Do landscape and patch spatial variables predict the presence of non-indigenous plant species?

by

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A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Master of Environmental Studies
in
Environment and Resources Studies

Waterloo, Ontario, Canada, 2009

Author's declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, included any required final revisions, as accepted by my examiners.

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Abstract

Studies in the fields of invasion and landscape ecology suggest that there is certainly a spatial component to the spread and establishment of exotic plant species. An understanding of how patch dynamics affects the dispersal process is necessary for effective management of invasive or problematic exotic plant species. However, studies that have specifically compared spatial attributes of natural areas with exotic plant species diversities (or abundances) have had disparities in their scale or methodology, and incongruent results. I sought to answer the question of whether landscape and patch variables (or soil texture) can predict the presence of exotic plant species in natural areas, using a case study of the cities of Kitchener and Waterloo, Ontario, Canada. The objective of this study was to determine whether landscape and patch variables can be used (albeit most effectively with long-term monitoring) as part of a management plan and monitoring priority list for Kitchener and Waterloo's urban natural areas. I collected vegetation and soil data at 98 edge and core sites in 5 natural areas, ranging in size from 16 – 146 ha. In total, 19 exotic and 39 native herbaceous species, and 11 exotic and 43 native tree and shrub species occurred in the 490 sampling plots at 98 sites. Using Usher's (1988) index of invasion, the parks cumulatively had 33% invasion of their herbaceous layer and 20% invasion of their shrub/tree layer. I conducted Kruskal-Wallis tests of the non-parametric data (for both species diversities and abundances) and results showed that native herbaceous, tree and shrub species richness was significantly higher than exotic species at core and edge sites at 4 of the 5 parks (Doon had significantly higher numbers of exotic species than native species at its edge sites). No significant relationship was found between soil texture and the presence of exotic herbaceous plants, tree or shrubs at any of the parks. Additionally, no significant relationship was found between any of the spatial attributes measured and exotic plant species diversities, although total park area, # of disjunct core areas, and perimeter: area ratio were the three strongest predictors of exotic plant species abundances. These results indicate that native plant species in Kitchener and Waterloo's urban natural areas are successfully out competing exotic plant species or, alternatively, that their core areas have become as equivalently invaded as their edges. Longer term and larger scale studies are necessary to substantiate these findings but smaller parks and parks with higher perimeter: area ratios should still be a management focus. Fragmentation of current natural areas and urbanization in close proximity to natural areas should also be minimized or avoided to decrease the risk of new invasions and establishment and exotic plant species.

Acknowledgements

The journey of this thesis, and being a Masters student, has contributed to my personal development and I am thankful for the experience.

This thesis wouldn't have been possible without financial support from OGS and the University of Waterloo.

Firstly, I would like to thank my advisor, Steve Murphy. It was an honour to be one of your graduate students. You are as much amiable as you are genius and I have learned many academic and life lessons from you. Thank you for your invaluable guidance on this thesis and for instilling in me a much needed academic confidence.

Secondly, I would like to thank, my co-advisor Pete Deadman, for your suggestions and encouragement. I would also like to thank Scott McFarlane for your invaluable help with GIS, FRAGSTATS and other aspects of spatial analysis. Thank you for allowing me to learn from your knowledge.

My colleagues Katelyn Inlow and Darby McGrath graciously helped out with my fieldwork. I am indebted to you both; I couldn't have completed my sampling without your assistance. Darby, thank you, additionally, for your feedback on methodology, analysis and various other aspects of this thesis. I was lucky to have you as a fellow student and roommate during the course of completing our Masters degrees together.

My family has unceasingly supported me in all of my academic endeavors and regardless of how endless or unfamiliar my degree may have seemed at times, you were always there when I needed you. I have always felt surrounded by your love and support during this journey.

To my husband Mark: thank you for the countless maps you've made for me, the days you spent sampling with me, and the many times you proof-read my work. You have been my rock over the last two years and have always reminded me of the great things I'm capable of. You won the race but I'm glad we ran it together.

I would like to dedicate this thesis to my father, the late Joseph Vincent. I know that you would have supported my academic pursuits and I wish you were here to see my accomplishments. I'd like to think that I inherited your love of nature and that what I'm doing is making you proud.

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Chapter 1.0: Introduction

Exotic plant species are both ecologically and economically damaging and the most successful approach to exotic plant species control is predictive rather than reactive (Hobbs & Humphries, 1995; Vitousek, 1990; Lonsdale, 1999a). Exotic plant species decrease the biodiversity of native plant communities through competition, predation or changes in ecosystem function. The spatial dynamics of the landscape are influential in the pattern of exotic plant establishment and spread. Ideally, if we could identify which exotic plant species are most threatening to a particular ecosystem, and which ecosystems are most susceptible to invasion (invasibility), then we could prevent the detrimental effects of exotic species establishment (Williamson, 1999; Kareiva, 1996; Rejmanek & Richardson, 1996; Hastings, 1996; Mack, 1996). Humans are the single most important large-scale cause of habitat fragmentation. As urbanization increases, the demand for land also increases. Humans are also undeniably responsible for the transport and introduction of introduced species, whose spread to new locations is influenced by the fragmentation of the landscape (Theorides & Dukes, 2007). It is becoming increasingly important to understand the ecological effects of fragmentation specifically in regards to the loss of biodiversity and changes in ecosystem functioning caused by exotic species. A better understanding of the effects of human-induced fragmentation on invasion patterns could decrease the critical threat that exotic species pose to both regional and global biodiversity (Wilcove et al., 1986). Bastin & Thomas (1999) analyzed the influence of spatial characteristics of habitat fragments on vegetation growth in urban regions and concluded that, “spatial dynamics at the scale of the landscape are of importance to the long-term persistence of many plant species in fragmented landscapes, and must be seriously considered in conservation planning and management.”

There has also been limited theoretical or empirical research on how landscape function and structure may affect the invasibility of natural areas (With, 2002). In 2000, Donna Havinga and the Ontario Invasive Plants Working Group released “Sustaining biodiversity: A strategic plan for managing invasive plants in Southern Ontario”, a report published by the City of Toronto. The plan recommended eight key strategies for the management of invasive plants, three of which were: the prevention of further introductions, the development of guidelines for managing priority species, and the identification of priority geographic areas for management. The overarching goals of the plan are to maintain healthy ecosystems and native biodiversity, reduce the ecological and economic impacts of invasive plants, to advance knowledge and societal attitudes about invasive plants, and to provide support and resources for land owners and Conservation Authorities. The Working Group recommended that landowners and Conservation Authorities take a systematic approach to identify areas within their jurisdictions that are priority geographic areas for management (Havinga, 2000). The Working Group also identified that there are, “few documented projects that specifically address species of concern in southern Ontario [and that] there are many unanswered questions and areas in need of exploration,” (Havinga, 2000). Included in the Working Group’s list of priority research topics, are best monitoring techniques, pre-invasion conditions for various species, and the screening for potential invasiveness of plants (Havinga, 2000). In their criteria for determining high priority geographic areas for management, the Working Group has identified public parks, ESAs and other protected areas (Havinga, 2000). However, in the nine years since its release, little progress has been made on how the landscape structure in Southern Ontario may affect the invasion process. Thus, there is an even greater need for further research into the prevention of future introductions and the identification of priority geographic areas for management. Additionally, cities will only be able to preserve a limited number of green spaces and the spatial configuration (particularly regarding the influence of patch characteristics on invasion patterns) of these natural areas should be given serious consideration.

My research examines the spatial distribution of exotic plant species in five parks in Kitchener-Waterloo, and uses the data gathered to test the feasibility of using landscape attributes as predictors of exotic plant species diversities. The City of Kitchener is currently developing a Parks Master Plan and I hope that my findings will be applicable to newly developed monitoring and management protocols for exotic plant species.

1.1 Research Questions

I am interested in the effect landscape fragmentation has on the diversity of native and exotic plant species in natural areas within Kitchener-Waterloo. My research will answer the question: do landscape and patch variables explain the existing invasion of exotic plant species in Kitchener-Waterloo's natural areas? And, if a long-term approach is taken, can landscape and patch variables predict the invasion of new exotic plant species in Kitchener-Waterloo's natural areas?

My research will also determine 1) the relevance of landscape ecology theory as a framework for the analysis for exotic plant species in an urban context; 2) the relative importance of community vs. landscape factors (specifically, how much of the variation in exotic species presence is explained by fragment characteristics and how much is explained by soil texture), and lastly; 3) the prospect of a management plan and monitoring priority list for Kitchener-Waterloo's urban natural areas.

