis $p = 0.813$) or *P. peltatum* (Kruskal Wallis $p = 0.849$), but there were such differences for *M. racemosum* subsp. *racemosum* (Kruskal Wallis $p < 0.01$).

The broader comparison including data from control plots indicated similar results. Peak survivorship between species in all quadrats did not vary in 2007 (Kruskal Wallis $p = 0.719$) but did vary in 2008 where *M. racemosum* subsp. *racemosum* was significantly lower (Kruskal Wallis $p < 0.01$). In all cases, this was not significantly affected by whether the plants were in treated or control plots. Comparisons of peak survivorship within species but between years (2007 versus 2008) in all quadrats indicated no significant differences for *C. giganteum* (Kruskal Wallis $p = 0.748$) or *P. peltatum* (Kruskal Wallis $p = 0.772$) but there were such differences for *M. racemosum* subsp. *racemosum* (Kruskal Wallis $p < 0.01$). Again, this was not significantly affected by whether the plants were in treated or control plots.

Herbivory of Restoration Species

Herbivory in Season One

Of the 216 plantings, 11 (5.1%) showed evidence of browsing in 2007. None of the *P. peltatum* shoot populations were browsed, however eight *M. racemosum* subsp. *racemosum* and three *C. giganteum* shoot populations were browsed, representing 11% and 4% of the shoot populations for these species respectively. Browsing was dispersed through all four blocks, but was more heavily concentrated in Blocks 1 and 2.

Herbivory in Season Two

In 2008, 21 shoot populations were browsed, representing 9.7% of the total population. As in 2007, no *P. peltatum* shoot populations were browsed, 12.5% of the *M. racemosum* subsp. *racemosum* shoot populations were browsed (9 of 72 plantings) and 16.7% of the *C. giganteum* shoot populations were browsed (12 of 72 plantings). Browsing was not observed in Block 1 in 2008, but was observed in

Blocks 2, 3 and 4, and was especially concentrated in Block 3 (57.1% of the browsing occurred in this block).

Dispersal of Restoration Species

In 2008 multiple aerial shoots of *P. peltatum* were observed in 16 of the 72 planting locations (22.22%). Multiple shoots were present at 6 of 72 (8.33%) of the *M. racemosum* subsp. *racemosum* planting locations in 2008. Dispersal was not able to be conclusively determined for *C. giganteum*.

Donor Sites

Finally some descriptive data can illustrate how the species in the donor plots responded to the harvesting of individuals from these plots. Stem counts were completed for the transplant species, and in the case of *P. peltatum* the number of sexual or vegetative stems was counted (for *P. peltatum* this change is evident from the time of emergence, making the distinction obvious). Tables 4.7 to 4.9 provide summaries of the stem count data.

Table 4.7 Donor site statistics for *C. giganteum*. For *C. giganteum* stem count includes group count, for *P. peltatum* stem count includes count of vegetative and sexual plants.

	C. giganteum			
	Total Number		Average Number	
	Number Stems	Number Groups	Number Stems (Standard Deviation)	Number Groups (Standard Deviation)
Before Harvesting May 2007	1131	158	37.70 (9.82)	5.27 (1.68)
After Harvesting May 2007	993	158	33.10 (NA)	5.27 (NA)
May 2008	1319	146	43.97 (18.76)	4.87 (0.97)
June 2008	1261	154	42.03 (16.73)	5.13 (1.20)

Table 4.8 Donor site statistics for *P. peltatum*. For *P. peltatum* stem count includes count of vegetative and sexual plants.

	P. peltatum					
	Total Number			Average Number		
	Number Stems	Number Vegetative [% Total]	Number Sexual [% Total]	Number Stems (Standard Deviation)	Number Vegetative (Standard Deviation)	Number Vegetative (Standard Deviation)
Before Harvesting May 2007	671	579 [86.3%]	92 [13.7%]	23.14 (7.58)	19.97 (7.71)	3.17 (3.54)
After Harvesting May 2007	599	507 [84.6%]	92 [15.4%]	20.66 (7.47)	17.48 (7.71)	3.17 (3.54)
May 2008	644	591 [91.8%]	53 [8.2%]	22.21 (8.72)	20.38 (9.08)	1.83 (2.71)
June 2008	602	554 [92.0%]	48 [8.0%]	20.76 (7.96)	19.10 (8.50)	1.66 (2.27)

Table 4.9 Donor site statistics for *M. racemosum* subsp. *racemosum*.

	M. racemosum subsp. *racemosum*	
	Total Number Stems	Average Number Stems (Standard Deviation)
Before Harvesting May 2007	829	34.54 (7.87)
After Harvesting May 2007	757	31.54 (7.87)
May 2008	884	36.83 (9.77)
June 2008	816	34 (9.76)

Both *C. giganteum* and *M. racemosum* subsp. *racemosum* showed an increase in the total number of shoot populations present in the donor plots in 2008, as compared to 2007 population before harvesting. *Podophyllum peltatum* showed a slight decrease (-4%) in the total number of stems. However, the number of vegetative shoots of *P. peltatum* increased from 2007 to 2008, while the number of sexual shoots decreased. No immediate impacts due to harvesting from the donor plots were observed.

Chapter 5 Discussion and Conclusion

I asked if different density combinations of three native forest understory species, *Podophyllum peltatum, Caulophyllum giganteum* and *Maianthemum racemosum* subsp. *racemosum* differed in their responses in an ecological restoration experiment, in a severely degraded upland forest in Kitchener, Ontario. Based on the results obtained in this study, density was not a significant factor in influencing restoration success. However, species responses were significantly different among the three species used. *Caulophyllum giganteum* and *P. peltatum* were restored with more than 97% success one year after the restoration experiment. *Maianthemum racemosum* subsp. *racemosum* experienced a lower success rate, but was still successfully restored in 51% of the plantings. These early restoration results are promising given that the plantings received no aftercare and were subject to various stresses, including extremely low levels of precipitation in the transplant season, and damage by windstorms, herbivory, and other events.

These results indicate that all three of the native herbaceous understory species used in this study are suitable for use in ecological restorations in urban forests in the City of Kitchener, even in areas which have been severely degraded. These results are consistent with restoration rates reported by other ecological restorations involving forest understory herbaceous species. The results of this study are generalizable to similar urban protected areas in eastern North America, and are of wide applicability given the wide range of all three restoration species. The density response reported in this study is applicable in a worldwide context, recognizing that individual species response rates will vary in other studies.

Discussion of Statistical Analysis

Density as a Factor in Restoration Success

Using multivariate analysis of variance, density was not found to have a statistically significant effect on shoot population restoration in this ecological restoration (Tables 4.5 and 4.6, Figure 4.5a-b). The very high restoration rates for *C. giganteum* and *P. peltatum* in all density treatments show that these species were not influenced by the selected planting densities. Similarly, the shoot population restoration rate for *M. racemosum* subsp. *racemosum* was 50% (24 of 48 plantings) in the 18 individuals per quadrat density, and 54.16% (13 of 24 plantings) in the 9 individuals per quadrat density. The similar shoot population restoration rate of *M. racemosum* subsp. *racemosum* in these two treatments demonstrates that the selected density treatments did not have a significant effect on this species.

The optimal density and spatial arrangement is known for few species (Drayton and Primack 2000). This study used lower planting densities in both treatments than the densities found to have

suppressive effects on invasive species by Murphy (2005). The lower planting densities used in this study could be one reason density was not a statistically significant factor; the suppression of invasive species seen by Murphy (2005) may become evident only if higher densities of native species are used. Ecological restoration of the site was the main goal of this study, not testing for suppression of invasive species.

Rust and Roth (1979) reported that clonal density did negatively affect seedling mortality; despite similar germination rates of seedlings within clonal areas and outside of clonal areas, *P. peltatum* seedling mortality was greatly increased in areas within the clone as compared with seedling mortality outside of clonal areas. Rust and Roth (1979) reported this mortality was likely due to the shading effect of the large umbrella leaves of *P. peltatum*. This result supports the rationale for planting *P. peltatum* at lower densities when the main goal is to restore (to control for competition within clonal shoots) and for planting at higher densities when suppression effects are the main goal. Highly physiologically integrated clonal species such as *P. peltatum* (Whigham 2004) have been suggested to regulate shoot densities to minimize intraspecific competition (Pitelka 1984, cited in Whigham 2004). Ecological restorations using highly physiologically integrated clonal species should consider this aspect when selecting planting densities. The densities used in this study did not influence species restoration rates, and did not result in intraspecific competition.

Discussion of Block Effect

The repeated measures MANOVA showed that block was found to have a significant effect for the total shoot population (all species) in 2008; the significant block was Block 3 (see Figure 4.4b). The effect of Block 3 ceased to be significantly different as of June 19 2008, at which time the shoot population decline in all blocks due to natural senescence caused the block effect to no longer be significant.

Antonovics and Primack (1982) found significant differences in seedling survival of *Plantago lanceolata* among small adjacent quadrats placed in an area selected for its apparent uniformity, this led them to conclude environmental heterogeneity plays a subtle but important role in transplants' survival and mortality. The statistical significance of Block 3 in 2008 has similar implications for subtle environmental differences in the field site. Although the blocks were selected based on similar characteristics and were situated closely to one another (see Figure 3.9), shoot population survivorship was statistically significantly higher in 2008 in Block 3.

There are several possible explanations for the statistical difference of Block 3. Dense leaf litter on the forest floor has been correlated with a lack of herbaceous growth in temperate forests (Sydes and

Grime 1981; Whitney and Foster 1988; Singleton 1998). In some areas of the ecological restoration experiment (e.g. Blocks 1 and 2, Block 4 quadrats 5 and 6) leaf litter was noted to accumulate more than in other areas (personal observation), due to variations in local topography. This could partially explain why the restoration rate was higher in Block 3, where leaf litter was still present, but did not accumulate as in other areas of the ecological restoration site.

The existence of a block effect demonstrates the subtle differences that may exist within a seemingly uniform site, and supports the distribution of plants in cells (small native vegetation pockets, also referred to as nuclei) throughout the restoration site (Daigle and Havinga 1996). As the plantings mature, vegetative and sexual dispersal by the initial planting recolonizes the suitable habitat. This distribution technique allows for unforeseen effects of environmental heterogeneity to affect portions of the restoration population, but not the entire ecological restoration experiment.

Control Quadrat Responses

In both 2007 and 2008 seven of the eight control quadrats did not contain any of the transplant species at any time. The one control quadrat that did contain *M. racemosum* subsp. *racemosum* in both years had similar percent cover estimates in both years. Therefore it would not appear that the soil seed bank in the control quadrats contributed to the emergence of new shoot populations of the transplant species in the second year of this study. It is therefore also unlikely that the shoot populations observed in the density treatment quadrats were a function of the natural regeneration from the soil seed bank, and is more likely that the shoot populations observed in the density treatment quadrats were a direct result of the ecological restoration transplants.

Donor Plots

No significant differences in the number of stems present were seen for any of the restoration species. *Caulophyllum giganteum* and *M. racemosum* subsp. *racemosum* donor plots had a greater number of stems present in 2008 than in 2007. The number of *P. peltatum* stems present in 2008 was lower than the initial number of stems present in 2007, but was higher than the number of stems present after harvesting, indicating that this species did rebound following harvest. The number of sexual shoots of *P. peltatum* was higher in 2007 than in 2008. Sohn and Policansky (1977) discovered a decrease in future survivorship if *P. peltatum* successfully bears fruit. This could explain the decrease in overall *P. peltatum* shoot populations, if the sexual shoot population of 2007 successfully bore fruit.

Overall, no significant differences in the abundance of the transplant species were observed in the donor plots. The differences observed are likely a result of natural variation. While long-term monitoring

is necessary to observe long-term impacts, the data collected in this study provide benchmarks for monitoring in future years.

Discussion of Specific Species

With a growing focus on active management of protected areas and reintroductions of species, key areas for research include the basic biology of the introduced species, as well as research on restoration techniques which are useful given the species' biology (Drayton and Primack 2000). A discussion of the species specific biology and responses to this ecological restoration experiment follows, along with a discussion of how species specific traits may have influenced restoration success.

Podophyllum peltatum Linnaeus

Podophyllum peltatum Linnaeus (Berberidaceae) (FNA Ed. Comm. 1997, cited in Brouillet et al. 2010+a) is an herbaceous rhizomatous perennial. Its vernacular name is May-apple, although it may also be referred to as Indian-apple or wild mandrake (FNA Ed. Comm. 1997, cited in Brouillet et al. 2010+a). It is alternatively referred to as a spring ephemeral (Gilliam and Roberts 2003a; Constable et al. 2007), spring green (Hull 2002), and a summer-green perennial (Rogers 1983; Neufeld and Young 2003). Based on the six photosynthetic guilds Givnish (1987) established from leaf phenology (excluding winter annuals), *P. peltatum* is most accurately described as an early summer species. This guild displays a variety of "umbrella-like" structures, which maximize light capture and minimize shading "with the lowest possible structural costs" (Whigham 2004, p 585).

Podophyllum peltatum typically occurs in deciduous forest habitats that are moist, shady, and rich, but it may also be found in areas of higher light at the edges of forests, as well as in thickets, marshy meadows and ditches (Krochman et al. 1974; Small and Catling 1999). In Canada it is found frequently throughout Ontario south of the Canadian Shield, and in Quebec where it is rare and occurs along the upper St. Lawrence River (Small and Catling 1999). *Podophyllum peltatum* is commonly associated with the understory of beech-maple forests in southern Canada and the eastern United States (Constable et al. 2007).

Podophyllum peltatum grows up to 60 cm tall and has one to three (although usually one or two) deeply lobed, umbrella shaped leaves (Small and Catling 1999). The underground rhizomes from which the shoots emerge are thick (approximately 6 mm), branched, and up to 1 m long (Small and Catling 1999); rhizomes grow 9 to 12 cm per year and are highly physiologically integrated (Whigham 2004). *Podophyllum peltatum* reproduces through both vegetative and sexual reproduction; an individual plant consists of at least one aerial shoot (ramet) attached to the underground rhizome, which branches often.

Each ramet is either vegetative or sexual, and the difference is readily apparent; sexual ramets have two leaves of nearly equal size attached to the petiole, vegetative ramets have one leaf (Sohn and Policansky 1977).

New ramets emerge from the terminal (youngest) rhizome segment (or multiple segments in branched rhizomes) each year (Geber et al. 1997). From the base of the ramet grow new rhizomes, the ramet dies back at the end of the growing season, and a new ramet is produced the following year. The more branched the rhizomes, the more ramets will be produced. The amount of new rhizome growth is similar in sexual ramets with fruit and vegetative ramets, while sexual ramets without fruit produce larger rhizomes (Sohn and Policansky 1977).

The number of ramets and their type (vegetative or sexual) are determined by developmental events in the current growing season, as well as previous seasons; the shoot type for a rhizome system can change from year to year, altering its demographic status (Geber et al. 1997). Flowering of sexual plants occurs in May, and the fruit ripens in July and August (Small and Catling 1999). Canadian researchers (Laverty and Plowright 1988; Laverty 1992) reported that *P. peltatum* flowers do not produce nectar, and that proximity to magnet species such as flowering wood betony (*Pedicularis canadensis* L.) attracts pollinators and facilitates pollination. Fruiting and germination success rates are often low, 100% of the flowers failing to set fruit is not uncommon (Sohn and Policansky 1977), and less than 10% of sexual ramets typically produce fruit due to flower or ovary abortion (Geber et al. 1997). Seedlings of this species are uncommon (Sohn and Policansky 1977).

Podophyllum peltatum often occurs in colonies of up to 1,000 ramets; while there are fewer genomes than ramets the extent to which colonies are clones of one or more plants remains unclear (Sohn and Policansky 1977). Policansky (1983) demonstrated that *P. peltatum* has considerable variability in the degree of self-fertility over its range, and that it is not completely self-sterile, although some populations are self-sterile. Policanksy (1983) further determined that patches of *P. peltatum* are not always clones of a single genotype, as pollination within patches sometimes resulted in seeds, and pollinations between patches always did. Although he did not determine the total number of genotypes in each patch, at least two genotypes were present to allow for successful intra-patch pollination. *Podophyllum peltatum* is noted to be polygamodioecious in its breeding system and partially self-incompatible (Whigham 2004).

Podophyllum peltatum in this Ecological Restoration Experiment

The transplants of *P. peltatum* used in this ecological restoration experiment were harvested from several distinct and separate patches in Natchez Hills ESPA, therefore several genotypes could exist in the transplants. All of the transplants selected for use in the ecological restoration experiment were vegetative

ramets. Sohn and Policansky (1977) demonstrated that if *P. peltatum* successfully bears fruit, there is a decrease in future survivorship. Although not all sexual plants successfully bear fruit, the possibility that sexual ramets would bear fruit and be less likely to survive in future years merited their exclusion from the ecological restoration experiment. Although organ preformation for 2008 (the second field season) was influenced as early as 2005 (Geber et al. 1997), and was completed by the end of the 2007 growing season with the formation of an overwintering bud (de Kroon et al. 1991), it was not possible to know the type of ramet that would be present in 2008. Determining whether the 2008 ramet would be vegetative or sexual by examining the overwintering bud would have required digging around the rhizome, and possibly damaging the new bud, and would have been impossible at the time of transplant selection. Therefore the only reproductive selection made concerning *P. peltatum* transplants was to use vegetative ramets for transplanting in the 2007 field season.

Figure 5.1 Photo of *Podophyllum peltatum*

Podophyllum peltatum Shoot Population Survivorship in Season One

Podophyllum peltatum shoot population survivorship was above 80% through July 6 2007. *Podophyllum peltatum* experienced the most rapid senescence of shoot population in 2007 (Figure 4.1),

but the decline in the *P. peltatum* shoot population was not remarkably different than the decline of the *C. giganteum* shoot population. The shoot population senescence of *P. peltatum* observed in this ecological restoration experiment was more pronounced than the senescence in *P. peltatum* seen by Ferguson (2007), however that may be due to the low precipitation levels received during 2007.

***Podophyllum peltatum* Shoot Population Survivorship in Season Two**

Podophyllum peltatum was very successful in the second field season. Cumulative shoot population survivorship was 97.22% for *P. peltatum*, representing 70 of 72 possible shoot populations present at some point during the season. Since some *P. peltatum* shoot populations senesced before others had emerged, the number of shoot populations present on the peak dates in 2008 are lower than the cumulative number present (Figures 4.2 and 4.3b).

Overall the *P. peltatum* shoot population used in the ecological restoration exhibited a strong restoration success rate in 2008. The *P. peltatum* shoot population was above 80% from April 23 2008 through June 12 2008. After June 12 2008 the *P. peltatum* shoot population senesced gradually, in contrast to the rapid decline observed in 2007. The shoot population remaining near the end of July in 2008 was 10% greater than the shoot population remaining one year previous. The restoration success rate observed for the *P. peltatum* shoot population in this ecological restoration was identical to the restoration rate reported for *P. peltatum* in Natchez Hills ESPA by Ferguson (2007). Both studies used 72 *P. peltatum* transplants, of which 70 returned in the second field season.

Discussion of *Podophyllum peltatum* Response

The approximate time from emergence to senescence for *P. peltatum* is twelve weeks (Constable et al. 2007). This is similar to the growth patterns observed in the transplanted populations of *P. peltatum*, indicating that as a population, the transplanted individuals emerged and senesced within the normal timeframe for a *P. peltatum* population, in the year following transplanting.

Natural populations of *P. peltatum* have long-lived rhizomes, with up to seven years of previous rhizomatous growth maintained by each subsequent aerial shoot (de Kroon et al. 1991). De Kroon et al. (1991) hypothesize that the costs of maintaining segments of older rhizome are outweighed by the benefits of increased survival and growth of the clone. Since the transplanted *P. peltatum* have severed rhizomes (a result of being transplanted), they have less rhizomatous growth and associated stored resources upon which to rely. Whether this will affect the survival of the transplanted shoot populations in future years remains to be seen.

Podophyllum peltatum emergence and senescence varies as a function of seasonal rainfall patterns; late season drought and high rainfall early in the growing season both seem to accelerate the onset of senescence (Watson and Lu 1999). The 2007 growing season was characterized by lower than average precipitation, especially from June through October 2007 (Environment Canada 2007b, 2010b, 2010c). Therefore, the more rapid senescence of the *P. peltatum* shoot population in this ecological restoration in 2007, as compared to the senescence witnessed by Ferguson (2007) in 2006, is not surprising. Water is often a limiting factor in transplant stress, and the very low precipitation levels in 2007, coupled with the non-watering methodology used in this ecological restoration experiment may have contributed to a more rapid senescence of the *P. peltatum* transplants.

Watson and Lu (1999) observed delayed senescence of nearly two months in their second field season as compared to their first field season; the second field season was wetter and more similar to an average year, while the first season was characterized by an early and persistent drought. These are very similar precipitation conditions to those experienced during this experiment; 2007 was a year of low precipitation, whereas 2008 was slightly wetter then the average (Table 4.1). The more gradual senescence of the *P. peltatum* population in 2008 indicates a similar response by the *P. peltatum* shoot population in this experiment to that observed by Watson and Lu (1999) (Figure 4.4b).

Severe wind and rain events occurred in Natchez Hills ESPA in 2007, but the *P. peltatum* shoot populations visibly affected by these events in 2007 were not damaged to the extent that survivorship in 2008 decreased. Similarly, Sohn and Policansky (1977) observed *P. peltatum* shoots that had been destroyed by severe rains and winds, including large patches that had been flattened by rain; these plants were also observed to produce new rhizomes. The ability of *P. peltatum* to withstand extreme weather events and continue rhizome growth increases its suitability as a restoration species; severe thunderstorm events are known to occur in southern Ontario.

Dispersal and Reproduction of Podophyllum Peltatum

Through excavation of *P. peltatum* rhizome systems Rust and Roth (1981) determined that seven years appeared to be the minimum age for flowering in *P. peltatum*, and that a plant germinated from seed remains as a single stem for at least five years before rhizome growth begins. Therefore, the minimum age at which the life cycle would be complete and sexual reproduction in second generation seedlings could occur would be at least 12 years (Rust and Roth 1981). In contrast, the mature plant material used in this ecological restoration experiment produced shoot populations which were observed to both flower and exhibit dispersal via rhizome growth in only the second year of the study. Therefore transplanting adult individuals from established populations of *P. peltatum* can significantly reduce the amount of time

before vegetative and sexual reproduction occurs. This evidence further supports the use of transplanted adult individuals of *P. peltatum* as opposed to seedlings, which are also considered difficult to propagate (Kim et al. 2007). The successful transplant rate achieved for *P. peltatum* in this early ecological restoration demonstrates that transplants provide a high restoration rate, in the short term. Long-term monitoring will need to be undertaken to determine if this is a sustained trend.

Vegetative Dispersal of Podophyllum Peltatum

Vegetative dispersal will only increase the shoot population if the rhizome from which the aerial shoots originates branches (Watson and Lu 1999). In 2008 multiple aerial shoots of *P. peltatum* were observed (see Figure 5.2); since no sexual plants were used as transplants in 2007 any increase in shoot populations were the result of vegetative dispersal. The additional shoots observed in this ecological restoration experiment provide evidence that the branching rhizomes contained enough resources to sustain multiple aerial shoot populations, despite transplant shock and the other stresses. Although the increase in the *P. peltatum* population size from multiple aerial shoots was not measured quantitatively in the presence/absence reporting of restoration success, the multiple shoots are an important indication that the transplanted rhizomes are becoming established at the restoration site, and that the transplanted rhizomes are able to allocate resources between branches. Further monitoring of *P. peltatum* at the restoration site will clearly indicate if population expansion is continuing.

Figure 5.2 Multiple vegetative shoot populations of *Podophyllum peltatum* emerging at one planting site in 2008.

Sexual Reproduction of Podophyllum Peltatum

In 2008, 17 of the 72 *P. peltatum* shoot populations (23.61%) were sexual shoots. The occurrence of sexual populations in the second year of the ecological restoration, despite the fact that no sexual plants were used in the first year is an indication that the rhizomes contained sufficient resources to sustain the transplant, as well as the preformed sexual shoot. Within the sexual shoots that appeared, some of the plants successfully set fruit, while other fruits were aborted. As Sohn and Policansky (1977) described, sexual plants which successfully set fruit have a decrease in future survivorship. Future monitoring could determine if this trend holds true in the restoration site.

The extent to which the fruits of the *P. peltatum* sexual shoots produced viable seeds was not explicitly tested in my study. The *P. peltatum* populations used in the ecological restoration were taken from several distinct patches in Natchez Hills ESPA; therefore although future sexual reproduction is not guaranteed, the presence of more than one genotype means conditions for successful pollination within the restoration site do exist. Although interclonal pollination does not guarantee successful sexual reproduction, the proximity of populations of different genotypes does facilitate the possibility of successful pollination, and the production of a viable seed set.

Overall the population of *Podophyllum peltatum* in the restoration blocks will be more likely to increase in shoot population size through vegetative dispersal than through sexual reproduction; seedling mortality in clones has been shown to be very high, as demonstrated by the 98.7% mortality rate after one year recorded by Rust and Roth (1981). Even outside of clones, seedling mortality is quite high. By the third year after seed planting, seedling mortality within the clones was 99.1%, and seedling mortality outside of the clones was 92.4% (Rust and Roth 1981).

***Maianthemum racemosum* subsp. *racemosum* (Linneaus) Link**

Maianthemum racemosum subsp. *racemosum* (Linneaus) Link (Asparagaceae) (FNA Ed. Comm. 2002, cited in Brouillet et al. 2010+c) (also placed in Convallariaceae, Liliaceae and Ruscaceae) (USDA 2011) is a herbaceous perennial summer herb (DeMars and Boerner 1997; Anderson 2003). Its accepted vernacular name is large false Solomon's seal, but it may be called one of seven other vernacular names, which are considered synonyms. It was formerly called *Smilacina racemosa* (LaFrankie 1986) and still referred to as such by many authors, eleven synonyms for the accepted name exist[5]. *Maianthemum racemosum* subsp. *racemosum* has been referred to as a summer-green (Neufeld and Young 2003), and based on Givnish's photosynthetic leaf guilds it is best termed an early summer species (Givnish 1987).

[5] For a full listing of the synonyms for the accepted and vernacular names of *Maianthemum racemosum* subsp. *racemosum* see Brouillet et al. 2010+c

Maianthemum racemosum subsp. *racemosum* occurs in mesic forests of southern Canada and the United States (Piper 1989), generally emerging in late April to early May, flowering throughout the spring, and senescing in August and September (DeMars and Boerner 1997). It has an arching, zigzag stem along which the alternate leaves (none of which are basal leaves) are displayed (Newcomb 1977; Givnish 1986).

Maianthemum racemosum subsp. *racemosum* reproduces vegetatively as well as through seed production, but clonal reproduction via rhizome growth is the main mode of reproduction and dispersal. *Maianthemum racemosum* subsp. *racemosum* is described as hermaphrodite in its breeding system, and seeds are apomictic (Bierzychudek 1982; Roy and de Blois 2006). Rhizome growth is approximately 1 to 2 cm per year (Whigham 2004). *Maianthemum racemosum* subsp. *racemosum* tends to occur in discrete patches, often with many ramets from the same clonal rhizome (Piper 1989), although physiological integration has not been observed (Whigham 2004). Ramets produce panicles with small white flowers (from 0 to 250 flowers), and the small, fleshy fruits ripen in early September. Seeds are dispersed by birds and the recruitment rate is low (Piper 1989), the main reason clonal reproduction is more important for perpetuating populations.

Figure 5.3 Photo of Maianthemum racemosum subsp. racemosum

Maianthemum racemosum subsp. racemosum in this Ecological Restoration Experiment

The individuals of *M. racemosum* subsp. *racemosum* used for transplanting in Natchez Hills ESPA were not selected based on whether they would flower or fruit for two main reasons. The first and most important reason was that flowering and fruiting was not always apparent, as transplanting was completed soon after *M. racemosum* subsp. *racemosum* emergence and early in plant development. The second reason was that since seed production in *M. racemosum* subsp. *racemosum* is asexual, fruiting and flowering did not represent potential for interclonal reproduction. Some transplanted individuals did later produce flowers and fruit, but this was not a criterion for transplant selection for this ecological restoration.

Maianthemum racemosum subsp. racemosum Shoot Population Survivorship in Season One

The *Maianthemum racemosum* subsp. *racemosum* shoot population had the most gradual senescence of the three transplant species in 2007. Shoot population survivorship was above 85% through July 23 2007. Shoot population survivorship declined more rapidly after July 23 2007, but was still above 20% on the final sampling date of October 3 2007. Although all three species used in this study were early summer species (see Givnish 1987), *M. racemosum* subsp. *racemosum* was observed to emerge and flower later than *C. giganteum* and *P. peltatum* in Natchez Hills ESPA, including at the reference site. Therefore it is not surprising that this species senesced last, as phenologically it was the latest of the restoration species.

Maianthemum racemosum subsp. racemosum Shoot Population Survivorship in Season Two

Maianthemum racemosum subsp. *racemosum* had a cumulative shoot population presence of 51.39% (37 of 72) in 2008; this is a lower restoration rate than both *C. giganteum* and *P. peltatum.* This restoration rate was lower than expected, given the successful transplanting and long-term persistence of the *M. racemosum* subsp. *racemosum* shoot population in 2007, and following the successful emergence of the other two transplant species shoot populations in 2008.