1.2 Hypotheses

Based on the Island Biogeography and Land Mosaic theories of landscape ecology, I hypothesis that landscape variables (connectivity, edge to interior ratio, patch size and patch shape complexity) will significantly predict the presence or absence of exotic plant species in natural areas. Specifically, I hypothesize that native species richness will be higher in core sites than edge (perimeter) sites, and higher in larger and more connected natural areas. I predict that species richness of exotic species will be higher in edge sites than core sites, and higher in smaller and more isolated natural areas. I also predict that natural areas with more complex shapes and higher edge: interior ratios will have a higher presence of exotic plant species than those with simpler geometric shapes and a lower edge: interior ratio.

Additionally, I hypothesize that landscape variables (as a whole) will be a better predictor of exotic plant species presence than soil texture.

1.3 Organization of Paper

This paper begins with a Literature Review, which introduces ecological theories introduced as a conceptual and theoretical foundation. The Literature Review is followed by Methodology (an explanation and justification of methods), and Results (survey and statistical analysis findings) sections. The results are then discussed and evaluated (and related back to my hypothesis and research questions) in the Discussion and Conclusion sections. Ancillary information is included in Appendixes A-C.

Chapter 2.0: Literature Review

2.1 Overview

Exotic plant species are often invasive in response to landscape-scale (i.e. spatially explicit) disturbances that, in turn, create landscape and large scale changes that often reduce ecological integrity of the dwindling protected natural areas in urban ecosystems. This follows from – and integrates - With (2002) and Hobbs and Humphries (1995). With (2002) has identified the importance of landscape structure in determining invasive spread. Hobbs and Humphries (1995) argued that plant invasions have three components: the spatial and temporal dynamics of spread, the invasibility of invaded habitats, and the influence of human activities on both of these components. For the human component, people are either directly or indirectly (through land-use change which causes fragmentation and disturbance) responsible for the introduction of exotic species (With, 2002; Hobbs & Huenneke, 1992). Increasing urbanization and urban sprawl is resulting in habitat loss and isolation as land is being converted for human activities. This applies most clearly in urban ecosystems where these land use changes are complex, characterized by high levels of heterogeneity and often have biodiversity concerns related to the management of undesirable (invasive or exotic species) within the urban ecosystem (Savard et al, 2000).

2.2 Landscape Ecology

Landscape ecology can be simply understood as the merging of geography and ecology. The study of landscape ecology looks at the structure, function, and change of an interacting ecosystem mosaic in an attempt to determine how spatial patterns affect processes at the landscape level (Forman & Godron 1986, Turner 1989). On the relationship between landscape structure and function, Forman & Godron (1986) state “landscapes are heterogeneous and differ structurally in the distribution of species, energy and materials among the patches, corridors, and matrix present. Consequently, landscapes differ functionally in the flows of species, energy, and materials among these structural landscape elements.” The concept of a landscape is often arbitrarily defined but Forman & Godron (1986) suggest several characteristics that should be similar across a landscape. These included a grouping of ecosystem types that interact with one another, similar geomorphology and climate, and a shared set of disturbance regimes. Although the study of landscapes usually occurs at broad spatial scales, determining patterns of how spatial characteristics impact abiotic and biotic processes can occur at any scale (Turner, 1989). Definitions of the scale of a landscape should be independent of size, and instead reflect the ecological phenomena being studied, the interaction between patch and mosaic, and the corresponding management decisions (McGarigal et al. 2002).

2.2.1 Habitat as a Component of Landscape Ecology

Landscape ecology can help address the ecological impacts of habitat fragmentation (Collinge, 1996). Fragmentation entails the breaking up of a large piece of habitat into smaller pieces (patches) which often become isolated from one another. For example, Lord & Norton (1990) refer to fragmentation simply as “the disruption of continuity.” Fahrig (2003) reviewed the literature on the effects of habitat fragmentation on biodiversity and found that there are four main effects of the process of fragmentation: a reduction in total habitat, an increase in the number of patches, a decrease in the size of patches, and an increase in the isolation of habitat patches. However, one of the difficulties of studying the effects of fragmentation is that it can be difficult to determine whether the results of a study are due to fragmentation effects or simply habitat loss (a reduction in habitat usually occurs with fragmentation) (Fahrig 2003).

Fragmentation can also alter the remaining habitat in ways independent of habitat loss by causing changes in ecological processes (Fahrig, 2003; Haila 2002). Fragmentation studies are also not limited to a particular scale as the same scale of fragmentation will be sensed differently by different organisms (Lord & Norton, 1990).

2.22 Habitat Fragmentation Caused by Human Activities

Anthropogenic activities (conversion of forests to cropland, exploitation of freshwater resources, consumption of water and energy) have long been responsible for altering ecosystem processes and patterns. The transformation of land for development is no exception. Globally, humans are converting land at a rate to which the environment may never be able to adapt. According to Foley et al, (2005), "Such changes in land use have enabled humans to appropriate an increasing share of the planet's resources, but they also potentially undermine the capacity of ecosystems to sustain food production, maintain freshwater and forest resources, regulate climate and air quality, and ameliorate infectious diseases."

Ecosystem fragmentation generally refers to both habitat loss and isolation (Collinge, 1996) both of which are land transformations caused by urban sprawl. A common myth regarding the study of land-use change is that urbanization plays an insignificant role in global land-cover change because urban areas occupy less than 5 % of the Earth's developed surface (Lambin et al, 2001; Grubler, 1994). Yet, increasing urbanization leads to an increased 'ecological footprint' which puts more stress on our landscapes (Lambin et al., 2001). Haila (2002) strongly believes that "habitat fragmentation is viewed as a particular form of human-induced environmental degradation."

Increasing urbanization has also been linked to increased levels of atmospheric greenhouse gases resulting from both land conversion activities and the release of methane, carbon dioxide and nitrous oxide from soils to the atmosphere (Kalnay & Cai, 2003; Kaye et al., 2004). There is concern that as humans continue to contribute to global climate changes, some exotic plant species that prefer increased nitrogen disposition and higher levels of atmospheric carbon dioxide will out-compete native plant species (Moore, 2004; Dukes & Mooney, 1999).The ecological changes that are occurring within anthropogenic landscapes are only recently starting to be understood. A new field, known as land-change science, which is an integration of sustainability science and global environmental change, claims to unite the fields of social and natural sciences to better understand the impacts of land-use changes on ecosystem processes (Rindfuss et al., 2004).

2.2.2 Island Biogeography Theory & Land Mosaic theory

Two main theories have provided the framework for understanding landscape fragmentation within the scope of landscape ecology: the theory of island biogeography (MacArthur and Wilson, 1967) and the land mosaic theory. Although the equilibrium theory of island biogeography was originally proposed to predict the colonization of species from the mainland to oceanic islands, it can also apply to isolated fragments (terrestrial ‘islands’) in a matrix (MacArthur and Wilson, 1967). The size of patches in a matrix and their distance from source populations determine the rate of species extinction and rate of immigration of new species, respectively. The two processes are linked in a dynamic equilibrium where the number of species present will be determined by the turnover rate of both these processes. When viewing fragments as components of a heterogeneous landscape, Forman & Godron (1986) predicted that there would be a decreased abundance of species specific to the interior and an increase in the abundance of edge species and species which can coexist in multiple landscape elements.

Niemela (1999) argued that it may be difficult to apply the theory of island biogeography to urban habitats because there is no real ‘mainland’, and there are usually connected habitat patches throughout the matrix allowing for dispersal (true islands don’t have this connectivity). This seems to be a shared view amongst most landscape ecologists and many authors have suggested that fragments now be viewed as components of heterogeneous landscape (a paradigm shift to land mosaic theory) rather than terrestrial islands surrounded by hostile environment (Haila, 2002, Wiens, 1995, Diamond, 1976, Laurance, 2008). The theory of island biogeography has greater relevance (when used as a landscape fragmentation model) when emphasis is placed on a single type of patch and the surrounding matrix is considered to be neutral (McGarigal, & Cushman, 2002). However, as soon as one begins to think of the landscape as a matrix/mosaic of connected patches and corridors (a more realistic representation), this model becomes less useful (McGarigal & Cushman, 2002). Additionally, because the fragment is usually the focus, the matrix is often understudied, and more research is needed to determine whether the area which surrounds these fragments is conducive to dispersal or is actually an ‘ecological desert’ (Haila, 2002). The theory of island biogeography may also have limited application to the study of invasion processes because habitat patches are rarely replicates of one another. In the context of this project, both theories will be applied and each natural area will be viewed as a terrestrial island (that may be connected to other natural areas) in a mosaic of urbanization.

2.2.4 Fragmentation Effects

Fragmentation requires a landscape-scale focus of the composition, configuration and relationship of patches within a landscape (Dunning et al. 1992). Ewers & Didham (2006) suggest that the effects of fragmentation can be grouped into five categories that provide a better perspective of the spatial characteristics of habitat patches: structure of the matrix, patch area, patch shape, isolation/connectivity, and edge effects. Ewers & Didham (2006) remind us that these five categories are not independent of one another; not only do they interact with and possibly enhance the effects from another category but there are additional factors that may confound the effects of these processes over large scales (see also Fletcher et al. 2007).

2.2.4.1 Surrounding Matrix

Fragmented landscapes can also be thought of as habitat patches within a matrix. As defined by Forman and Godron (1986), the matrix is the most extensive type of element in the landscape For example, in a city, urban natural areas would be the patches and urbanization, the matrix. It has the greatest area, highest connectivity and the most control over dynamics. Rodewald (2003) also defines the matrix as “the background within which patches and corridors are embedded.” The matrix model also applies to cities; the focal points or patches can be considered the remnant natural areas in a surrounding matrix of urbanization. The quality of a matrix and the pattern of habitat patches within it affect the emigration and extinction rate of species both native and exotic (Fahrig, 2003; Ricketts, 2001). The designation of a matrix is dependent on the scope of the study, but for the purposes of this study the matrix will include all the land surrounding designated natural areas.

2.2.4.2 Area

Smaller and isolated habitats usually have higher extinction rates and can support smaller local populations. Saunders et al (1991) found that smaller fragments undergo more biogeographical changes than larger fragments. This is due to smaller patches having reduced habitat area, which can have negative effects on species richness, trophic structure, dispersal and colonization success, and genetic diversity (Fahrig, 2003). Birds, for example, are one of the best studied taxa and several studies have shown their abundances to increase with woodlot size (e.g. Hagan et al, 1996, Jokimaki & Huhanta, 1996). Harrison & Bruna (1999) conducted a review of fragmentation studies and concluded that habitat fragments support few habitat specialists and more generalists (usually characteristic of invasive species) and usually have altered biodiversity. Smaller habitat fragments cannot support equivalent species diversity compared to larger habitats and the loss of keystone species may cause altered ecological interactions and decreases in species diversity at lower trophic levels (Harrison & Bruna, 1999). Kolb (2008) found that when decreased area (caused by fragmentation) reduced the population size of plants, this had a negative effect on reproductive success by affecting plant-pollinator interactions, which is enhanced by herbivory. In the traditional sense of an island there is certainly evidence that plant species richness is positively correlated with island area (Kohn & Walsh, 1994).

The reduction of habitat area and decrease in species numbers is not as simple as the species-area relationship curve would make it. Also, many authors find this to be too simplistic a measure and to be scale-dependent (Connor & McCoy, 1979; Pimm, 1986, Pigolotti & Cencini 2009, Triantis et al, 2008, Lomolino, 2001, Scheiner, 2003, Dolnik & Breuer, 2008). Species most vulnerable to expiration are small populations with specialized niches. Holt et al (1999) suggest that there may be a relationship between species specialists and generalists and that at higher trophic levels, the species-area relationship becomes stronger. Although the authors admit several additional explanations for this relationship (e.g. energetics, open systems, non-equilibrium communities, strong top-down effects) the relationship is strongest and most consistent in closed communities such as those found in habitat islands.

The spatial configuration and size of these remaining habitat patches does not change linearly with habitat loss (Ewers & Didham, 2006). In fact, many researchers believe that there are "landscape thresholds" in fragmented landscapes which means that there are critical amounts of habitat below which the success of a certain species is unlikely (Radford et al, 2005, Ewars & Didham, 2006, Andren, 1994, With & Crist, 1995, Harrison & Bruna, 1999, Jansson & Angelstam 1999, Radford & Bennett, 2004). This critical value is dependent on the individual species and its sensitivity to fragmentation in addition to the spatial arrangement of habitat patches (With & King, 2001). Wu & Vankat (1991) developed an area-based model to describe the temporal dynamics of species abundances in forest islands and found that changes in species richness occur when the area of a forest is below a certain threshold (but this threshold is different for edge and interior species).

2.2.4.3 Shape Complexity

The shape complexity of habitat patches has been shown to influence woody plant colonization. Hardt and Forman (1989) showed that there was twice as much colonization by woody plants in forest patches with concave boundaries compared to those with convex boundaries. The authors proposed that the differences between concave and convex boundaries may be due to the influence shape complexity has on substrate conditions, microenvironment conditions near the fragment edge, woody species sources, herbivory and propagule success (Hardt & Forman, 1989). Patch shape is also important for the foraging success of many animal species which may serve as seed dispersers.

2.2.4.4 Isolation/Connectivity

Landscape connectivity can be thought to have both structural and functional components. Taylor et al (1993) identified landscape connectivity as, "the degree to which the landscape facilitates or impedes movement among resource patches," and With et al (1997) defined landscape connectivity as, "the functional relationship among habitat patches, owing to the spatial contagion of habitat and the movement responses of organisms to landscape structure." Since its introduction, the term 'landscape connectivity' has taken on a variety of definitions but after a review of 33 studies, which use the term, Tischendorf & Fahrig (2000), have concluded that Taylor's definition is the best explanation and that measures of connectivity must be based on each particular organism's movement through a landscape. This connectivity most frequently occurs through corridors which can be thought of as strips of land attached to a patch. Well-connected areas generally have lower probabilities of extinction because wildlife can move easier in a fragmented landscape when there is increased connectivity and fewer barriers to dispersal (Schippers et al. 1996). Landscape connectivity can also determine whether populations behave as sources or sinks: fewer individuals will disperse from a poorly connected source and, likewise, a well connected sink will attract more individuals (Taylor et al. 1993). A truly isolated patch (one with zero connectivity) is assumed to have no immigration or emigration of species.

Corridors certainly have conservation value even though their effectiveness is often debated (Beier & Noss, 1998; Kaiser, 2001). Although structural habitat corridors facilitate the dispersal of species and have the ability to function as habitat, it is important to recognize that they can also function as barriers and may potentially facilitate disease, fire or pest outbreaks (McGarigal et al., 2002; Adams & Dove, 1989). Either way, the establishment of corridors should not permit the establishment of smaller reserves (Frankel & Soule, 1981). The value of corridors is dependent both on scale and the dispersal ability of the target species for management. For example, Primack & Miao (1992) compared the dispersal ability of four annual plant species, *Plantago aristata, Abutilon theophrasti, Hypericum gentianoides* and *Impatiens capensis (*by experimentally dispersing their seeds at various distances from their source populations) and found that at 31 of their 34 sites, populations either died out or failed to establish. This indicates that these species may be unable to disperse naturally when presented with fragmentation barriers.

2.2.4.5 Edge Effects

Usually, the smaller a patch is, (or the more complex shape a patch has) the greater the influence of edge effects (Laurance & Yensen, 1991, Saunders et al. 1991, Malcolm, 1994). Edge effects can be explained as the ecological effects (commonly an interruption of ecosystem functions) caused by the exterior portion of a patch that is exposed to the matrix. The edges of forest fragments experience the most physical changes, which include changes in light, temperature, moisture conditions, vegetation structure, and a higher density of pioneer and exotic plant species (Collinge, 1996). It is important to recognize that the edge effects of a fragment are additive. This means that the total edge effect at a site is a sum of all edge effects at all points along all the edges of that fragment (Malcolm, 1994). As an example, a fragment that may have a relatively large area but has a high edge: interior ratio or a complex geometric shape will have high cumulative edge effects. An increased edge: interior ratio usually means a higher proportion of generalist species and a decrease in the proportion of rare and (or in the case of plants) shade-tolerant trees (Hill & Curran, 2001).

The width of an edge is often subjectively designated and Yahner (1988) suggests that "edge width is best defined, not by the investigator, but by the functional use of edges by wildlife" and "there is no consensus or standardized protocol for quantifying edge effect, but one is certainly overdue." Gates & Mosher (1981) compared structural edge (calculated by measuring changes in edge vegetation at three sites) and functional edge (calculated by measuring patterns of avian nest density at the same three sites) and found that this resulted in two distinctly different measures of edge width. Structural edge measure resulted in width of on average 12 m while functional edge measures resulted in an edge width greater than 40 m on average (Gates & Mosher, 1981). Matlack (1993) found that edge effects can be detected up to 50 m from the forest edge, which included changes in litter moisture, temperature, humidity and shrub cover. Ranney et al (1981) & Wales (1972) have also determined edge widths, of 50 m and 20 m, respectively.