Discussion of Maianthemum racemosum subsp. racemosum Response

Although the *M. racemosum* subsp. *racemosum* population had the lowest restoration success rate of the three species used, the restoration success rate of *M. racemosum* subsp. *racemosum* was successful compared to other ecological restorations. In a restoration experiment in eastern North America using adult plant material, seven of eight perennial plant species had lower rates of survival two years after introduction (Drayton and Primack 2000). Roy and de Blois (2006) found that certain species, including *M. racemosum* subsp. *racemosum,* were able to disperse more readily into hedgerows than other forest understory species such as *C. giganteum*, possibly demonstrating a better ability to recolonize after

disturbances. Similarly Singleton (1998) found that on average *M. racemosum* subsp. *racemosum* (*Smilacina racemosa*) was present more frequently in post-agricultural forests, as opposed to old woods. Future long-term monitoring of the restoration site will be necessary to determine how *M. racemosum* subsp. *racemosum* responds at the disturbed site in Natchez Hills ESPA.

Maianthemum racemosum subsp. *racemosum* shoot populations at the restoration site were observed to be affected by a number of factors, including severe rain and wind events, herbivory, and low precipitation levels, in addition to the effects of transplant shock. The *M. racemosum* subsp. *racemosum* shoot populations transplanted into this restoration site were subject to herbivory in both field seasons, but only two of the eight *M. racemosum* subsp. *racemosum* shoot populations browsed in 2007 (11%) did not produce shoot populations in 2008. Since the subset of browsed *M. racemosum* subsp. *racemosum* shoot populations returned at a higher rate than the overall *M. racemosum* subsp. *racemosum* shoot population, herbivory did not appear to negatively affect restoration success of this species in the ecological restoration experiment.

Reproduction and Dispersal of Maianthemum racemosum subsp. racemosum

The *M. racemosum* subsp. *racemosum* shoot population did show evidence of dispersal in the second field season; multiple shoots were present at six of the planting locations in 2008 (Figure 5.4). In 2007 two of these six shoot populations had produced fruit, although the fruit were few in number and unhealthy looking (unlike some other *M. racemosum* subsp. *racemosum* shoot populations in the ecological restoration which produced more than 40 healthy looking fruit). Therefore it is more likely that the dispersal seen in the six plantings was due to vegetative dispersal via rhizome growth, rather than sexual reproduction through seedling growth.

Production of fruit in 2007 did not appear to influence survival rates in 2008. 47.37% of the *M. racemosum* subsp. *racemosum* shoot populations which produced fruit in 2007 were present in 2008 (9 out of 19), similar to the 2008 return rate for the entire *M. racemosum* subsp. *racemosum* shoot population. Of the *M. racemosum* subsp. *racemosum* shoot populations present in 2008, 5.4% (2 of 37) produced flowers, and 2.7% (1 of 37) produced fruit. Further study of the reproductive phenology of *M. racemosum* subsp. *racemosum* would determine the extent to which flowering and fruiting affect survival, or whether these measures are a good indication of establishment at the restoration site.

Figure 5.4 Multiple aerial shoots of *Maianthemum racemosum* subsp. *racemosum* emerging at one planting site in 2008.

***Caulophyllum giganteum* (Farwell) Loconte & W.H. Blackwell**

Caulophyllum giganteum (Farwell) Loconte & W.H. Blackwell (Berberidaceae) was previously classified as *Caulophyllum thalictroides* var. *giganteum*. *Caulophyllum thalictroides* (Berberidaceae) has in the past been classified as having two varieties, *C. thalictroides* (L.) Michx. var. *thalictroides* and *C. thalictroides* var. *giganteum* Farwell. The flowers of *C. thalictroides* (L.) Michx. var. *thalictroides* are yellow, whereas those of the *giganteum* variety are purple (Small and Catling 1999). *Caulophyllum thalictroides* var. *giganteum* Farwell is the more northern of the two varieties (Hannan and Prucher 1996). *Caulophyllum thalictroides* var. *giganteum* Farwell was the variety used in this study, it is referred to throughout this thesis by its accepted name, *Caulophyllum giganteum* (FNA Ed. Comm. 1997, cited in Brouillet et al. 2010+b). *Caulophyllum thalictroides* var. *giganteum* Farwell is listed as the synonym for the accepted name *C. giganteum*. The vernacular name for this species is giant blue cohosh, a synonym is blue cohosh (Brouillet et al. 2010+b).

Caulophyllum giganteum is a summer-green perennial species (Neufeld and Young 2003) or an early summer species (Givnish 1987), well-known to botanists in eastern North America. It is one of the earliest forest species to flower in the spring (Hannan and Prucher 1996), and in the late summer its blue

seeds are quite recognizable. It is native to eastern North American and can be found from the Canadian Shield south to Georgia (Brett 1981). *Caulophyllum giganteum* can be found in deciduous and mixed forests throughout its range; in the northern part of its range forests of beech-maple (*Fagus grandifloia* Ehrh., *Acer* spp.) are common associations (Brett 1981). Rich, wet to mesic shady woods are the preferred habitat; in Canada *C. giganteum* is most frequently found in maple woods on limestone, in rocky calcareous and organic substrates (Small and Catling 1999). Herbaceous species associations of *C. giganteum* include spring plants common to eastern North American deciduous and mixed forests, and include *Hepatica* spp., *Trillium* spp., *Viola* spp., *Claytonia* spp., *Dicentra* spp., *Podophyllum peltatum* L., *Sanguinaria canadensis* L., *Asarum canadense* L., *Allium* spp., *Erythronium americanum* Ker., *Actaea* spp., *Thalictrum* spp., *Tiarella cordifloia* L., and *Mitella diphylla* L. (Brett 1981). *Caulophyllum giganteum* is harvested from the wild in some areas of North America, as a result it is at risk from overcollecting in some areas (Small and Catling 1999).

Caulophyllum giganteum is an herbaceous perennial with an elongated rhizome from which erect stems emerge. The rhizome is linear, but lateral branching may occur; the rhizome surface is covered with numerous thickened roots (Brett 1981). Shoots from the rhizome are either vegetative or flowering, and emerge early in the spring (Hannan and Prucher 1996). *Caulophyllum giganteum* flowers each produce one or two large, naked, distinctly blue seeds, which often fall close to the parent plant. In many populations, individual plants are observed growing closely clumped together, and are often mistakenly thought to be shoots of one plant, until they are dug up and determined to be two to three individuals. This is due to both the close proximity of seedling growth based on seed fall, as well as the decaying of rhizomes into several portions, which then continue to grow as separate individuals (Brett 1981).

Vegetative reproduction in *C. giganteum* is thought to be more important than sexual reproduction based on a low sexual reproductive output; this is consistent with the reproductive strategies of many perennial herbaceous plants (Harper 1977, cited in Brett 1981; Jolls 2003). Varieties of *C. giganteum* are considered self-compatible and are classified as hermaphrodites in their breeding systems, however *Caulophyllum giganteum* is self-sterile to a great extent than *Caulophyllum thalictroides*, (Hannan and Prucher 1996; Whigham 2004). Self-pollination resulted in smaller seed sets in *C. giganteum* as compared to *C. thalictroides* (Hannan and Prucher 1996).

Caulophyllum giganteum in this Ecological Restoration Experiment

Since the vegetative and sexual forms of *C. giganteum* are not as apparent as in *P. peltatum*, transplants of *C. giganteum* used in the ecological restoration were both vegetative and sexual, as evidenced by flowers present on transplants after transplanting was completed. Literature on *C.*

giganteum does not indicate a difference between survival of vegetative and sexual plants, therefore this was not expected to influence ecological restoration success rates for *C. giganteum*. *Caulophyllum giganteum* in Natchez Hills ESPA was found to grow in groups of several stems close together (as noted in Brett 1981); in order to protect plant and rhizome health these close growing stems were not forcibly separated, and were transplanted as one transplant. Stem counts for *C. giganteum* transplants ranged from one stem to six stems per transplant.

Figure 5.5 Photo of *Caulophyllum giganteum*

Caulophyllum giganteum Shoot Population Survivorship in Season One

Caulophyllum giganteum shoot population survivorship remained above 80% through July 6 2007. From July 23 2007 to August 10 2007 30% of the population senesced, and from August 10 2007 to August 26 2007 the population again decreased, from 47% present to 3% remaining. This represents an earlier die off than Ferguson (2007) experienced following transplanting of *C. giganteum* in an ecological restoration in the same ESPA one year earlier, but it is probably not entirely surprising given the low levels of precipitation experienced during 2007.

Caulophyllum giganteum Shoot Population Survivorship in Season Two

The maximum *C. giganteum* shoot population survivorship was 98.61% (71 of 72 possible plantings present); this maximum population was sustained from May 12 2008 though June 25 2008. The shoot population survivorship of *C. giganteum* in 2008 was very successful, and could even be described

as more successful when compared with the 2007 shoot population of *C. giganteum* (see Figure 4.3a). The shoot population survivorship rate from 2008 demonstrates that in the short term, *C. giganteum* is capable of withstanding transplant shock, herbivory and low precipitation levels, and returning in healthy shoot population numbers in the subsequent year. Continued monitoring will be required to determine if this is a sustained trend for *C. giganteum*.

Discussion of *Caulophyllum giganteum* Response

The ecological restoration rates observed for *C. giganteum* were very high and were comparable to those observed by Ferguson (2007). Singleton (1998) observed survival rates of between 24% for transplanted *C. giganteum* rhizomes into post-agricultural forests, to approximately 45% in old woods, two to three years after transplanting. No other ecological restoration experiments reporting the use and restoration rates for *C. giganteum* were found.

Reproduction and Dispersal of Caulophyllum giganteum

Shoot populations of *C. giganteum* are naturally present in groups of dense, multiple stems, making dispersal measurement for this species more difficult than for species with single stem shoot populations. Further confusing conclusive dispersal evidence is the fact that although *C. giganteum* can reproduce prolifically via vegetative dispersal, rhizome growth is often limited to a few centimetres per year (Singleton 1998). Similarly, *C. giganteum* seeds generally fall close to the parent plant, making new seedling growth difficult to distinguish from the parent plant. Small dispersal distances and multiple stems grouped together may have counted multiple shoot populations as one shoot population, reducing dispersal estimates. No conclusive evidence of dispersal was noted for *C. giganteum*.

Seedling germination in 2008 from seeds present in 2007 is a possibility, since several plantings did produce fruit in 2007. However, given that *C. giganteum* reproduces more successfully via vegetative dispersal than sexual reproduction (Brett 1981), any new *C. giganteum* shoot populations observed in 2008 were more likely to be a result of vegetative dispersal. Vertebrate dispersal is the more common dispersal mechanism for the seeds of *C. giganteum* (Singleton 1998), however, any vertebrate dispersed seeds which germinated were unlikely to have remained in the restoration quadrat, and would therefore not be counted in dispersal estimates.

Singleton (1998) discusses the possibility that stored reserves and the preformation of buds in *C. giganteum* in the year prior to transplanting may overinflate the true results for species responses to a transplant site, in the short term. Since the previous year determines preformation, true results for the

restoration success rate may not be realised until two years following the ecological restoration. Future monitoring of *C. giganteum* at the restoration site will be required to explore this idea.

Comparison of Ecological Restoration Experiment Results to Other Studies

For *P. peltatum* and *C. giganteum*, the species responses obtained in this study were comparable to those seen by Ferguson (2007). Ferguson (2007) reported restoration rates between 92% and 100% for three native herbaceous forest understory species in Natchez Hills ESPA; one generous watering was a component of the methodology followed in Ferguson's study. The restoration rate observed for *C. giganteum* in this study (98.61%) was higher than the 24% to 45% restoration rate observed by Singleton (1998) in central New York. In the Boston area Drayton and Primack (2000) reported response rates for adult plant material which varied from 1.25% to 51.4% for eight perennial species two years after transplanting, with a total success rate across all species of 16.6%. One species of the adult plant material (*Hedyotis*) in this experiment was watered once soon after transplanting because of unusually dry conditions, this species had the lowest (1.25%) survival rate after two years. No other transplants in their experiment were watered, nor was aftercare provided, similar to the practical methodology used in this ecological restoration. In Iowa, the mean survival rate one year after transplanting was 91% for 19 woodland perennial species for Mottl et al. (2006), individual species restoration rates varied from 42.1% to 100%. However, these plantings received much aftercare, which was not similar to the methodology used in this study. Plants transplanted during a rainy period were not watered, while others added later in the summer were watered. Other aftercare in the Mottl et al. (2006) ecological restoration included weeding of plots, to eliminate competition from weedy species.

Lefler (2006) reported on several ecological restoration experiments in the Region of Waterloo, including on previous restoration work in the forest understory in Natchez Hills ESPA. Lefler (2006) found the restoration plantings from 2003 did not appear to be thriving or expanding, and no woody regeneration was noted; a more intensive planting regime was recommended. In contrast this ecological restoration noted high restoration response rates, and dispersal of the restoration species in several of the quadrats in the first two years of the study.

These studies, many of which used adult plant material as transplants, demonstrate that the 82.41% overall restoration rate and the individual restoration rates for each species were successful. Longer-term monitoring is needed to provide further information about the long-term success of the ecological restoration experiment.

Suitability of Restoration Species for Future Ecological Restorations

Based on the excellent restoration rate observed for the shoot populations of *P. peltatum*, as well as the early indications that the initial introduced population may be expanding, *P. peltatum* is a reasonable species to use in ecological restorations in the City of Kitchener. The absence of any predation on this species further reinforces the use of *P. peltatum* in future restorations. Similar results were obtained for this species by Ferguson (2007) in the same ESPA, further supporting the future use of *P. peltatum* in forested areas of similar composition in the City of Kitchener.

The shoot populations of *C. giganteum* in this ecological restoration were also observed to have an excellent restoration rate. Although this species was subject to herbivory, it did not impact the shoot population restoration rate in 2008. Similar to *P. peltatum*, Ferguson (2007) also reported high restoration return rates for *C. giganteum* in this upland forest, supporting its use as a restoration species in City of Kitchener.

Maianthemum racemosum subsp. *racemosum* experienced the lowest restoration success rate of the three species used in this study, however in comparison with restoration rates from other studies using perennial herbaceous species *M. racemosum* subsp. *racemosum* was restored with good success. *Maianthemum racemosum* subsp. *racemosum* was subject to herbivory in Natchez Hills ESPA, however, in the short-term the impacts of herbivory did not appear to adversely impact the restoration shoot populations. Longer-term monitoring of the restoration site is needed to determine if herbivory is negatively affecting the *M. racemosum* subsp. *racemosum* shoot populations, and to determine if future restorations using this species should require exclosures or other measures to combat herbivory, or if this species should be excluded from some restoration sites in the City of Kitchener.