2.3 Invasion Theory

The factors that determine the invasibility of communities and the invasion potential of species are still not fully understood and there is no unified theory that can be used to predict when and where invasions will occur. Several authors have contributed to the development of conceptual frameworks for various aspects and characteristics of the invasion process. Parker et al. (1999) address the lack of attention given to connecting ecological theory with measuring the impact of invasive species. They suggest the need for invasion ecology to build a framework which seeks to identify how the impact of an invasive species depends on its environment at multiple scales.

Sax et al. (2005) also recognize that we lack robust and well developed theories that integrate ecological drivers of invasion, and that we need, more specifically, "a broadly powerful conceptual framework for understanding species invasions, their impacts, and their management." Understanding evolutionary processes and how they relate to invasion patterns at the global level is necessary to build stronger invasion theories globally and regionally (Sax et al. 2005). Developing a predictive invasion theory has taken two approaches. Research has either focused on the characteristics of successful invaders and their interaction with native species (Kolar and Lodge, 2001; Levine, 2001; Davis et al, 2000) or examined the properties of habitats that affect invasion success, i.e. invasibility (Cummings, 2003; Lonsdale, 1999b).

The categories may affect the invasibility of habitats: community properties and landscape properties. Community properties include factors such as species diversity and species interactions (Levine 2000; Elton, 1958, Dukes 2002), resources availability (Davis et al., 2000), and disturbance (Hobbs and Huenneke, 1992). Experimental studies in invasion ecology have long focused on the community characteristics that determine invasibility and research focusing on the spatial analysis of invasion patterns has been insufficient. Recently, the role of landscape features in plant invasions has been brought to the forefront and subsequently so is the debate over the relative importance of community characteristics versus landscape characteristics. Again, community and landscape features do not act independently of one another in determining the invasiveness of a particular habitat.

Landscape properties influence invasive spread but With (2002) recognized the lack of theoretical and empirical research devoted to the landscape ecology of invasive spread, which she specifically describes as, "understanding how spatial pattern, such as habitat fragmentation or resource distributions, affects the various stages of the invasion process." Habitat loss and fragmentation certainly influences invasive spread but it is not known at what critical level of habitat loss spread occurs, the stages of invasive spread most impacted by fragmentation, and how landscapes should be managed or restored to mitigate invasive spread (With, 2002). With (2002) also developed a framework for conceptualizing how landscape structure may affect the process of invasive spread and described how each stage of the invasion process (introduction, colonization, establishment, dispersal and spread) is influenced by the spatial configuration and distribution of habitat. Landscape pattern may not be equally applicable for predicting invasive spread in all situations and may be related to the dispersal ability of a species but the differences between invasion ecology and landscape ecology are recognized (With, 2002, Bruno et al., 2004; Byers et al. 2002). Cummings (2003) predicted that differences in invasion rates depend on more than just levels of connectivity and that the more geometrically complex the shape of a habitat is, the higher the rate of invasion. This provides an additional example of the role of landscape features in the biological invasion process.

2.3.1 Invasion Ecology Definitions

As is the case with most expanding fields - and invasion ecology is no exception - there has been a proliferation of terms used to explain a variety of concepts and this has led to a lack of clarity and increased confusion (Richardson et al. 2000). There has also been academic criticism surrounding both the subjective labeling of exotic species and the negative connotations that some of the terms (i.e. 'invasive') have (Sagoff, 2005; Brown & Sax, 2004). After an extensive and critical research review, Richardson et al (2000) proposed a list of unified terms commonly found in invasion ecology. The authors urge that it is fundamentally important to agree on the terminology found in invasion ecology because a lack of consistency may compromise our research efforts. Richardson et al. (2000) along with Sauer (1998) recommend that the term alien (or exotic, exotic, nonindigenous) be given to plant taxa that are intentionally or accidentally introduced by human activity, and the term invasive be given to naturalized plants (alien plants with sustained populations that do not necessarily invade other ecosystems) that produce large numbers of reproductive offspring (seeds or other propagules) and have the potential to spread over a large area. According to Richardson et al.’s (2000) suggested terminology, an invaded site (or area of invasion) would be classified as an area where plants have spread from their introduced area within the region and are able to persist. The terminology used for this project will apply Richardson et al.’s (2000) recommendations. Colautti & MacIsaac (2004) also recommend that the term ‘invasive’ be applied to individual populations of a species and not to the entire species.

An invasible habitat or ecosystem will refer to a community in which an invasive species is able to survive and thrive (Burke & Grime, 1996). Generally, the term indigenous refers to native species specific to a local area; the term native can be slightly misleading because a species can be native to a country or region but not native to a local area. Additionally, Duguay et al. (2007) identified “introduced species” as, “a species that did not occur in the study area before European settlement and that arrived as a result of human activity,” and I will apply this same definition.

2.4 Exotic Plant Species: Reproduction and Dispersal Techniques

Life history traits are directly related to reproduction which in turn is related to survival. There is a link between life history traits and species invasive performances (NRC, 2002). The likelihood of the establishment of an exotic plant depends on a large number of forces and factors such as the number of organisms introduced, the chance of extinction, climate, phenotypic plasticity, competition for resources, reproduction traits and other life-history strategies (NRC, 2002). Undoubtedly, the patchiness of an environment will also confound the interaction of these forces. Dispersal is certainly an important component in invasive potential but must be examined in a landscape context, i.e. taking into consideration propagule distance and spatial pattern.

Some correlations between invasive success and life history traits have been found and although there is skepticism regarding the usefulness of these studies (see Crawley, 1986 and Williamson & Fitter 1996a, 1996b), it is still beneficial to examine these relationships. Reichard & Hamilton (1997) analyzed the attributes of invasive and non-invasive woody plant species to determine the invasion potential of new species. The authors analyzed fourteen characteristics (five of which included native range, invasion history, reproductive system, dispersal mechanism and seed size) and found that the ability to produce vegetatively and previous invasion history both corresponded positively with invasiveness. The applicability of these findings may be debatable seeing as most plant species found in natural forested areas are long-lived perennials that can undergo clonal reproduction (Honnay et al, 2005). Graae (2000) found that species with propagules that were ant-dispesered, dispersed through animal feces, and wind induced fruit swinging were more influenced by forest fragmentation than those species dispersed by adhesion to animals or wind. Invasiveness was also found to be higher in woody plant species that had shorter juvenile periods, smaller seed mass and a shorter interval between large seed crops (Rejmanek & Richardson, 1996). In an attempt to determine whether there was a relationship between biological traits and colonization patterns, Komuro & Koike (2005) compared the dispersal distances of mammal-, bird- and feces-dispersed tree species, a mechanically dispersed liana species, and species that are able to reproduce vegetatively. The authors found that although dispersal strategy affected colonization distance (there was no correlation between life forms or vegetative reproduction) the distance the seed had to travel from the seed source population was the most important determinant of colonization success. Similarly, Verheyen & Hermy (2001) found that the colonization of forest plant species is limited by both dispersal and resource limitation but the effect is species-specific.

2.5 Invasion Theory and Ecosystem Fragmentation Applied to Non-Native Plant Species

The invasion processes and patterns of exotic plants are understudied compared to their animal counterparts even though it has been suggested that introduced plants may cause more consequential impacts to their introduced ecosystems (Simberloff, 2005). Part of the difficulty in creating a hypothesis to predict the invasion success for plants is due to the variety of invasive plants that exist, and because most of the factors responsible for successful plant invasions are usually interrelated (Blumental, 2005). The role of spatial factors in invasive spread may be even less understood. Harrison et al. (2001) recognize that in respect to exotic plants, "little attention has been given to landscape factors, such as the spatial juxtaposition of different habitats among which organisms may disperse." There have also been difficulties determining predictors of invasibility because invasive patterns may differ in native ecosystems compared to disturbed landscapes (Hobbs & Huenneke, 1992). Davies & Sheley (2007) suggested a conceptual framework for preventing the spatial dispersal of invasive plants, and although their framework is based on the seed characteristics that promote specific vector dispersal, they suggest that an effective conceptual framework must integrate "the ecology of invasive plant dispersal with prevention management." Similarly, Hobbs & Humphries (1995) suggest an integrated, more effective approach to the management of plant invasions that focuses on the early detection of invasions by considering the characteristics of an invaded ecosystem (those factors which enhance invasibility) instead of the invader. The Committee on the Scientific Basis for Predicting the Invasive Potential of Nonindigenous Plants and Plant Pests in the United States (2002) reviewed the strength and weaknesses of current predictive systems and concluded that the history of a plant's invasiveness is currently the strongest predictor and that although the scientific principles surrounding invasive traits are weak, the potential for a conceptual hypothesis to be developed into a stronger predictive theory certainly exists.

Invasiveness research has focused on traits such as broad native ranges, rapid dispersal techniques, history of past invasions, phenotypic plasticity, and particular life forms or functional group (Alpert et al., 2000). Extensive research has shown that although some of these traits correlate with invasiveness, overall there has been a failure to provide relevant and useful generalizations between particular traits and invasiveness (Moles et al., 2008). Alpert et al (2000) suggest that it has been easier to determine the characteristics that are associated with an invasible habitat than the traits associated with invasiveness. Although it appears that the two are studied independently, there is an interaction between invasibility and invasiveness, as the success of the invasion by a exotic species depends on the probability that it will establish itself in a new particular habitat (Alpert et al., 2000).

Fragmentation also influences the invasibility of habitat and as habitats become increasingly disturbed and fragmented, they will become vulnerable to invasion by exotic species which will lead to further ecological degradation (Crooks and Soule, 1999). Recently, Minor et al. (2009) found that invasive species can spread easier through fragmented landscapes than native species, especially exotic species that are dispersed by animals. Metapopulation theory helps to explain plant species dispersal between patches in a matrix (Hanski & Gilpin, 1991). Although the landscape determines the degree of interaction, metapopulation theory focuses more on periodic extinction and colonization between sub-populations and predicts that the proximity, connectivity and number of habitat patches in a matrix will determine the colonizing success of dispersing species (Collinge, 1996; Hanski & Gilpin, 1991). The spatial arrangement of patches within a matrix can have implications for the dispersal success of species and the local populations of species with smaller dispersal distances are significantly impacted by patch configuration (Fahrig & Paloheimo, 1988). Although most plants have smaller habitat area requirements compared to other taxa (disregarding dispersal capabilities), population numbers are still dependent on the presence of other individuals of the same species which is influenced by availability of suitable habitat in a patch. Thus, patches may become too small to support viable populations of individuals (native or exotic).

Although the effects of fragmentation have been shown to be detrimental to the success of most species, it is not certain whether exotic plant species are always negatively or positively affected by fragmentation. As an example, the area and age of fragment size has been shown to have significant negative effects on native plant success (Soule et al. 1992) but is this also the case for exotic plants? Study results appear to be conflicting. Harrison et al. (2001) found that small patches and patches with a high edge: interior ratio supported a higher population of invasive plant species even though there was no difference in soil structure or composition between large patch interiors and small patches or habitat edges. However, Duguay et al. (2007) found a positive relationship between fragment area and the proportion of introduced species.

It is also generally recognized that the edges of forest habitats usually have a higher density of pioneer and exotic plant species (Collinge, 1996). In fact, Timmins & Williams (1991) looked at 234 nature reserves (in a fragmented landscape) throughout New Zealand and, after analyzing 15 variables, found that the most important characteristics for predicting invasive plant species presence were distance to urbanization, disturbance, patch size and species diversity. The authors also found that the natural areas with the greatest abundance of invasive plants were patches with higher edge to interior ratios and with strong human-influenced disturbance. The percentage of exotic species found in natural areas of different sizes is difficult to predict. Although, it should be expected that larger reserves would have a lower risk of invasion by exotic species, some studies have shown that larger reserve size favours a higher number of both native and exotic species (Lonsdale, 1999b). Shape complexity of forest habitats may also be a predictor of invasibility. Cummings (2003) considered how habitat shape influenced the dispersal of invasive populations and found that, although differences in invasion rates cannot be predicted by simple differences in overall connectivity, more geometrically complex habitats will have higher rates of species invasion. Connectivity, which creates important dispersal routes for species, is an important component in fragmented landscapes and has been shown to increase native plant species richness (and prevent establishment of exotic species) at large scales (Damshen et al. 2006).

2.5.1 Ecological Impacts of Invasion by Exotic Plants

In 1997, Walker and Steffen concluded that after land use change (habitat loss and landscape fragmentation), invasion by alien species is the next biggest threat to global biodiversity. The authors predict that alien species will continue to be a problem because of the globalization of economies and the increasing vulnerability of disturbed ecosystems to the establishment of new invasive species (Walker and Steffen, 1997). Exotic species are also destructive to our economy: Pimental et al (2000) estimate that the introduction of non-indigenous plants costs the US alone $137 billion per year. In Ontario, our native biodiversity is at risk from exotic plant species such as Norway maple (*Acer platinoides*), Dog Strangling Vine (*Cynanchum* or *Vincetoxicum* spp.), Garlic Mustard (*Alliaria petiolata*), and Buckthorns (*Rhamnus* spp.).

Plant invaders can cause a variety of ecological changes to the structure and composition of an invaded habitat. For a variety of reasons (species interference, competition, parasitism, niche exploitation, alteration of nutrient cycles, phenotypic plasticity, changes in soil development and hydrology), most authors agree that exotic plants are at an advantage and have the ability to displace native plants in fragmented areas (Chen et al., 2005; McIntyre & Lavoral, 1994, Liao et al, 2008; Funk 2008, Vitousek, 1990; Vitousek et al., 1987; Vitousek, 1986). Parker et al (1999) also provide an excellent review of the ways that invaders can impact and alter individuals (changes in demographic rates), genetics (altered gene flow or hybridization), populations (changes in abundance distribution and structure), communities (changes in diversity and trophic structure) and ecosystems (changes in disturbance regime, nutrient cycling, resource allocation).The probability of an exotic plant causing ecosystem-scale changes to a habitat are low but Vitousek (1990) suggests that this is possible if an invader is able to utilize different resources than a competing native species, and/or alter the trophic structure or disturbance frequency of the invading habitat. Regardless, Simberloff (2008) cautions fellow invasion biologists to ignore those who have called “invasion biology a thinly veiled form of xenophobia” because the risks and consequences of invasions are real; we must be aware of introduced plants that may be innocuous as well as native plants that may start to spread (Simberloff, 2008).

2.6 Urban Ecology

Increasing urbanization will require an increased focus on the management of green areas, or ‘urban ecology’. Over 50 per cent of the world’s population lives in cities and this is predicted to increase to 60 per cent by 2030 (Population Reference Bureau, 2007). Ontario’s population is projected to reach 14 million by the year 2016 and Toronto (and cities surrounding the GTA) will feel the brunt of this population growth (FON, 2001). Urban areas worldwide are experiencing impoverished biodiversity, and as urbanization increases, more and more people are going to lose the opportunity to benefit from the ecological, social and economic benefits it provides (Turner et al. 2004). Unless our communities find ways to accommodate this increase in population without resulting in urban sprawl, our cities' natural areas will be at risk of biodiversity loss and species invasions. If this trend continues, it suggests detrimental consequences for the conservation of native biodiversity, globally. Even though urban ecosystems increase the sensitivity of urban citizens to environmental issues and their understanding of natural ecosystems, urban systems are often given unaccredited value and are understudied, as a result (Savard, 2000). This may be because of the misconception that net productivity in an urban landscape is very low or negative (Forman & Godron, 1986). But urban ecosystems are often highly dynamic and complex and characterized by high levels of heterogeneity and because of this, studies from urban ecosystems can provide insights into the proper management of undesirable species within the urban ecosystem and other ecosystems (Savard, 2000).

Landscape classifications are arbitrary and subjectively designated, and as a result, a variety of classification schemes exist (see Johnson & Patil, 2006). One example is by Forman & Godron (1986) which classifies landscapes into five different types, with pristine, natural landscapes at one end and urban ecology (ecology in a matrix of high-density urbanization) at the other end. Most of the natural areas in Waterloo and Kitchener are a combination of suburban and urban landscapes, so for the considerations of this project, I will classify all natural areas found within the city as urban ecology. Urban ecology is the study of ecological systems caused by human activities (i.e. cities) and requires an integration of both the ecological (physical) sciences and social sciences (Grimm et al., 2000). In 1997, Vitousek et al., introduced the idea of human-dominated ecosystems and suggested that we cannot exclude humans from the study of ecosystems if we are to successfully and accurately understand how they function. Taking this one step further, Grimm et al, (2000) provided three reasons for studying human-dominated ecosystems. First, because humans dominate ecosystems they therefore should be included in models that attempt to understand these functioning systems. Secondly, developing models that incorporate humans will allow us to find solutions to a number of environmental problems. And, thirdly, the study of the city as an ecosystem will provide a valuable contribution to the study of all ecosystems. Additional research in the field of urban ecology is needed. It is important that we have a better understanding of some of the processes (and the associated interactions and feedbacks) that occur in an urban setting, something that has only recently been studied. This entails the study of both the biophysical and socioeconomic drivers that contribute to ecosystem dynamics in an urban setting (Grimm et al, 2000).