Opportunities for Future Research

Long-term Monitoring of the Ecological Restoration Site

Visits recording presence or absence of shoot populations could be conducted on a relatively infrequent basis (e.g. biweekly, monthly, or once per season), although more frequent visits may be preferable at the time shoot populations are emerging. Simple vegetative monitoring techniques, such as presence/absence measures, are more accurate and have reduced observer effects since they require low personal judgement on the part of the observer (Korb and Fulé 2008). These types of measurements are recommended when observers are untrained. The information recorded through these observations can still provide important information on the overall state of the ecological restoration experiment through time.

Frequent monitoring is not necessary to obtain basic survivorship data, and less frequent monitoring (e.g. as infrequent as once every 7 years as in Menges 2008) can still provide information about the ecological restoration if resources for extensive monitoring are not available. In order to separate year-to-year variability from treatment effects, long-term sampling should be conducted during the same phenological time frame each year and over a short amount of time. The sampling period should be appropriate given the research objectives, and feasible with respect to personnel and financial constraints (Korb and Fulé 2008).

The survival of reintroduced plant material is only the first level of success for an ecological restoration; expansion and reproduction of native species populations represent further progress along successional trajectories, and indicate if restoration progress is indeed successful (Drayton and Primack 2000; Lefler 2006). Completion of the life cycle (through stages such as fruiting, flowering, dispersal and seedling recruitment for many vegetative species) is a benchmark that allows for greater conclusiveness of the endurance of transplanted species (Menges 2008). The use of adult plant material in this ecological restoration meant that flowering and fruiting of mature shoot populations was observed, however, this could be the primary focus of future studies. During the two years of this study monitoring was undertaken frequently to provide information on plant phenology as well as survivorship. Future monitoring of the response of individual shoot populations on a frequent (e.g. weekly) basis would allow for phenological data to be assessed over the long-term. In such a monitoring program, information on vegetative and sexual shoots of *P. peltatum* could be collected, as well as information on flowering and fruiting in all three species. The viability of seed sets was not examined, and provides potential for another future experiment at the restoration site.

Dispersal of the transplant species within the restoration quadrats was observed, however, a longer monitoring timeframe is needed to observe whether there will be widespread dispersal of these species at the restoration site. Future studies could monitor if the transplanted individuals disperse further, and produce shoot populations which are viable and self-sustaining populations. This would indicate if the restoration quadrats are indeed acting as cells for the dispersal of the study species (Daigle and Havinga 1996). If the restoration species are effectively reproducing and dispersing, the early restoration success indicated by the results of this study can be expanded to indicate progress towards longer term restoration success.

The impact of herbivory in Natchez Hills ESPA deserves further monitoring. The occurrence of herbivory in both field seasons was unexpected, given that this was not recorded in Ferguson (2007) or Lefler's (2006) observations of herbaceous species in Natchez Hills ESPA. The methodology used did not

involve handling of the restoration species, and from a methodological perspective there should not have been a increase or decrease in the presence of herbivory due to experimenter interaction.

Although many of the individual restored shoot populations were not browsed, browsing of *C. giganteum* and *M. racemosum* subsp. *racemosum* was evident in the restoration quadrats for both species, and in the donor plots for *M. racemosum* subsp. *racemosum.* The degree of herbivory observed varied from defoliation of a small area of the plant (as was often the case in shoot populations of *C. giganteum*) to complete removal of the top of the plant and all leaves, (as observed for some shoot populations of *M. racemosum* subsp. *racemosum,* see Figure 5.6). Defoliation may negatively impact reproductive performance of some species, but these effects are often delayed and not significant until defoliation is at least 50% (Rockwood and Lobstein 1994). The greater the amount of tissue removed, the greater the impacts on growth and reproduction. Partial defoliation may have no impact, whereas complete or partial leaf removal can cease or decrease flowering (Rockwood and Lobstein 1994; Whigham 2004). Of the 11 shoot populations browsed in 2007, two did not produce shoot populations in 2008. In the short term herbivory does not appear to be significantly impacting the ecological restoration. However, the long-term effects of herbivory cannot be determined without further monitoring.

Maianthemum racemosum subsp. *racemosum* was one of several herbaceous species heavily grazed by *Odocoileus virginianus* Zimmerman (white-tailed deer) at a nature preserve in Illinois, before the deer population was reduced (Frankland and Nelson 2003). Kraft et al. (2004) note that several species of the Liliaceae family are heavily browsed by deer, including *M. racemosum* subsp. *racemosum.* Increased abundance of native wildlife that have a negative impact on forest understory herbs has been a contributing factor in the decline of a number of woodland herbaceous species (Whigham 2004). The extent to which forest understory herbivory is an issue in Natchez Hills ESPA is unknown. However based on the data from this study indicating that herbivores fed on this species, the use of *M. racemosum* subsp. *racemosum* in future ecological restorations may need to involve exclosures at the restoration site.

Future monitoring of both the *M. racemosum* subsp. *racemosum* and *C. giganteum* populations are necessary to determine the extent of the impacts of herbivory on these species. Future studies could also examine if fruiting and flowering plants were browsed more heavily, a variable that was not examined in this study. Although de Kroon et al. (2006) reported herbivory of *P. peltatum* in Maryland, *P. peltatum* was not subject to herbivory in this experiment, and is known to be avoided due to its toxic properties. Based on this study and Ferguson (2007) future ecological restorations in upland forest areas in the City of Kitchener could use this species without using exclosures. Other native forest understory herbaceous species which are not browsed by herbivores offer other possibilities. Future work could

identify the suitability of these species in early ecological restorations. Methodologies which do not involve touching the herbaceous plants should be used or gloves should be worn, given the uncertainty about and conflicting results on a herbivory effect.

Figure 5.6 Browsed *Maianthemum racemosum* subsp. *racemosum* Shoot Population in the Ecological Restoration Site in 2008.

Density Treatments in Future Ecological Restoration Experiments

While density was not found to have a statistically significant effect on species' shoot population survivorship, the impact of density as a treatment should not be dismissed from future restorations. Murphy (2005) demonstrated that the native perennial herb, *Sanguinaria canadensis* could suppress the establishment of the invasive species *Alliaria petiolata* in densities as low as five individuals of *S. canadensis* per square meter. Both my study and Ferguson (2007) demonstrated that *P. peltatum* is a suitable species for use in ecological restorations in upland maple beech hardwood forests in Kitchener, Ontario. Further research could examine if the umbrella like shape of *P. peltatum* (which is structurally similar to *S. canadensis*) could be used to control invasive species such as *A. petiolata*. In Natchez Hills

ESPA *S. canadensis* was observed to emerge slightly earlier than *P. peltatum*, the use of the two species together may be worth investigating through future research.

Watering and Drought

Watering at remote or inaccessible restoration sites is not a practical aspect of restoration, nor one which would often be undertaken. Using a non watering methodology this study demonstrated that *C. giganteum, P. peltatum,* and *M. racemosum* subsp. *racemosum* were capable of producing shoot populations in the second field season, even when subjected to low precipitation in the transplant year.

Herbaceous foliage is often poor in structural polysaccharides (lignin), the source of structural stability during periods of water stress (Muller 2003). Since the transplanted shoot populations were not watered at any time, the lack of precipitation in the first field season would have adversely impacted the transplanted shoot populations. The fact that the transplanted rhizomes were able to persist and produce new shoot populations in the second field season demonstrates that even in periods of low precipitation, ecological restorations of native perennial understory herbaceous species can be successful.

It is worth noting that this ecological restoration was successful, despite the fact that the period of low precipitation occurred during the same time period that the transplants could be experiencing transplant shock. The high return rate of *C. giganteum* and *P. peltatum* despite these conditions are especially promising. The use of soil clods may have decreased transplant shock, increasing the number of shoot populations present in the second field season. Since all of the transplants in this ecological restoration were completed using soil clods, testing for a soil clod effect is not possible. However, Ferguson (2007) did not use soil clods for *P. peltatum* (but did use soil clods for *C. giganteum*), and still observed high restoration success rates for both species. Future work in understory forest restoration could compare the use of soil clods with bare root transplants, to determine if there is an impact in early ecological restorations.

Transplanting as an Ecological Restoration Technique

Translocations of species are often necessary to install key species, and to aid in the recovery of species diversity (Menges 2008). Although wild-digging is generally discouraged (Daigle and Havinga 1996), in certain cases transplanting may be an acceptable and highly effective means for carrying out ecological restorations. Transplanting is one method of obtaining plant material which can address several of the practical concerns associated with ecological restorations. Transplanting can minimize costs for plant materials and dispel uncertainty regarding suitability of the materials for the site, if the transplant

source site is known and comparable to the ecological restoration site. When using transplants, genetic sources can be from as near or far as determined suitable (e.g. local or regional).

Transplanting provides source material for native forest understory species with reproductive characteristics that make establishment from seed a difficult or slow process, for example, those species that reproduce mainly via clonal growth. Many forest understory species are difficult to re-establish from seed, making transplants an effective strategy for reintroducing species to degraded sites (e.g. Drayton and Primack 2000; Mottl et al. 2006; Ferguson 2007). Results from reintroductions of perennial plant species have indicated that seedling and adult materials are more successful than sowing seeds, and that larger adult material responds with greater success than seedlings (Drayton and Primack 2000). If transplants are successfully established they can be used to bypass the high mortality associated with the seed to seedling stage, and reduce the time required before plants are able to begin spreading through the restoration site, vegetatively and by seed (Mottl et al. 2006). Certain native herbaceous forest understory species, such as *P. peltatum* keep older nodes on the rhizome segment alive, likely in order to enable the rhizome system to recover from damage to forward segments (de Kroon et al. 1991). The regeneration capacity of *P. peltatum* following removal of rhizome segments has been found to be very good (de Kroon et al. 1991), making transplanting an effective method of acquiring adult plant material without destroying the parent plant. Other clonal herbaceous species may exhibit similar regenerative capacities.

While some argue against using transplants (e.g. Fahselt 2007) for fear intact natural areas will be destroyed by those who claim that they can simply be moved elsewhere, ecological restoration experiments demonstrate that it is impossible to precisely recreate pre-disturbance conditions. Ecological restorations are viewed by practitioners and theorists as a tool against degradation, not as a means to move entire ecosystems. Generalizations against using transplants limit our ability to determine the suitability of various species for use in ecological restoration experiments, unless substantial budgets for plant materials are available. This decreases the probability that ecological restorations will be undertaken, contributing less knowledge to the field of restoration ecology from new experiments. Data concerning the effectiveness of transplants as a strategy for reintroduction is largely lacking, despite the possible contribution of this technique to reintroduction and restoration strategies (Maunder 1992).

In this experiment the rhizomes of transplanted species may have provided stored resources to the transplant, offsetting both transplant shock and unfavourable conditions in the transplant site (de Kroon et al. 1991; Singleton 1998). It is possible that stored resources are part of the reason the transplants were able to survive the extremely low precipitation levels in 2007. A longer term study based on the

restoration blocks in this study could explicitly test the relationship between restoration rates for all species and precipitation levels.

Broad Scale Landscape Theories and Their Relevance to Ecological Restoration

Several landscape scale theories are relevant to ecological restoration, even at the site specific level. These theories are examined briefly below.

Landscape Ecology

A landscape can be defined as "[a] mosaic of two or more ecosystems that exchange organisms, energy, water and nutrients" (Parks Canada 2008, p 81). A more flexible definition is "an area that is spatially heterogeneous in at least one factor of interest" (Turner 2005, p 320). Spatial pattern and ecological processes operate over a variety of scales, from the very fine scale (e.g. spatial heterogeneity in soils, relevant to microbes and individual plants) to much larger scales (e.g. an entire watershed) (Turner 2005). The pattern or process in question determines the scale of the landscape.

The field of landscape ecology has grown immensely in the past few decades, and it is now widely used in many branches of ecology (Turner 2005). Most simply, landscape ecology is "the study of spatial variation in landscapes at a variety of scales" (IALE 2011). A common element in most definitions of landscape ecology is a focus on spatial heterogeneity and ecological processes, and on understanding the interactions between these elements. Landscape ecology has influenced how we understand natural and human dominated landscapes, including the causes and consequences of spatial heterogeneity (Turner 2005).

Landscape pattern has been demonstrated as important for a number of taxa, and elements of disturbance remain prominent in landscape studies. The ecosystems we see today are influenced by natural and anthropogenic disturbances and past and present land-uses, often leading to a complex mosaic of natural, semi-natural, and human dominated patches (Turner 2005). Theories associated with landscape ecology, including how landscape organization influences processes such as nutrient flow and genetic transfer (and how these impact species' populations), how organisms use different patches or types of habitat to meet different needs, and how disturbance influences landscape function and species dispersal are all important, even when focusing at the site specific level. In turn, site specific studies, such as ecological restorations at the community level, can provide opportunities for experimentation, and inform larger scale landscape understanding (Bell et al. 1997).

Theories such as island biogeography theory (MacArthur and Wilson 1967, cited in Daigle and Havinga 1996) help conceptualize how area and distance play a role in species distributions and

populations. The area effect suggests that smaller islands support less species richness and smaller populations of species. The distance effect suggests that species richness decreases with increasing distance from other land. Therefore, the smaller and more distant an island is, the less likely it is to be colonized, and the less likely that existing small populations will be self-sustaining.

The theory of island biogeography was developed based on oceanic islands, but its application to fragmented terrestrial habitats was soon noted by ecologists (Daigle and Havinga 1996). Ecosystem fragmentation occurs when human or non-human determinants disconnect habitats, landscapes, and ecosystems (SER 2008). Fragmentation is often exacerbated by further decreases in the size and ecological integrity of remnant areas (Daigle and Havinga 1996). The fragmentation of contiguous areas into smaller areas leads to shrinkage, attrition and isolation (SER 2008). The altered landscape surrounding remnant natural areas is often a mix of urban, agricultural, industrial and other converted land uses, which act to further isolate remaining natural areas, in effect creating islands of natural areas in the midst of developed areas (Daigle and Havinga 1996). Ecological restoration provides us with an important tool to strengthen and connect fragments in the landscape.

Criticisms of applying island biogeography theory to terrestrial fragments remind us of the intricacy in dealing with complex systems. Haila (2002) presents a number of criticisms, the foremost on using the concept of habitat fragmentation as a unitary phenomenon to describe human impacts across the landscape. His argument is based on empirical studies demonstrating that fragmentation effects vary widely, depending on the organism, habitat type, and geographic region. Fragmentation effects are thus often context specific, based on the natural history of a region or continent (Haila 2002). Another issue is that habitat fragmentation is often regarded as the same issue habitat loss. Although habitat loss can lead to species loss and impoverishment, habitat fragmentation and new spatial configurations (including an increase of edge) may actually result in an influx of species as conditions change (Haila 2002). The question of whether the new species have beneficial or harmful effects on the ecosystem is again context specific. Ecological restoration and adaptive ecosystem management provide tools to implement plans based on our best understanding, and to adapt those plans as context specific factors move restoration progress towards or away from the goal trajectory.