2.6.1 Urban Ecology as Manifested in Parks, Protected Areas, and Natural Areas

The U.S. Forestry Service describes an Urban Forest as trees located in urban areas that “counter the concrete jungle” and that are “dynamic ecosystems that provide environmental services such as clean air and water…improve social connections…create walkable communities.” Similarly, the City of Waterloo defines an urban forest as, “trees on city streets, rights of way and public lands,” (City of Waterloo, 1998). As outlined in their Urban Forest Policy, the City of Waterloo believes that an urban forest should be comprised of a diversity of indigenous species, serve to introduce wildlife habitats into an urban area, produce oxygen and absorb pollutants, reduce storm water run-off and prevent soil erosion, as well as providing aesthetic and social benefits (City of Waterloo, 1998).

Although there are multiple methods available for the selection of a natural area, most are not chosen/designated with respect to biological criteria and landscape issues (Nebbia & Zalba, 2007). The designation of natural areas in an urban context is often driven by previous agricultural and development decisions and the city of Kitchener-Waterloo is no exception. Regardless of previous designation, one should still analyze the best configuration and composition of natural areas in an urban context. Generally, the larger the area of a reserve, the more habitat is available for a variety of different species (with ranges of differing sizes) and the greater the probability of dispersal and colonization events. Diamond (1975) applied the theory of island biogeography to the design of nature reserves and proposed the following principles: larger areas support more species than smaller areas (even if the small separated areas are the same total area), connected and/or adjacent areas (to reduce the rates of extinction and promote immigration) are better than isolated areas and compact or clustered areas are better than linear areas. However this SLOSS (Single Large or Several Small Reserves) debate is an active topic in ecology and not all authors agree (see Wright, 1990). Simberloff & Abele (1976) promote several smaller reserves to encourage different groups of species and provide protection from disasters or disease. Reserve area may be a more important factor in the design of nature reserves because the proximity of natural areas is irrelevant for species (such as non-weedy plants) that cannot disperse through inhospitable habitat (Frankel & Soule 1981).

The design of natural areas in an urban area may depend on both the connectivity and the relative sizes of each area or “node”. Rudd et al (2002) used a model to determine the best connectivity network for 54 green spaces in the Greater Vancouver Area and found that the ideal matrix included 324 linkages which could serve as corridors for plants and animals. The authors also found that the success of smaller “satellite nodes” may depend on a larger “mother node” as a population source. This is because smaller natural areas have higher rates of extinction and rely on their mother node for the immigration of individuals (Rudd et al. 2002). Collingham & Huntley (2000) investigated the influence of fragmentation on the dispersal techniques of a wind-dispersed tree, *Tilia cordata* and found that migration rates were lowest in landscapes without larger patches (when habitat availability was less than 25% of landscape area). These results support the idea that landscapes should be composed of larger patches interconnected by smaller patches.

The minimum size for a natural area (based on vegetation) is open to interpretation. Levenson (1981) found that woody species richness increased with woodlot size up to approximately 2.3 ha, after which point it declined. This was the minimum size at which distinct edge and core ecotones existed. In their analysis, Holl & Crone (2004) considered forest fragments at least 0.25 ha in size in their analysis because they believe that smaller patches did not represent remnant vegetation conditions and could potentially misrepresent the results from overlapping GIS layers. And, Vestal & Heermans (1945) found 1.6 ha to be the minimum area for reference stands of mixed forest types.

2.6.2 Urban Natural Areas and Exotic Plant Species

Humans play a significant role in the dispersal of exotic plant species so it is not surprising that invasions by exotic plant species are more common in urban areas than forested or agricultural areas (Hodkinson & Thompson, 1997; Duguay et al, 2007). Though natural dispersal of plants over long distances does occur (see Cain et al, 2000), it is insignificant compared to human-mediated dispersal and there exists a long list of examples of both accidental and deliberate (e.g. for agricultural or landscaping purposes) introductions of plants throughout history (NRC, 2000;

Mack & Lonsdale, 2001). Commonly, improperly managed natural areas have an increased abundance of exotic species; therefore these areas should be given greater attention for management and monitoring (Coblentz, 1990; Nebbia & Zalba, 2007). We should strive for a better understanding of invasions in urban landscapes before focusing our attention on rural or pristine landscapes (Niemela, 1999; Hobbs 1988).

2.7 Scale Choices

2.7.1 Spatial

Defining the proper scale of an invasibility landscape study is essential. Yet, there is often confusion surrounding the proper choice of scale either because of the variety of terms related to the subject (see Dungan et al., 2002) or due to a previous insensitivity to the significance of scale (Wiens, 1989). This study will be at both a patch scale and landscape scale. Observations will be taken at single patches and collected data will allow for a landscape scale perspective. Inferences cannot be made to other landscapes because a patch scale is used, but this is a standard approach (Fahrig, 2003). In this project, reference to landscape scale does not imply that comparisons will be made between landscapes but that the information collected will allow for observation of a single landscape. Fragmentation is a process that occurs at the landscape scale (McGarigal & Cushman, 2002) and because this study is interested in the effects of habitat fragmentation and patch characteristics on invasion patterns, the scale of this study will be both at patch and landscape scale.

There are many different ways to define a landscape and a chosen definition usually depends on the scope of the study and whether it has an anthropogenic or wildlife perspective (McGarigal et al, 2002). Dunning et al (1992) refer to the landscape scale as, “the mosaic of habitat patches in which a particular patch (i.e., a "focal patch") is embedded" and suggest that a landscape perspective is necessary when studying ecological processes in a mosaic of habitat patches. This scale will allow for the observation of patch characteristics and connectivity in a matrix of urbanization. It will be assumed that patches are interspersed with uninhabitable areas and their boundaries are recognized by abrupt discontinuities in environmental characteristics (Wiens 1989).

Different scales can change the outcome of a study. Knight et al. (2005) found that at patch scale, native cover was positively correlated with invasive species cover. However, at the landscape scale, the opposite was true: native cover and richness were negatively correlated with invasive species cover. The authors suggest that landscape patterns of invasive spread should consider native diversity in addition to dispersal ability. Hulme (2008) also tested alien and native species-area relationships and found that relationships were significantly scale-dependent. Davies et al. (2005) proposed that at large scales, landscapes with greater spatial heterogeneity should have a positive correlation between species diversity of native and exotic species (spatial-heterogeneity hypothesis). Additionally, they found that the presence of both native and invasive species was positively correlated with abiotic conditions that were dependent on spatial heterogeneity. At the same scale, specialists will usually be more affected than generalists and as a scale becomes finer, edge effects increase and smaller organisms and more complex ecosystems are affected (Lord & Norton, 1990).

2.7.2 Temporal

The importance of temporal scaling is often overshadowed by spatial scaling, as indicated by the number of once-replicated or short-term studies. The processes of different ecological phenomena will follow different time trajectories but it is important that as spatial scales increase, time scales of these processes also increase (Wiens, 1989). Ideally, studies should be conducted at broad spatial scales over long periods of time.

Chapter 3.0: Methodology

3.1 Introduction and Criteria for Methodology Selection

I hypothesized that landscape variables (i.e. connectivity, edge: interior ratio, patch shape complexity and patch size) will significantly predict the presence of exotic plant species (trees, shrubs and herbaceous ground cover) in urban woodlots, and specifically that isolated and smaller patches will be most prone to invasion by exotic species (assuming all other factors are equal). To test this hypothesis, I measured these landscape variables on five forest patches within the cities of Kitchener and Waterloo. Soil texture was also measured to determine whether it has significance in predicting the presence and abundance of non- native plants.

This methodology is designed such that my dependent (response or measured) variable is the density and distribution of native and exotic plant species, and my independent (predictor or controlled) variables are my landscape metrics and soil texture.

My selection of variables to test if there is a relationship between patch characteristics in an urban landscape and patterns of exotic plant species was determined if the variables met the following set of criteria:

1) Does the chosen variable answer the project question?
2) Is the chosen variable predictable and practical?
3) Is the chosen variable quantifiable?
4) Is the chosen variable defendable?

3.2 Study Area

3.2.1 Region of Waterloo

Waterloo Region is located in Southern Ontario, Canada (Figure 3.1) and is comprised of three urban municipalities: Waterloo, Kitchener and Cambridge. The townships of North Dumfries, Wellesley, Wilmot, and Woolwich all surround the city. The region has a population of over 450,000 and covers an area of 1368.6 km^2 (Ontario Trillium Foundation, 2008). According to the 2006 census, the Region of Waterloo was the fourth fastest growing region in the province with a growth rate of 8.3 % per year (Ontario Trillium Foundation, 2008). The City of Waterloo has a population of 117,000, is located at 43.28°N, 80.31°W and has an elevation of 329m (City of Waterloo, 2008). Kitchener, has a population of 219, 000, is located at 43.27°N, 80.23°W, and has an elevation of 314m (City of Kitchener, 2007).Annual temperatures in the Region reach an average high of 11.8°C and an average low of 1.6°C (City of Kitchener, 2007). Average snowfall accumulation is approximately 195 cm whereas average annual rainfall is 765.0 mm (City of Kitchener, 2007). Kitchener-Waterloo has a mean growing season of 201 days (Presant & Wicklund, 1971).

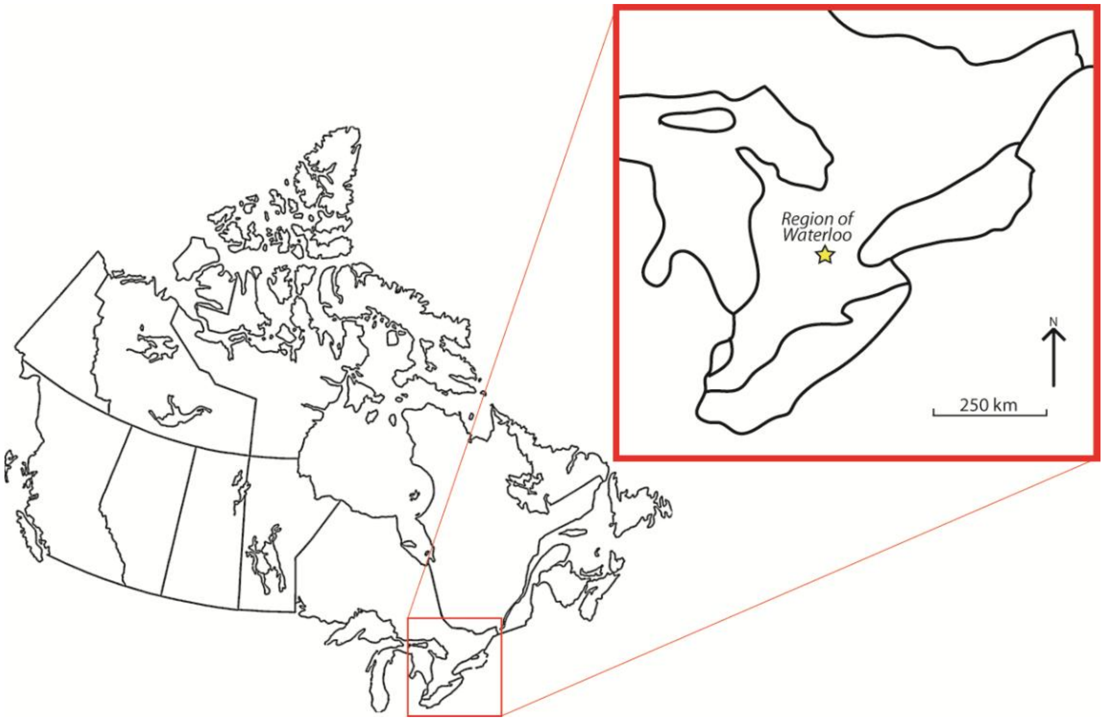

Figure 3.1: Map showing location of the region of Waterloo, Ontario, Canada.

3.2.2 Kitchener and Waterloo's Management Approach to Exotic Plant Species

Waterloo

The city of Waterloo considers itself an "environment first" community; a city with a focus on the environment and sustainable development. In 2002, the city released the Environmental Strategic Plan and one of its Phase One strategic actions included the expansion and protection of greenspaces (natural areas) and the monitoring of terrestrial resources (City of Waterloo, 2002). In recognition of the environmental, social and economic benefits of greenspaces, the city has identified potential strategic actions for their enhancement and management, including the creation of new healthy greenspaces, the protection and restoration of existing greenspaces and the continued monitoring of terrestrial resources (City of Waterloo, 2002). There are approximately 855 hectares of woodland (greenspace) in the city of Waterloo, 160 of which are acquired environmental lands (City of Waterloo, 2002). Although both cities are experiencing urban sprawl, Waterloo is suffering further from development because the sprawl is concentrated to the west of the cities near the Waterloo Moraine. Fortunately the number of woodlands has not decreased since 1997 (from 2002 when the report was published) but as growth in the city continues, many natural areas will be surrounded by development, making monitoring and best management practices essential (City of Waterloo, 2002).

The City is also focused on maintaining native biodiversity in natural areas by actively attempting to prevent the establishment and spread of invasive plants. Management activities include the reduction of conditions conducive to their spread, their removal, pruning and planting of competitive native species (City of Waterloo, 2009). In 1996, the City of Waterloo established Partners in Parks Program (PIPP) which provides opportunities for the community to be involved in projects (activities such as stream enhancement, tree plantings, litter clean-up) that enhance greenspaces throughout the city (City of Waterloo, 2007). And, one of PIPP's long-term projects includes the monitoring and removal of exotic invasive plant species (e.g. *Alliaria petiolata)* that threaten native biodiversity within greenspaces.

Additionally, public environmental lands in the Region of Waterloo are monitored every five years (both spring and fall) to monitor ecological health and to detect the establishment and spread of invasive plant species (City of Waterloo, 2007). Although these monitoring efforts are a step in the right direction, reliance on the public and a monitoring schedule of every five years may be insufficient in the detection of new exotic and invasive plants. It is my hope that the research and monitoring of exotic plant diversity studied in this project will contribute to a better understanding and an enhanced knowledge and data set for the Region.

Kitchener

In the City of Kitchener's Strategic Plan for the Environment (2005), the City recognized natural areas (which includes woodlots) as valuable components of natural heritage and environmental benefits. In response to a need for a new natural areas conservation program (there are over 1300 ha of natural areas in the city), the City of Kitchener recently established a Natural Areas Program (KNAP), a collaboration between Kitchener-Waterloo Field Naturalists, Waterloo Stewardship Network and the City of Kitchener, which focuses on educating and engaging the people of Kitchener in environmental issues and stewardship projects (City of Kitchener, 2009a). The City organizes trail guides and nature walks and has also started park series factsheets providing information about parks throughout the city.

In 2009, the City of Kitchener began developing a Parks Master Plan which would provide a framework for resources and a strategy for conservation and management of natural areas. The most up-to-date program is Kitchener's Woodland Management Program, which was created in 1994. McCartney (2005) reviewed this program in the context of invasive plant species and found that it was severely lacking in management strategies dedicated to prevention, control and monitoring. The Woodland Management Program failed to include long-term monitoring and risk assessment or address the need for prevention of future introductions.

3.2.3 Area Ecology & Geology

Kitchener-Waterloo is located in the Mixedwood Plains or Great Lakes-St. Lawrence vegetation zone of Canada. This zone is characteristic of a variety of deciduous trees (i.e. red and sugar maple, American beech, butternut hickory, white oak) and coniferous trees (i.e. red and white pine, eastern hemlock, white spruce) and occasionally shares some of the rare plant species from the Carolinian zone to the south.