Haila's (2002) criticisms of broadly held assumptions remind us to be critical. Haila criticizes the widely held assumption that human-influenced environments are in "strict contract" (2002, p 325) from the natural environment, which is viewed as an "idealized, spatially homogenous" background (2002, p 326). This idea unavoidably presents human induced fragmentation in a way that negatively impacts species and processes. This assumption is important; most natural environments are heterogeneous, and

are fragmented to some degree before the introduction of any human impacts. The degree of long-term isolation which shapes the evolutionary differences found in oceanic islands cannot be compared to the isolation of remnant habitats, which evolve in conjunction with their surroundings (Haila 2002). Since terrestrial islands are never fully isolated from the contiguous landscape, the surrounding landscape matrix will always exert some influence. The reminder that the surrounding landscape has always been influenced by its surroundings, and has always been heterogeneous to some extent, no matter the current context, are important considerations for ecological restorations.

Haila (2002) does not argue that habitat fragmentation is without consequences. He instead challenges the notion that fragmentation always has the same effects, gives weight to context specific effects, and suggests that we view habitat fragmentation as a specific type of human-induced change (Haila 2002). Fragmentation from human induced activities generally occurs more quickly and over a larger spatial scale than fragmentation from non-human sources (Byers 2002; Booth et al. 2010), and fragmentation from natural sources tends to have a higher degree of connectivity with the surrounding naturally heterogeneous landscape (Haila 1999, cited in Honnay et al. 2005). Island biogeography theory as applied to terrestrial habitat fragments is an imperfect analogy, as Haila (2002) argues, but still a useful one. The application of island biogeography theory to terrestrial areas has undoubtedly helped to inform decisions on how best to protect and restore these areas.

Fragmented natural areas that are in contrast to the surrounding landscape are referred to as patches if they are non linear, and corridors if they are linear (Daigle and Havinga 1996). Corridors can also be referred to as the connections between patches (Theberge and Theberge 2009). Vegetated waterways and ravines are often intact corridors, some patches are linked by corridors while other patches are completely isolated from one another in the landscape (Daigle and Havinga 1996). Ecological processes (e.g. nutrient flows, exchange of genetic material) are interrupted or discontinuous as a result of fragmentation, and ecosystem health (and human well-being) is decreased as a result of the interruption to the flow of ecological goods and services (SER 2008). Ecosystem fragmentation, combined with habitat loss, is creating “profound changes” to the natural world at all spatial scales (SER 2008, p 1), and has been referred to as “the most serious threat to biological diversity”, and as “the primary cause of the present extinction crisis” (Wilcox and Murphy 1985, p 884).

Along with interrupting ecological processes, fragmented landscapes often create subdivided populations called metapopulations. Metapopulation theory is based on the idea that some populations in suitable habitat patches are spatially discrete due to various barriers, including an inability to disperse across geographic boundaries or an inhospitable landscape matrix between patches. Metapopulations in

different patches have various probabilities of extinction, and small isolated populations will eventually go extinct (McCullough 1996). Connected populations, even those that only periodically exchange members or genetic material, are more likely to persist (Theberge and Theberge 2009). Overall, each metapopulations' survival depends on immigration rates which are higher than the mean extinction rate (Levins 1970, cited in Wilcox and Murphy 1985). When metapopulations cannot disperse and connect with other populations on their own, ecological restoration provides us with tools to reintroduce and reconnect populations.

These landscape level issues are some of the reasons many authors (e.g. Hobbs and Harris 2001; Holl et al. 2003; Suding et al. 2004) put an emphasis on ecological restoration as one of most important and powerful tools in the coming century. Protected areas, such as the urban protected area used in this ecological restoration, provide us with a core network to connect across the landscape, and can even provide information to inform restoration ecology when they are suitable sites for ecological restoration.

Protected Area Theory and Design

A protected area can be defined as "[a]n area of land and/or sea especially dedicated to the protection of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means" (IUCN 2011). Different type of protected areas allow for different uses, and often have specific management objectives which are related to their allowed uses (Dearden and Rollins 2009). In order to adequately conserve the majority of biological diversity, protected area planning must take place at the landscape, regional, and ecosystem level (Franklin 1993). It is well accepted that as the area surrounding a protected area becomes less natural, the maintenance of its integrity becomes more difficult (Noss 1995). Managing for ecological integrity includes developing a comprehensive land conservation process, and adapting protected area design as needed through time (Noss 1995). When protected areas are degraded, ecological restoration can help restore ecological integrity.

Protected area planning to ensure biological diversity and ecological integrity in an era of fragmentation must consider the size and shape of protected areas, the contextual landscape in which they are located, and the level of connectivity between areas. In order to effectively link protected areas with other natural areas, the capacity of each area to function both independently, and as part of a larger system, must be considered. A network of habitat reserves are essential elements to the preservation of biological diversity and ecological integrity, and attention to the number, size, geographic distribution and intervening landscape must all figure into the design of a reserve system (Franklin 1993). Incorporating theories of fragmentation and metapopulations into biological thinking necessitated that protected area

planning recognize the importance of scale. The 'hands-off' preservation stage that guided park management for decades was no longer adequate. Most protected areas are too small and isolated to maintain viable populations, they have been historically modified from their natural state, and they may also lack key ecological processes such as fire or predation. Conflicting land-uses, both inside and outside of protected areas, impact both the species and ecosystems inside protected areas (Woodley 2009).

The single large or several small (SLOSS) theory is based on the concepts of island biogeography, and is concerned with whether a single large reserve is likely to support more species than several small reserves (Wilcox and Murphy 1985). The general consensus in the conservation field is that habitat fragmentation has a negative effect on population survival and biological diversity, and that a reduction in habitat fragmentation should be a primary consideration in conservation strategies (Wilcox and Murphy 1985). In fragmented landscapes, larger, more intact and well-connected reserves are preferred to multiple smaller reserves. Three principles that emerged regarding the optimum pattern for a system of reserves are "(1) blocks of habitat close together are better than blocks far apart; (2) habitat in contiguous blocks is better than fragmented habitat; and (3) interconnected blocks of habitat are better than isolated blocks" (Noss et al. 1997, cited in Theberge and Theberge 2009, p 90).

Two ways to combat fragmentation and plan across ecoregional boundaries are to increase patch area, and to increase connectivity between patches (Noss 1987). Useful tools include buffer zones, corridors, stepping stones islands of biodiversity and bioregional planning on a landscape level (SER 2008). Dispersal mechanisms, such as corridors, provide potential to link patches and metapopulations across the landscape. Corridors are not a complete solution to the problem of reserves that are too small, nor are they without their own problems (e.g. the issues of corridors as ecological traps, or as dispersal conduits for disease transmission or invasive species)[6]. However, corridors do provide linkages that come as close to approximating the original interconnected landscape as we are likely to achieve (Noss 1987). The best potential usage for corridors is to plan their type and extent based on the species expected to use them (Theberge and Theberge 2009). Ecological restoration provides us with the tools to restore and connect protected areas across the landscape.

Conclusion

Ecological restoration of forest communities often focuses on trees and woody vegetation, but the native herbaceous understory represents an important component of the forest ecosystem. For some degraded sites, the recovery of the native herbaceous vegetation is a key first step to altering local

[6] See Noss 1987 for a review of the potential advantages and disadvantages of corridors.

conditions, making the site suitable for regenerating woody vegetation (Lefler 2006), and restoring ecological integrity. Plant reintroductions through ecological restoration are important components in active management of protected areas, and more research on how to maintain and restore herbaceous forest understory species in areas affected by human activities are needed (Whigham 2004). Active management through the reintroduction of native understory plant species is often necessary when forested areas have been disturbed, and especially if the understory is dominated by species that readily invade the site, or if the landscape is highly fragmented (Whigham 2004; Mottl et al. 2006). These challenges were all existing in Natchez Hills ESPA, requiring an active management program using ecological restoration.

Restoring native perennial herbaceous species in degraded forests presents significant challenges (Mottl et al. 2006), but this early ecological restoration followed a methodology designed for real-world constraints. Techniques which are practical, cost-effective, and implementable are important considerations to further the practice of ecological restoration. Theories of how restoration ecology can improve as a science also require considerations of species, community, and landscape ecology. Instead of the typical single species approach (Drayton and Primack 2000), three forest understory species, representing different growth forms, families, and basic biology were transplanted in different densities and multiple replications, to assess early restoration progress at a degraded forested site. The methodology used in this study was found to be successful, and can be recommended for future early ecological restorations.

In this early ecological restoration experiment the density treatments selected were not found to have a statistically significant impact on restoration success, however, the restoration success rate was species dependent. Two of the three species had excellent early ecological restoration success rates close to 100%; all three species had over 50% restoration success. All three of the native forest understory species used can be recommended for future understory ecological restorations in the City of Kitchener in maple beech upland forests, on the basis of the early ecological restoration success observed in this study. Previous studies on the restoration of herbaceous forest understory species using transplants in Natchez Hills ESPA (Ferguson 2007) as well as other temperate forests (Drayton and Primack 2000; Murphy 2005; Mottl et al. 2006) have experienced similar short term success (e.g. Ferguson 2007), strengthening the use of mature transplants in forest understory restorations.

This study focused explicitly on the restoration of the native herbaceous layer of the forest understory, and in doing so enhanced the ecological integrity of the study site by restoring a missing component of the forest ecosystem. The successful establishment of the initial planting represents

restoration success for this early ecological restoration experiment. Long-term monitoring to assess further progress is recommended, and strategies to do so are outlined. Future signs of continued restoration success will include the long-term persistence of the restored populations and dispersal of the initial populations, as well as colonization of the site by other native species. The results obtained through this ecological restoration add to the information collected as part of a longer term ongoing study in the Regional Municipality of Waterloo by Murphy (2005), Lefler (2006), and Ferguson (2007), on practical and theoretical aspects of urban restoration. The information in this study also contributes to ongoing efforts in this area by authors working at the landscape-scale, including Weaver (2007) and Vincent (2009).

References

Allaby, M. 2004. Recreatability [Internet]. A Dictionary of Ecology. [cited 2011 March 15]. Available from http://www.encyclopedia.com/doc/1O14-recreatability.html

Anderson, WB. 2003. Interactions of nutrient effects with other biotic factors in the herbaceous layer. In Gilliam FS, Roberts MR, editors. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. p 91-101.

Antonovics J, Primack RB. 1982. Experimental ecological genetics in Plantago: VI. The demography of seedling transplants of P. Lanceolata. Journal of Ecology. 70(1):55-75.

Bell SS, Fonseca MS, Motten LB. 1997. Linking restoration and landscape ecology. Restoration Ecology. 5(4):318-323.

Bierzychudek P. 1982. Life histories and demography of shade-tolerant temperate forest herbs: A Review. New Phytologist. 90(4):757-776.

Booth BD, Murphy SD, Swanton CJ. 2010. Invasive plant ecology in natural and agricultural systems. 2nd ed. Cambridge (UK): Oxford University/CABI Press. 329 p.

Bradley KL, Damschen EI, Young LM, Kuefler D, Went S, Wray G, Haddad NM, Knops JMH, Louda SM. 2003. Spatial heterogeneity, not visitation bias, dominates variation in herbivory. Ecology. 84(8): 2214-2221.

Bradshaw AD. 1996. Underlying principles of restoration. Canadian Journal of Fisheries and Aquatic Science. 53(Suppl. 1):3-9.

Brett JF. 1981. The morphology and taxonomy of Caulophyllum Thalictroides (L.) Michx (Berberidaceae) in North America [master's thesis]. [Guelph (ON)]: University of Guelph.

Brouillet L, Coursol F, Favreau M, Anions M, Bélisle P, Desmet P. 2010+a. Podophyllum peltatum Linnaeus – Vascan. [Internet]. [cited 2011 Mar 19]. Available from: http://data.canadensys.net/vascan/taxon/3672

Brouillet L, Coursol F, Favreau M, Anions M, Bélisle P, Desmet P. 2010+b. Caulophyllum giganteum (Farwell) Loconte & W.H. Blackwell – Vascan. [Internet]. [cited 2011 Mar 23]. Available from: http://data.canadensys.net/vascan/taxon/3669

Brouillet L, Coursol F, Favreau M, Anions M, Bélisle P, Desmet P. 2010+c. Maianthemum racemosum subsp. racemosum (Linnaeus) Link – Vascan. [Internet[. [cited 2011 Mar 23]. Available from: http://data.canadensys.net/vascan/taxon/2739

Byers JE. 2002. Impact of non-indigenous species on natives enhanced by anthropogenic alteration of selection regimes. Oikos. 97(3):449-458.

Cahill JF, Castelli JP, Casper BB. 2001. The herbivory uncertainty principle: visiting plants can alter herbivory. Ecology. 82(2):307-312.

Cahill JF, Castelli JP, Casper BB. 2002. Separate effects of human visitation and touch on plant growth and herbivory in an old-field community. American Journal of Botany. 89(9):1401-1409.

Canada National Parks Act. 2000. C. 32. [Internet]. Department of Justice; [Current to 2011 Mar 24; Amended 2010 Dec 10; Cited 2011 Apr 14]. Available from: http://laws-lois.justice.gc.ca/eng/acts/N-14.01/index.html

Choi YD. 2004. Theories for ecological restoration in changing environment: Toward 'futuristic' restoration. Ecological Research. 19:75-81.

City of Kitchener. 2000. Grand River South Community Plan [Internet]. Department of Development and Technical Services. Kitchener (ON): City of Kitchener; [cited 2009 Nov 25] 23p. Available from: http://app.kitchener.ca/communityplans/es_cpMain.aspx

City of Kitchener. 2006. Urban Natural Area Management Plan: City of Kitchener. Forestry Section, Parks Operations. Draft Report. Prepared by: Leah Lefler.

City of Kitchener. 2008. The City of Kitchener Municipal Data [computer file]. Department of Information Services. Kitchener (ON): City of Kitchener. Available from: University of Waterloo Map Library, Waterloo, ON.

Clapham AR. 1936. Over-dispersion in grassland communities and the use of statistical methods of plant ecology. Journal of Ecology. 24:232-251. (cited in Whitford 1949)

Clewell A, Rieger JP. 1997. What practitioners need from restoration ecologists. Restoration Ecology. 5(4):350-354.

Constable JVH, Peffer BJ, DeNicola DM. 2007. Temporal and light-based changes in carbon uptake and storage in the spring ephemeral *Podophyllum peltatum* (Berberidaceae). Environmental and Experimental Botany. 60(1):112-120.

Daigle J, Havinga D. 1996. Restoring nature's place: a guide to naturalizing Ontario parks and Greenspace. Schomberg (ON): Ecological Outlook Consulting and Ontario Parks Association. 226p.

de Kroon H, Whigham DF, Watson MA. 1991. Developmental ecology of mayapple: effects of rhizome severing, fertilization and timing of shoot senescence. Functional Ecology. 5:360-368.

Dearden P, Rollins R. 2009. Parks and Protected Areas in Canada Planning and Management. 3rd ed. Toronto (ON): Oxford University press. Chapter 1, Parks and Protected Areas in Canada; p 3-23.

DeMars BG, Boerner RE. 1997. Foliar phosphorus and nitrogen resorption in three woodland herbs of contrasting phenology. Castanea. 62(1):43-54.

Drayton B, Primack RB. 2000. Rates of success in the reintroduction by four methods of several perennial plant species in eastern Massachusetts. Rhodora. 102(911):299-331.

Dutilleul P. 1993. Spatial heterogeneity and the design of ecological field experiments. Ecology. 74(6): 1646-1658.

Ecological Framework of Canada. [Date unknown]. Ecoregion: MANITOULIN-LAKE SIMCOE. [Internet]. [cited 2011 Apr 6]. Available from: http://www.ecozones.ca/english/region/134.html

Ehrenfeld JG. 2000. Defining the limits of restoration: The need for realistic goals. Restoration Ecology. 8(1):2-9.

Ehrlén J, Eriksson O. 2000. Dispersal limitation and patch occupancy in forest herbs. Ecology. 81(6):1667-1674.

Elliott KA. 1998. The forests of southern Ontario. The Forestry Chronicle. 74(6):850-854.

Environment Canada. 2004a. Great Lakes Fact Sheet: How Much Habitat is Enough [Internet]. 2nd ed. Ottawa (ON): Government of Canada; [cited 2009 Jun 4]. 12 p. Available from: http://www.on.ec.gc.ca/wildlife/publications-e.html

Environment Canada. 2004b. How Much Habitat is Enough: A Framework for Guiding Habitat Rehabilitation in Great Lakes Areas of Concern [Internet]. 2nd ed. Ottawa (ON): Government of Canada; [cited 2009 Nov 26]. 88 p. Available from: http://www.on.ec.gc.ca/wildlife/publications-e.html

Environment Canada. 2006. Canadian Protected Areas Status Report, 2000-2005. Ottawa (ON): Government of Canada. 81 p. Available from: http://www.ec.gc.ca/Publications/default.asp?lang=En&xml=5A5F8028-B497-4441-92DE-4015F1E4F5D1

Environment Canada. 2007a. Ontario Weather Review - September 2007. [Internet]. Ottawa (ON): Government of Canada; [updated 2007 Oct 01; cited 2011 Mar 23]. Available from: http://www.on.ec.gc.ca/announce.cfm?ID=778&Lang=e

Environment Canada. 2007b. Ontario Weather Review - October 2007. [Internet]. Ottawa (ON): Government of Canada; [updated 2007 Nov 01; cited 2011 Mar 29]. Available from: http://www.on.ec.gc.ca/announce.cfm?ID=779&Lang=e

Environment Canada. 2010a. Environment Canada – Environmental Indicators – Protected Areas Data [Internet]. Ottawa (ON): Government of Canada; [updated 2010 Nov 09; cited 2011 Apr 06]. Available from: http://www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en&n=2F9B17BE-1

Environment Canada. 2010b. Daily Observation Data, Canada's National Climate Archive. [Internet]. Ottawa (ON): Government of Canada; [updated 2010 Sept 21; cited 2009 Mar 2]. Available from: http://climate.weatheroffice.gc.ca/climateData/dailydata_e.html?timeframe=2&Prov=CA&StationID=32008&Year=2006&Month=4&Day=14
Notes: URL given identifies daily climate data for Region of Waterloo Int'l Airport Ontario weather station in April 2006, within page search required to access info for April to September of 2006 to 2008.

Environment Canada. 2010c. Canadian Climate Normals 1971-2000. [Internet]. Ottawa (ON): Government of Canada; [updated 2010 Sept 28; cited 2009 Mar 2]. Available from: http://climate.weatheroffice.gc.ca/climate_normals/results_e.html?Province=ALL&StationName=waterloo&SearchType=BeginsWith&LocateBy=Province&Proximity=25&ProximityFrom=City&StationNumber=&IDType=MSC&CityName=&ParkName=&LatitudeDegrees=&LatitudeMinutes=&LongitudeDegrees=&LongitudeMinutes=&NormalsClass=A&SelNormals=&StnId=4832&
Notes: URL given identifies Canadian Climate Normals for Waterloo Wellington A* weather station.

Fahselt D. 2007. Is transplanting an effective means of preserving vegetation? Canadian Journal of Botany-Revue Canadienne de Botanique. 85(10):1007-1017.

Ferguson KA. 2007. Upland hardwood forest restoration: Testing combinations of vegetation in the progress of early restoration [master's thesis]. [Waterloo (ON)]: University of Waterloo.

First Base Solutions. 2006. SWOOP Southwestern Ontario Orthoimagery Project: Orthoimagery 2006 [computer file]. FIRST**BASE**SOLUTIONS. Available from: University of Waterloo Map Library, Waterloo, ON.

Fisher RA. 1971. The Design of Experiments. 8th ed. London (UK): Oliver and Boyd. 248p. (cited in Potvin 2001)

[FNA Ed. Comm.] Flora of North America Editorial Committee. 1997. Flora of North America north of Mexico. Volume 3: Magnoliophyta: Magnoliidae and Hamamelidae. New York (NY): Oxford University Press. (cited in Brouillet et al. 2010+a, Brouillet et al. 2010+b)

[FNA Ed. Comm.] Flora of North America Editorial Committee. 2002. Flora of North America north of Mexico. Volume 26: Agavaceae, Aloceae, Burmanniaceae, Dioscoreaceae, Haemodoraceae, Iridaceae, Liliaceae, Orchidaceae, Pontederiaceae, Smilaccaceae and Stemonaceae. New York (NY): Oxford University Press. (cited in Brouillet et al. 2010+c)

Foley JA, DeFries R, Asner GP, Barford C, Bonan G, Carpenter SR, Chapin FS, Coe MT, Daily GC, Gibbs HK, et al. 2005. Global Consequences of Land Use. Science. 309(5734):570-574.

Frankland F, Nelson T. 2003. Impacts of white-tailed deer on spring wildflowers in Illinois, USA. Natural Areas Journal. 23(4):341-348.

Franklin JF. 1993. Preserving biodiversity: Species, ecosystems, or landscapes? Ecological Applications. 3(2):202-205.

Franklin JF. 1995. Structural and functional diversity in temperate forests. In: Wilson EO, editor. Biodiversity. 12th ed. Washington (DC): National Academies Press. p 166-175.

Friesen L. 1998. Impacts of urbanization on plant and bird communities in forest ecosystems. The Forestry Chronicle. 74(6):855-860.

Geber MA, de Kroon H, Watson MA. 1997. Organ preformation in mayapple as a mechanism for historical effects on demography. Journal of Ecology. 85(2):211-223.

Gilliam FS, Roberts MR. 2003a. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. Chapter 1, Introduction, Conceptual framework for studies of the herbaceous layer. p 3-11.

Gilliam FS, Roberts MR. 2003b. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. Preface. p vi-x.

Gilliam FS, Roberts MR. 2003c. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. Chapter 8, Interactions between the herbaceous layer and overstory canopy of eastern forests A mechanism for linkage. p 198-223.

Givnish TJ. 1986. On the economy of plant form and function: proceedings of the Sixth Maria Moors Cabot Symposium, Evolutionary Constraints on Primary Productivity, Adaptive Patterns of Energy Capture in Plants, Harvard Forest, August 1983. New York (NY): Cambridge University Press. Biomechanical constraints on crown geometry in forest herbs. p 525-583.

Givnish TJ. 1987. Comparative studies of leaf form: Assessing the relative roles of selective pressure and phylogenetic constraints. New Phytologist. 106(1):131-160.

[GRCA] Grand River Conservation Authority. 2003. Grand River Watershed Data [computer file]. Cambridge, Ontario: Grand River Conservation Authority. Available from: University of Waterloo Map Library, Waterloo, ON.

[GRCA] Grand River Conservation Authority. 2004. A Watershed Forest Plan for the Grand River [Internet]. [cited 2009 Nov 16]. Available from: http://grandriver.ca/forestry/ForestPlan_complete.pdf

Gray P. 2006. Adaptive management in the 21st century. In Lemieux CJ, Zorn PA, Bellhouse TJ, Nelson JG, editors. Parks Research Forum of Ontario: State of the Art Workshop Series No. 2. Monitoring in Ontario's Parks and Protected Areas. Waterloo (ON): Parks Research Forum of Ontario. p 3-7.

Haila Y. 2002. A conceptual genealogy of fragmentation research: From island biogeography to landscape ecology. Ecological Applications. 12(2): 321-334.

Haila Y. 1999. Islands and fragments. In Hunter ML, editor. Maintaining Biodiversity in Forest Ecosystems. Cambridge (UK): Cambridge University Press. p 234-264. (cited in Honnay et al. 2005)

Hannan GL, Prucher HA. 1996. Reproductive biology of Caulophyllum thalictroides (Berberidaceae), an early flowering perennial of eastern North America. The American Midland Naturalist. 136(2):267-277.

Harper JL. 1977. Population biology of plants. London (UK): Academic Press Ltd. 892 p. (cited in Brett 1981)

Heritage Resources Centre. 2006. Grand Actions: A Registry of Accomplishments and Commitments, 2006. [Internet]. Waterloo (ON): Grand River Conservation Authority. [cited 2011 Apr 15]. 2 p. Available from: http://www.grandriver.ca/GrandActionsRegistry/2006/2459.pdf

Higgs E. 2005. The two-culture problem: Ecological restoration and the integration of knowledge. Restoration Ecology. 13(1):159-164.

Hobbs RJ, Harris JA. 2001. Restoration ecology: Repairing the Earth's ecosystems in the new millennium. Restoration Ecology. 9(2):239-246.

Holl KD, Crone EE, Schultz CB. 2003. Landscape restoration: Moving from generalities to methodologies Bioscience. 53(5):491-502.

Holland PG. 1980. Trout lily in Nova Scotia: an assessment of the status of its geographic range. Journal of Biogeography. 7(4):363-381. (cited in Middleton and Merriam 1983)

Honnay O, Jacquemyn H, Bossuyt B, Hermy M. 2005. Forest fragmentation effects on patch occupancy and population viability of herbaceous plant species. New Phytologist. 166(3):723-736.

Hornik, K. 2011. The R FAQ. [Internet]. [Updated 2011 Apr 7, cited 2011 Apr 15]. Available from: http://CRAN.R-project.org/doc/FAQ/R-FAQ.html

Hull JC. 2002. Photosynthetic induction dynamics to sunflecks of four deciduous forest understory herbs with different phenologies. International Journal of Plant Sciences. 163(6):913-924.

Hurlbert SH. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs. 54(2):187-211. (cited by Dutilleul 1993)

Hüttl RF, Schneider BU, Farrell EP. 2000. Forests of the temperate region: gaps in knowledge and research needs. Forest Ecology and Management. 132(1):83-96.

[IALE] International Association of Landscape Ecology. c2011. [Internet]. [cited 2011 Mar 7]. Available from: http://www.landscape-ecology.org/

[IUCN] World Conservation Union. 2011. IUCN Definitions - English [Internet]. [Cited 2011 Mar 08]. 76 p. Available from: http://www.iucn.org/knowledge/publications_doc/publications/

Jacquemyn H, Butaye J, Hermy M. 2003. Impacts of restored patch density and distance from natural forests on colonization success. Restoration Ecology. 11(4):417-423.

Jolls CL. 2003. Populations of and threats to rare plants of the herb layer. In Gilliam FS, Roberts MR, editors. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. p 105–159.

Karr JR. 1996. Ecological integrity and ecological health are not the same. In Schulze P, Frosch R, Risser P, editors. Engineering within ecological constraints. Washington (DC): National Academy of Engineering. 213 p.

Karr JR, Dudley DR. 1981. Ecological perspective on water quality goals. Environmental Management. 5(1):55-68.

Kim YS, Lim S, Choi YE, Anbazhagan VR. 2007. High frequency plant regeneration via somatic embryogenesis in *Podophyllum peltatum* L., an important source of anticancer drug. Current Science. 92(5):662-666.

Korb JE, Fulé PZ. 2008. Intra- and interannual vegetation change: implications for long-term research. Restoration Ecology. 16(1):5-11.

Kraft LS, Crow TR, Buckley DS, Nauertz EA, Zasada JC. 2004. Effects of harvesting and deer browsing on attributes of understory plants in northern hardwood forests, Upper Michigan, USA. Forest Ecology and Management. 199(2-3):219-230.

Krochman A, Wilkins L, Van Lear D, Chien M. 1974. Mayapple, *Podophyllum peltatum* L. USDA Forest Service Research Paper NE-296. Upper Darby (PA): U.S. Department of Agriculture. 9 p.

LaFrankie Jr. JV. 1986. Transfer of the species of smilacina to maianthemum (liliaceae). Taxon. 35(3):584-9.

Lake PS. 2001. On the maturing of restoration: Linking ecological research and restoration. Ecological Management & Restoration. 2(2):110-115.

Lapointe L, Molard J. 1997. Costs and benefits of mycorrhizal infection in a spring ephemeral, erythronium americanum. New Phytol. 135(3):491-500.

Larson BM, Riley JL, Snell EA, Godschalk HG. 1999. The woodland heritage of southern Ontario: A study of ecological change, distribution and significance. Don Mills (ON): Federation of Ontario Naturalists.

Laverty TM. 1992. Plant interactions for pollinator visits - a test of the magnet species effect. Oecologia. 89(4):502-8.

Laverty TM, Plowright RC. 1988. Fruit and seed set in mayapple (podophyllum, peltatum) - influence of intraspecific factors and local enhancement near pedicularis-canadensis. Canadian Journal of Botany-Revue Canadienne De Botanique. 66(1):173-8.

Lawton JH. 2000. Community ecology in a changing world. Luhe (Germany): Ecology Institute. 227 p. (cited in Simberloff 2004)

Lee HT, Bakowsky WD, Riley J, Bowles J, Puddister M, Uhlig P, McMurray S. 1998. Ecological land classification for southern Ontario: First approximation and its application. North Bay (ON): Ontario Ministry of Natural Resources. Southcentral Science Section, Science Development and Transfer Branch.

Lefler LE. 2006. An assessment of early-stage forest restoration outcomes and the instruments used to evaluate ecosystem recovery [master's thesis]. [Waterloo (ON)]: University of Waterloo.

Levins R. 1970. Extinction. In: Gerstenhaber M, editor. Some mathematical questions in biology. Vol. II. Providence (RI): American Mathematical Society. p 77-107. (cited in Wilcox and Murphy 1985)

Likens GE, Bormann FH, Pierce RS, Eaton JS, Johnson NM. 1977. Biogeochemistry of a forested ecosystem. New York (NY): Springer-Verlag. (cited in Muller 2003)

MacArthur RH, Wilson EO. 1967. The theory of island biogeography. Princeton (NJ): Princeton University Press. 203 p. (cited in Daigle and Havinga 1996)

Manning AD, Lindenmayer DB, Fischer J. 2006. Stretch goals and backcasting: Approaches for overcoming barriers to large-scale ecological restoration. Restor Ecol. 14(4):487-92.

Maunder M. 1992. Plant reintroduction - an overview. Biodivers Conserv. 1(1):51-61.

McCullough DR. 1996. Metapopulations and wildlife conservation. Washington (DC): Island Press. Introduction; p 1-10.

McCune B, Grace JB. 2002. Analysis of ecological communities. Oregon: MjM Software Design. 300 p.

McLachlan SM, Bazely DR. 2003. Outcomes of longterm deciduous forest restoration in southwestern Ontario, Canada. Biol Conserv. 113(2):159-69.

Menges ES. 2008. Restoration demography and genetics of plants: When is a translocation successful? Aust J Bot. 56(3):187-96.

Michener WK. 1997. Quantitatively evaluating restoration experiments: Research design, statistical analysis, and data management considerations. Restor Ecol. 5(4):324-37.

Middleton J, Merriam G. 1983. Distribution of woodland species in farmland woods. J Appl Ecol. 20(2):625-44.

Mottl LM, Mabry CM, Farrar DR. 2006. Seven-year survival of perennial herbaceous transplants in temperate woodland restoration. Restor Ecol. 14(3):330-8.

Muller RN. 2003. Nutrient relations of the herbaceous layer in deciduous forest ecosystems. In Gilliam FS, Roberts MR, editors. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. p 15-37.

Murphy SD. 2005. Concurrent management of an exotic species and initial restoration efforts in forests. Restor Ecol. 13(4):584-93.

Murphy SD. 2006. Ecological restoration in parks and protected areas: Using protected areas for testing methods of restoring forest communities. In Society for Ecological Restoration of Ontario. 2006. 2007-2008 Native Plant Resource Guide for Ontario. Waterloo (ON): Society for Ecological Restoration of Ontario. p 11-14.

Murphy SD, Flanagan J, Noll K, Wilson D, Duncan B. 2007. How incomplete exotic species management can make matters worse: Experiments in forest restoration in Ontario, Canada. Ecological Restoration. 25(2):85-93.

[NHIC] National Heritage Information Centre. 2011. Natural Areas Report: Natchez Hills Upland Woods. [Internet]. Element Summary Report for Ontario Ministry of Natural Resources. Peterborough (ON): Ontario Ministry of Natural Resources; [Updated 2010 July 30; Cited 2011 Apr 19]. Available from: https://www.biodiversityexplorer.mnr.gov.on.ca/nhicWEB/eoNaturalAreasDetailReport.do?naSearchResultsId=4422

Natural Resources Canada. 2008. The State of Canada's Forests. Annual Report 2008. [Internet]. Ottawa (ON): Canadian Forest Service; [Modified 2011 Apr 19; cited 2009 July 9]. 48 p. Available from: http://cfs.nrcan.gc.ca/publications/?id=28889

Neufeld HS, Young DR. 2003. Ecophysiology of the herbaceous layer in temperate deciduous forests. In Gilliam FS, Roberts MR, editors. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. p 38-90.

Newcomb L. 1977. Newcomb's Wildflower Guide. New York (NY): Little, Brown and Company. 490 p.

Newton A, Featherstone AW. 2005. Restoring temperate forests. In Mansourian S, Vallauri D, Dudley N, editors. Forest restoration in landscapes. Beyond planting trees. New York (NY): Springer. p 320-330.

Noss RF. 1987. Corridors in real landscapes: A Reply to Simberloff and Cox. Conservation Biology. 1(2):159-164.

Noss RF. 1995. Maintaining ecological integrity in representative reserve networks. Toronto (ON): World Wildlife Fund. 77 p.

Noss RF, O'Connell MA, Murphy DD. 1997. The science of conservation planning: Habitat conservation under the Endangered Species Act. Washington (DC): Island Press. 246 p.

Ontario Ministry of Natural Resources. 2008. Natural Resources and Values Information System (NRVIS) [computer file]. Toronto, Ontario: The Ontario Ministry of Natural Resources. Available from: University of Waterloo Map Library, Waterloo, ON.

Paleczny D. 2006. Ontario Parks: Policy and planning context. In Lemieux CJ, Zorn PA, Bellhouse TJ, Nelson JG, editors. Parks Research Forum of Ontario: State of the Art Workshop Series No. 2. Monitoring in Ontario's Parks and Protected Areas. Waterloo (ON): Parks Research Forum of Ontario. p 25-28.

Palmer MA, Ambrose RF, Poff NL. 1997. Ecological theory and community restoration ecology. Restor Ecol. 5(4):291-300.

Parks Canada. 2008. Principles and guidelines for ecological restoration in Canada's protected natural areas. National Parks Branch, Canadian Parks Council. Gatineau (QC): National Parks Directorate, Parks Canada Agency. 99 p.

Piper JK. 1989. Light, flowering, and fruiting within patches of smilacina-racemosa and smilacina-stellata (liliaceae). Bulletin of the Torrey Botanical Club. 116(3):247-57.

Pitelka LF. 1984. Application of the - 3/2 power law to clonal herbs. Am Nat. 123(4):442-9. (cited in Whigham 2004)

Plasilová J. 1970. A study of the root systems and root ecology of perennial herbs in the undergrowth of deciduous forests. Preslia. 42:136-152. (cited in Neufeld & Young 2003)

Policansky D. 1983. Patches, clones and self-fertility of mayapples (podophyllum-peltatum L). Rhodora. 85(842):253-6.

Potvin C. 2001. ANOVA Experimental Layout and Analysis. In Scheiner SM and Gurevitch J, editors. Design and Analysis of Ecological Experiments. 2nd ed. New York (NY): Oxford University Press. p 63-76.

Quon SP, Martin LRG, Murphy SD. 2001. Effective planning and implementation of ecological rehabilitation projects: A case study of the Regional Municipality of Waterloo (Ontario, Canada). Environ Manage. 27(3):421-33.

Regional Municipality of Waterloo. 2006. Tri-Cities and Surrounding Communities Orthomosaics 2006 [computer file]. Waterloo (ON): The Regional Municipality of Waterloo. Available from: University of Waterloo Map Library, Waterloo, ON.

Regional Municipality of Waterloo. 2009. Regional Municipality of Waterloo Streets and Planning Data [computer file]. Kitchener (ON): Regional Municipality of Waterloo. Available from: University of Waterloo Map Library, Waterloo, ON.

Regional Municipality of Waterloo. 2011. Living in the Region of Waterloo, Rural and Urban Land. [Internet]. [Cited 2011 Apr 19] Available from: http://www.region.waterloo.on.ca/web/region.nsf/8ef02c0fded0c82a85256e590071a3ce/5e2a3cdde465e5ae85257678004dc58c!OpenDocument

Ricketts TH, Dinerstein E, Olson DM, Loucks C. 1999. Who's where in North America? Bioscience. 49(5):369-381.

Riitters K, Wickham J, O'Neill R, Jones B, Smith E. 2000. Global-scale patterns of forest fragmentation. Conserv Ecol. 4(2):3.

Roberts MR, Gilliam FS. 2003. Response of the herbaceous layer to disturbance in eastern forests. In Gilliam FS, Roberts MR, editors. The herbaceous layer in forests of eastern North America. New York (NY): Oxford University Press. p 302-320.

Rockwood LL, Lobstein MB. 1994. The effects of experimental defoliation on reproduction in four species of herbaceous perennials from northern Virginia. Castanea. 59(1):41-50.

Rosenzweig ML. 2003. Reconciliation ecology and the future of species diversity. Oryx. 37(2):194-205.

Ross N, Eyles J, Cole D, Iannantuono A. 1997. The ecosystem health metaphor in science and policy. Canadian Geographer-Geographe Canadien. 41(2):114-27.

Roy V, de Blois S. 2006. Using functional traits to assess the role of hedgerow corridors as environmental filters for forest herbs. Biol Conserv. 130(4):592-603.

Rust RW, Roth RR. 1981. Seed production and seedling establishment in the mayapple, podophyllum-peltatum L. Am Midl Nat. 105(1):51-60.

Scheiner SM. 2001. MANOVA Multiple response variables and multispecies interactions. In Scheiner SM, Gurevitch J, editors. Design and Analysis of Ecological Experiments. 2nd ed. New York (NY): Oxford University Press. p 99-115.

Schmitt, D. 1995. Woodland management program. Kitchener (ON): City of Kitchener, Forestry Section, Parks Operations.

Schmitt D. 2004. Urban woodland conservation in southern Ontario [master's thesis]. [Waterloo (ON)]: University of Waterloo.

Schmitt D, Suffling R. 2006. Managing eastern North American woodlands in a cultural context. Landscape Urban Plann. 78(4):457-64.

Shea, J. 2011 April 6. Natchez and Lackner Woods Kitchener [electronic mail on the Internet]. Message to: Megan Ihrig. [cited 2011 April 6]. [1 screen].

Simberloff D. 2004. Community ecology: Is it time to move on? Am Nat. 163(6):787-99.

Singleton, T. 1998. Recovery of the forest herb community in post-agricultural forests in central New York: pattern and process [dissertation]. [Ithaca (NY)]: Cornell University.

Slocombe DS. 1998. Lessons from experience with ecosystem-based management. Landscape Urban Plann. 40(1-3):31-9.

Small E, Catling PM. 1999. Canadian medicinal crops. Ottawa (ON): National Research Council of Canada Research Press. 240 p.

Smith RL, Smith TM. 1998. Elements of ecology. 4th ed. San Francisco (CA): Benjamin/Cummings Science Publishing. 555 p.

Sohn JJ, Policansky D. 1977. Costs of reproduction in mayapple podophyllum-peltatum (berberidaceae). Ecology. 58(6):1366-74.

[SER] Society for Ecological Restoration International Science & Policy Working Group. 2004. The SER International Primer on Ecological Restoration. Version 2, Oct 2004. [Internet]. www.ser.org & Tucson (AZ): Society for Ecological Restoration International. [Cited 2011 Apr 19]. 15 p. Available from: http://www.ser.org/content/ecological_restoration_primer.asp

[SER] Society for Ecological Restoration. 2008. Society for Ecological Restoration International Policy Position Statement: Ecological Restoration as a Tool for Reversing Ecosystem Fragmentation. [Internet] Washington (DC): SER; [Released 2008 Oct 8; cited 2011 Apr 19]. 4 p. Available from: http://www.ser.org/content/pressroom.asp

Strobl S. 1998. Towards a list of science priorities for the conservation and management of southern Ontario forests - results of a workshop. For Chron. 74(6):838-49.

Suding KN, Gross KL, Houseman GR. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology & Evolution. 19(1):46-53.

Sydes C, Grime JP. 1981. Effects of tree leaf litter on herbaceous vegetation in deciduous woodland .1. field investigations. J Ecol. 69(1):237-48.

Theberge JC, Theberge JB. 2009. Application of ecological concepts to the management of protected areas. In Dearden P, Rollins R, editors. Parks and protected areas in Canada Planning and management 3rd ed. Toronto (ON): Oxford University Press. p 84-110.

Turner MG. 2005. Landscape ecology: What is the state of the science? Annual Review of Ecology Evolution and Systematics. 36:319-44.

[USDA] United States Department of Agriculture, Agricultural Research Service, National Genetic Resources Program. 2011. *Germplasm Resources Information Network – (GRIN) Maianthemum racemosum* information from NPGS/GRIN. [Online Database]. National Germplasm Resources Laboratory, Beltsville, Maryland; [cited 2011 Apr 7]. 1 pg. Available from: http://www.ars-grin.gov/cgi-bin/npgs/html/taxon.pl?433198

Venter O, Brodeur NN, Nemiroff L, Belland B, Dolinsek IJ, Grant JWA. 2006. Threats to endangered species in Canada. Bioscience. 56(11):903-10.

Von Ende CN. 2001. Repeated-measures analysis growth and other time-dependent measures. In Scheiner SM, Gurevitch J, editors. Design and analysis of ecological experiments. 2nd ed. New York (NY): Oxford University Press. P 134-157.

Vincent AM. 2009. Do landscape and patch spatial variables predict the presence of non-indigenous plant species? [master's thesis]. [Waterloo (ON)]: University of Waterloo.

Watmough SA, Hutchinson TC, Sager EPS. 1998. Changes in tree ring chemistry in sugar maple (Acer saccharum) along an urban-rural gradient in southern Ontario. Environmental Pollution. 101(3):381-90.

The Walter Bean Community Trails Foundation. [Date unknown]. The Walter Bean Grand River Trail – Natchez Hills [Internet]. Waterloo (ON): The Walter Bean Community Trails Foundation; [cited 2009 May 27]. Available from: http://www.walterbeantrail.ca/natchez.htm

Watson MA, Lu Y. 1999. Timing of shoot senescence and demographic expression in the clonal perennial podophyllum peltatum (berberidaceae). Oikos. 86(1):67-78.

Weaver J. 2007. Managing an urban forest: have street tree populations of Acer platanoides invaded forested parks? [master's thesis]. [Waterloo (ON)]: University of Waterloo.

Whigham DE. 2004. Ecology of woodland herbs in temperate deciduous forests. Annual Review of Ecology Evolution and Systematics. 35:583-621.

White PS, Walker JL. 1997. Approximating nature's variation: Selecting and using reference information in restoration ecology. Restor Ecol. 5(4):338-49.

Whitford PB. 1949. Distribution of woodland plants in relation to succession and clonal growth. Ecology. 30(2):199-208.

Whitney GG, Foster DR. 1988. Overstorey composition and age as determinants of the understorey flora of woods of central New-England. J Ecol. 76(3):867-76.

Wickham JD, Jones KB, Riitters KH, Wade TG, O'Neill RV. 1999. Transitions in forest fragmentation: Implications for restoration opportunities at regional scales. Landscape Ecol. 14(2):137-45.

Wilcox BA, Murphy DD. 1985. Conservation strategy - the effects of fragmentation on extinction. Am Nat. 125(6):879-87.

Woodley SJ. 1994. Ecological integrity monitoring in national parks. In Skibicki A, Stadel A, Welch D, Nelson JG, editors. Ecological monitoring and National Parks: Proceedings of a workshop held at the University of Waterloo 17-20 January 1994 Working Paper #7. Waterloo (ON): Heritage Resources Centre: University of Waterloo. p 5-12.

Woodley, S. 2009. Planning and managing for ecological integrity in Canada's National Parks. In Dearden P, Rollins R, editors. Parks and Protected Areas in Canada Planning and Management 3rd ed. Toronto (ON): Oxford University Press. p 111-132.

Zorn P. 2006. Monitoring, inventory and research needs in Parks Canada. In Lemieux CJ, Zorn PA, Bellhouse TJ, Nelson JG, editors. Parks Research Forum of Ontario: State of the Art Workshop Series No. 2. Monitoring in Ontario's Parks and Protected Areas. Waterloo (ON): Parks Research Forum of Ontario. p 67-70.

Appendix Ecological Restoration Planting Listing

Listing of species planting positions in density treatments for all blocks. Planting positions in all blocks are as shown in Figure A1. In listing Q signifies quadrat, PP signifies planting position within quadrat.

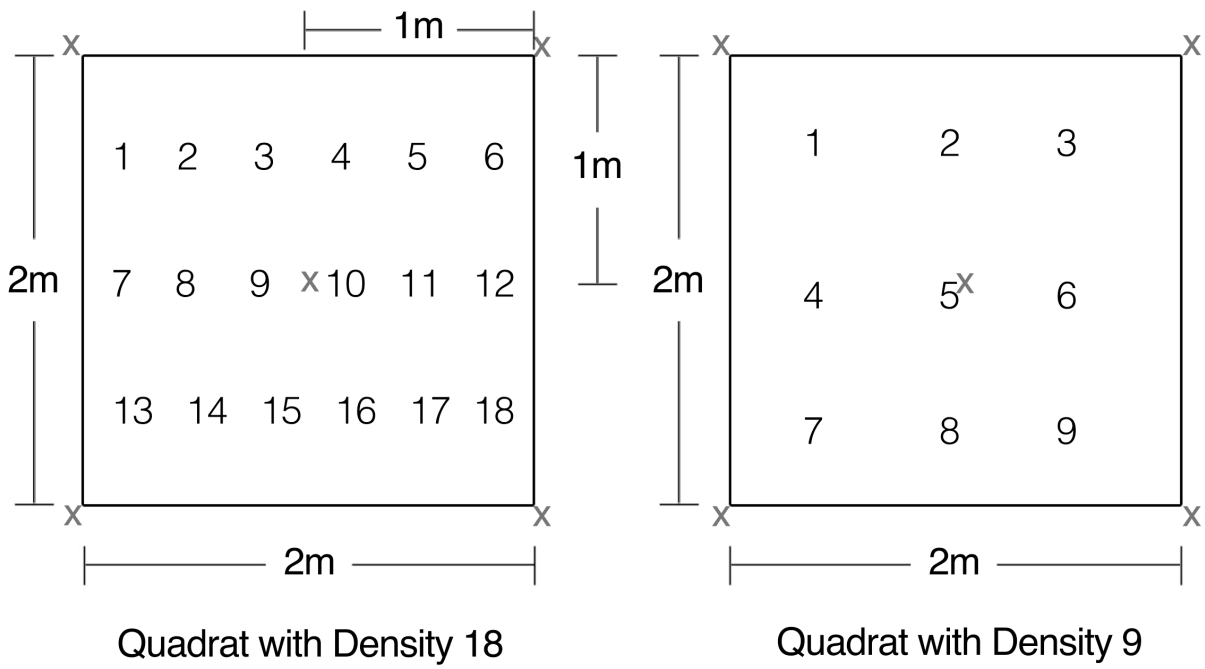

Figure A1. Quadrat layout for Density 18 and Density 9 treatments.

Block 1

Q	PP	Species
1	1	*Caulophyllum giganteum*
1	2	*Caulophyllum giganteum*
1	3	*Podophyllum peltatum*
1	4	*Podophyllum peltatum*
1	5	*Maianthemum racemosum* subsp. *racemosum*
1	6	*Caulophyllum giganteum*
1	7	*Maianthemum racemosum* subsp. *racemosum*
1	8	*Maianthemum racemosum* subsp. *racemosum*
1	9	*Podophyllum peltatum*
1	10	*Maianthemum racemosum* subsp. *racemosum*
1	11	*Maianthemum racemosum* subsp. *racemosum*
1	12	*Caulophyllum giganteum*
1	13	*Podophyllum peltatum*
1	14	*Maianthemum racemosum* subsp. *racemosum*
1	15	*Caulophyllum giganteum*
1	16	*Podophyllum peltatum*
1	17	*Caulophyllum giganteum*
1	18	*Podophyllum peltatum*
2	1	*Podophyllum peltatum*
2	2	*Maianthemum racemosum* subsp. *racemosum*
2	3	*Caulophyllum giganteum*
2	4	*Caulophyllum giganteum*
2	5	*Maianthemum racemosum* subsp. *racemosum*
2	6	*Podophyllum peltatum*
2	7	*Caulophyllum giganteum*
2	8	*Podophyllum peltatum*
2	9	*Maianthemum racemosum* subsp. *racemosum*
3	NA	Control Plot

Q	PP	Species
4	1	*Maianthemum racemosum* subsp. *racemosum*
4	2	*Podophyllum peltatum*
4	3	*Maianthemum racemosum* subsp. *racemosum*
4	4	*Caulophyllum giganteum*
4	5	*Podophyllum peltatum*
4	6	*Podophyllum peltatum*
4	7	*Caulophyllum giganteum*
4	8	*Podophyllum peltatum*
4	9	*Podophyllum peltatum*
4	10	*Podophyllum peltatum*
4	11	*Maianthemum racemosum* subsp. *racemosum*
4	12	*Caulophyllum giganteum*
4	13	*Caulophyllum giganteum*
4	14	*Maianthemum racemosum* subsp. *racemosum*
4	15	*Maianthemum racemosum* subsp. *racemosum*
4	16	*Caulophyllum giganteum*
4	17	*Maianthemum racemosum* subsp. *racemosum*
4	18	*Caulophyllum giganteum*
5	1	*Caulophyllum giganteum*
5	2	*Caulophyllum giganteum*
5	3	*Maianthemum racemosum* subsp. *racemosum*
5	4	*Maianthemum racemosum* subsp. *racemosum*
5	5	*Maianthemum racemosum* subsp. *racemosum*
5	6	*Podophyllum peltatum*
5	7	*Podophyllum peltatum*
5	8	*Caulophyllum giganteum*
5	9	*Podophyllum peltatum*
6	NA	Control Plot

Block 2

Q	PP	Species
1	1	*Caulophyllum giganteum*
1	2	*Maianthemum racemosum* subsp. *racemosum*
1	3	*Caulophyllum giganteum*
1	4	*Maianthemum racemosum* subsp. *racemosum*
1	5	*Podophyllum peltatum*
1	6	*Caulophyllum giganteum*
1	7	*Caulophyllum giganteum*
1	8	*Maianthemum racemosum* subsp. *racemosum*
1	9	*Caulophyllum giganteum*
1	10	*Podophyllum peltatum*
1	11	*Maianthemum racemosum* subsp. *racemosum*
1	12	*Podophyllum peltatum*
1	13	*Maianthemum racemosum* subsp. *racemosum*
1	14	*Maianthemum racemosum* subsp. *racemosum*
1	15	*Caulophyllum giganteum*
1	16	*Podophyllum peltatum*
1	17	*Podophyllum peltatum*
1	18	*Podophyllum peltatum*
2	1	*Maianthemum racemosum* subsp. *racemosum*
2	2	*Maianthemum racemosum* subsp. *racemosum*
2	3	*Caulophyllum giganteum*
2	4	*Podophyllum peltatum*
2	5	*Caulophyllum giganteum*
2	6	*Podophyllum peltatum*
2	7	*Podophyllum peltatum*
2	8	*Caulophyllum giganteum*
2	9	*Maianthemum racemosum* subsp. *racemosum*
3	NA	Control Plot
4	NA	Control Plot
5	1	*Caulophyllum giganteum*
5	2	*Caulophyllum giganteum*
5	3	*Maianthemum racemosum* subsp. *racemosum*
5	4	*Podophyllum peltatum*
5	5	*Caulophyllum giganteum*
5	6	*Maianthemum racemosum* subsp. *racemosum*
5	7	*Maianthemum racemosum* subsp. *racemosum*
5	8	*Caulophyllum giganteum*
5	9	*Caulophyllum giganteum*
5	10	*Maianthemum racemosum* subsp. *racemosum*
5	11	*Maianthemum racemosum* subsp. *racemosum*
5	12	*Maianthemum racemosum* subsp. *racemosum*
5	13	*Podophyllum peltatum*
5	14	*Podophyllum peltatum*
5	15	*Podophyllum peltatum*
5	16	*Podophyllum peltatum*
5	17	*Podophyllum peltatum*
5	18	*Caulophyllum giganteum*
6	1	*Caulophyllum giganteum*
6	2	*Podophyllum peltatum*
6	3	*Maianthemum racemosum* subsp. *racemosum*
6	4	*Caulophyllum giganteum*
6	5	*Maianthemum racemosum* subsp. *racemosum*
6	6	*Podophyllum peltatum*
6	7	*Podophyllum peltatum*
6	8	*Maianthemum racemosum* subsp. *racemosum*
6	9	*Caulophyllum giganteum*

Block 3

Q	PP	Species
1	1	*Maianthemum racemosum* subsp. *racemosum*
1	2	*Podophyllum peltatum*
1	3	*Maianthemum racemosum* subsp. *racemosum*
1	4	*Maianthemum racemosum* subsp. *racemosum*
1	5	*Caulophyllum giganteum*
1	6	*Podophyllum peltatum*
1	7	*Podophyllum peltatum*
1	8	*Podophyllum peltatum*
1	9	*Caulophyllum giganteum*
1	10	*Caulophyllum giganteum*
1	11	*Caulophyllum giganteum*
1	12	*Podophyllum peltatum*
1	13	*Maianthemum racemosum* subsp. *racemosum*
1	14	*Caulophyllum giganteum*
1	15	*Maianthemum racemosum* subsp. *racemosum*
1	16	*Maianthemum racemosum* subsp. *racemosum*
1	17	*Podophyllum peltatum*
1	18	*Caulophyllum giganteum*
2	1	*Podophyllum peltatum*
2	2	*Caulophyllum giganteum*
2	3	*Caulophyllum giganteum*
2	4	*Maianthemum racemosum* subsp. *racemosum*
2	5	*Maianthemum racemosum* subsp. *racemosum*
2	6	*Maianthemum racemosum* subsp. *racemosum*
2	7	*Podophyllum peltatum*
2	8	*Caulophyllum giganteum*
2	9	*Podophyllum peltatum*
3	NA	Control Plot

Q	PP	Species
4	1	*Maianthemum racemosum* subsp. *racemosum*
4	2	*Podophyllum peltatum*
4	3	*Caulophyllum giganteum*
4	4	*Maianthemum racemosum* subsp. *racemosum*
4	5	*Podophyllum peltatum*
4	6	*Podophyllum peltatum*
4	7	*Maianthemum racemosum* subsp. *racemosum*
4	8	*Caulophyllum giganteum*
4	9	*Caulophyllum giganteum*
5	1	*Caulophyllum giganteum*
5	2	*Caulophyllum giganteum*
5	3	*Podophyllum peltatum*
5	4	*Maianthemum racemosum* subsp. *racemosum*
5	5	*Maianthemum racemosum* subsp. *racemosum*
5	6	*Podophyllum peltatum*
5	7	*Podophyllum peltatum*
5	8	*Maianthemum racemosum* subsp. *racemosum*
5	9	*Caulophyllum giganteum*
5	10	*Maianthemum racemosum* subsp. *racemosum*
5	11	*Maianthemum racemosum* subsp. *racemosum*
5	12	*Podophyllum peltatum*
5	13	*Maianthemum racemosum* subsp. *racemosum*
5	14	*Caulophyllum giganteum*
5	15	*Podophyllum peltatum*
5	16	*Podophyllum peltatum*
5	17	*Caulophyllum giganteum*
5	18	*Caulophyllum giganteum*
6	NA	Control Plot

Block 4

Q	PP	Species
1	NA	Control Plot
2	1	*Maianthemum racemosum* subsp. *racemosum*
2	2	*Caulophyllum giganteum*
2	3	*Caulophyllum giganteum*
2	4	*Podophyllum peltatum*
2	5	*Caulophyllum giganteum*
2	6	*Maianthemum racemosum* subsp. *racemosum*
2	7	*Podophyllum peltatum*
2	8	*Podophyllum peltatum*
2	9	*Podophyllum peltatum*
2	10	*Maianthemum racemosum* subsp. *racemosum*
2	11	*Podophyllum peltatum*
2	12	*Maianthemum racemosum* subsp. *racemosum*
2	13	*Maianthemum racemosum* subsp. *racemosum*
2	14	*Podophyllum peltatum*
2	15	*Maianthemum racemosum* subsp. *racemosum*
2	16	*Caulophyllum giganteum*
2	17	*Caulophyllum giganteum*
2	18	*Caulophyllum giganteum*
3	1	*Caulophyllum giganteum*
3	2	*Podophyllum peltatum*
3	3	*Maianthemum racemosum* subsp. *racemosum*
3	4	*Podophyllum peltatum*
3	5	*Maianthemum racemosum* subsp. *racemosum*
3	6	*Caulophyllum giganteum*
3	7	*Caulophyllum giganteum*
3	8	*Podophyllum peltatum*
3	9	*Maianthemum racemosum* subsp. *racemosum*

Q	PP	Species
4	NA	Control Plot
5	1	*Caulophyllum giganteum*
5	2	*Podophyllum peltatum*
5	3	*Caulophyllum giganteum*
5	4	*Maianthemum racemosum* subsp. *racemosum*
5	5	*Maianthemum racemosum* subsp. *racemosum*
5	6	*Caulophyllum giganteum*
5	7	*Podophyllum peltatum*
5	8	*Maianthemum racemosum* subsp. *racemosum*
5	9	*Podophyllum peltatum*
5	10	*Podophyllum peltatum*
5	11	*Podophyllum peltatum*
5	12	*Maianthemum racemosum* subsp. *racemosum*
5	13	*Caulophyllum giganteum*
5	14	*Maianthemum racemosum* subsp. *racemosum*
5	15	*Caulophyllum giganteum*
5	16	*Podophyllum peltatum*
5	17	*Caulophyllum giganteum*
5	18	*Maianthemum racemosum* subsp. *racemosum*
6	1	*Caulophyllum giganteum*
6	2	*Maianthemum racemosum* subsp. *racemosum*
6	3	*Maianthemum racemosum* subsp. *racemosum*
6	4	*Podophyllum peltatum*
6	5	*Caulophyllum giganteum*
6	6	*Maianthemum racemosum* subsp. *racemosum*
6	7	*Caulophyllum giganteum*
6	8	*Podophyllum peltatum*
6	9	*Podophyllum peltatum*