The soil types found in Waterloo Country are reflective of glaciofluvial processes that occurred during the Pleistocene ice age, in which large quantities of unconsolidated glacial debris and till were deposited by retreating glaciers. This process resulted in the deposition of the Waterloo Moraine, which consists of sand and silt interspersed with clay and gravel on rolling topography (Present & Wicklund, 1971). The Waterloo Moraine shows variation in composition, as the soil in the southern section has a significantly higher sand and lower clay content than that of the northern end which has a higher loam content (Present & Wicklund, 1971)

3.2.4 Study Site Locations

There are 75 natural areas in the city of Kitchener and another 80 natural areas that the Region of Waterloo has designated as Environmentally Sensitive Policy Areas. I chose five of these natural areas for my study sites (Figure 3.2). Two were isolated from the surrounding landscape (Breithaupt Park and Huron Natural Area), two were relatively connected to nearby protected landscapes (Doon Regional Forest along the Grand River and Homer Watson Park), and one was used as reference states – which is at the city limits of Waterloo (Forested Hills ESPA in the Waterloo Moraine). Breithaupt Park is Kitchener’s oldest natural area and is 29.1 hectares in size and 16 hectares of this park is natural. The Huron Natural area is an approximately 107.5 hectare site and is the only site to include one of Kitchener’s only coldwater streams, the Strasburg Creek (City of Kitchener, 2009a). Homer Watson Park is approximately 70 ha in size and Doon Regional Forest is 29.1 ha in size. An undeveloped section of the Waterloo Moraine

(Forested Hills ESPA), which is 146.1 ha, will serve as a reference. Mueller-Dombois & Ellenberg (1974) outlined three requirements for the selection of a study area: “1) it should be large enough to contain all species belonging to the plant community; 2) The habitat should be uniform within the stand area, as far as one can determine this; 3) The plant cover should be as homogeneous as possible.” These fragments were selected because all have variations in soil conditions, fragmentation extent and shape, state of exotic plant invasions and fit the above-mentioned criteria. Four out of five of these parks (Forested Hills excluded) were comprised of discrete patches (separated by roads or waterways).

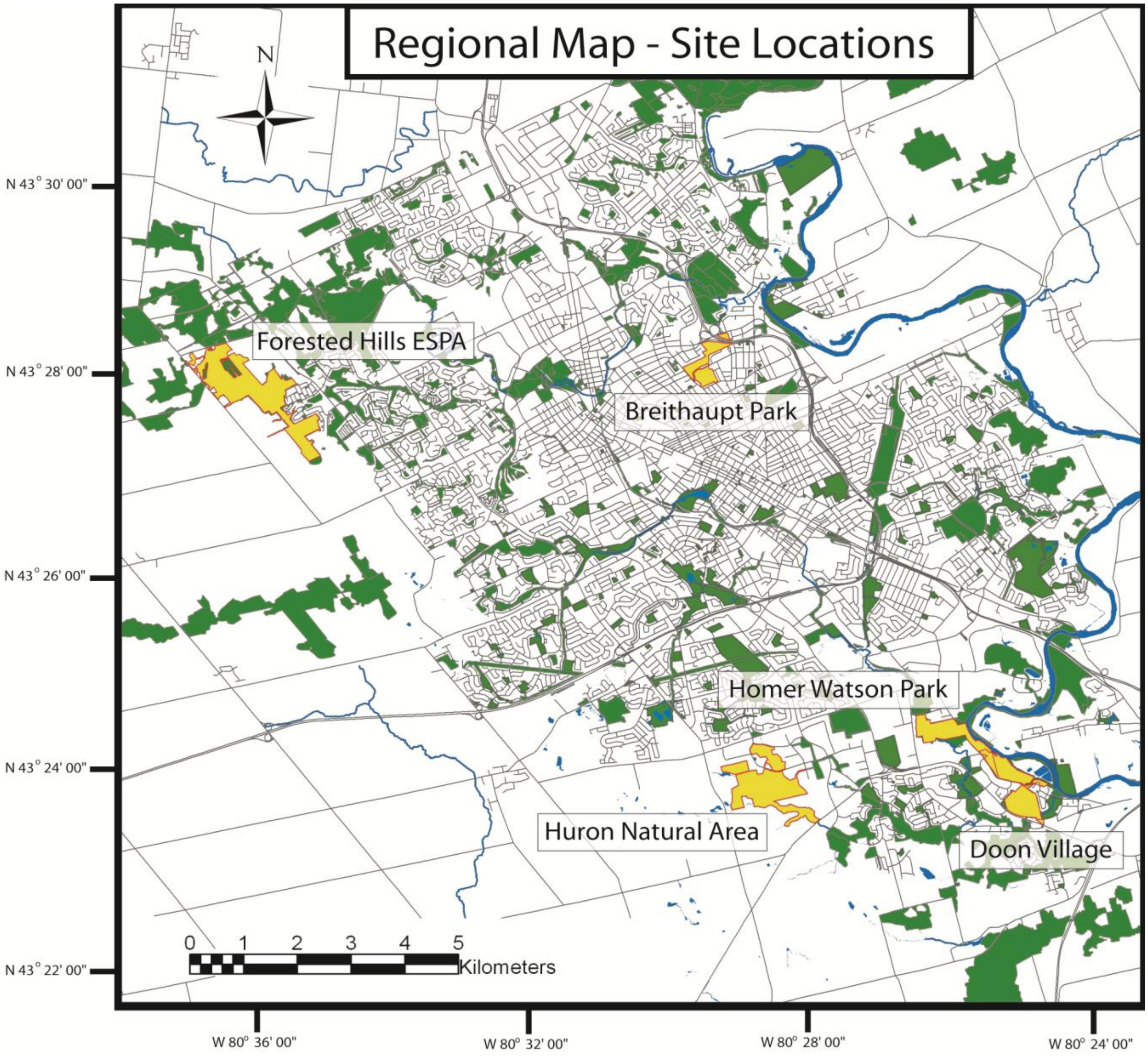

Figure 3.2: Regional map showing locations of the five sampled natural areas.

3.3 Vegetation Survey

A representative (systematic) vegetative survey was conducted to determine the ratio of native plant density to exotic plant density. A nested plot was created to measure density counts of herbaceous broadleaf plants, shrubs and trees. Herbaceous broadleaf plants were counted in a 1-m^2 area (placed on the outside corner of each of the 4 outside 5x5 m plots and on the northern corner of the middle plot), and shrubs and trees were counted in a 25-m^2 area; all of which was nested in a 400-m^2 area (Figure 3.3). The quadrants were designed so that they would include a random sample of the community under investigation and increased in proportion to the size of the plants (Williams, 1964).

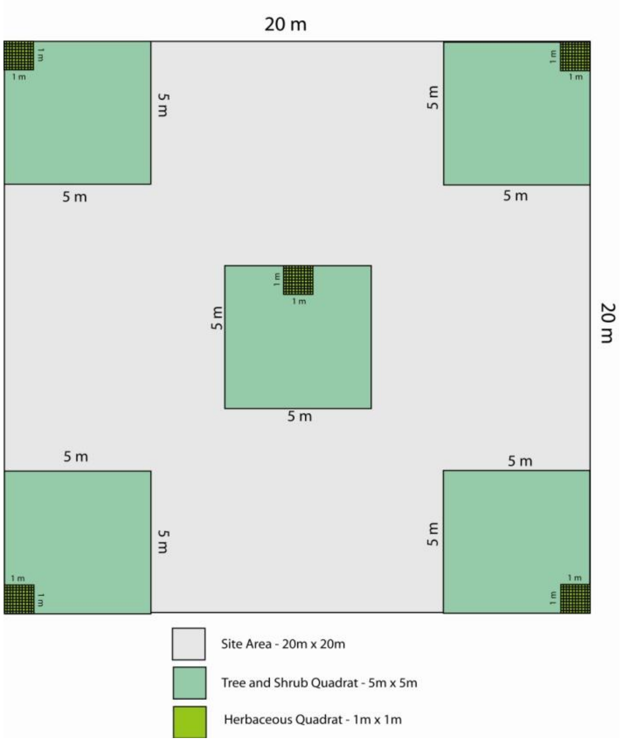

Figure 3.3: Layout of 1m x 1m quadrats nested in 5m x 5m quadrats, nested in a 20x20 m plot.

Plots were set up by pacing approximately 1 metre steps in each of the cardinal directions and stakes marked with flagging tape were used to mark the plot corners. Exterior (edge) plots were placed every 250 paces around the perimeter of the site (measured by paces) and interior (core) plots were determined by walking a N-W and S-E transect and placing sites 150 m apart.

Appendix A shows the plot locations of edge and core sites at each natural area. This sampling design was developed from existing approaches found in the literature and the Environmental Monitoring and Assessment Network (Environment Canada, 2007; Haber, 1997; Ohlemuller et al., 2006).

A plot frame was made using wooden dowels and pliable wire which divided the 1-m^2 plot into 100 sub-sections, and 10 (randomly selected) sub-sections were counted. Although Mueller-Dombois & Ellenberg (1974) claim “there is no absolutely objective guide either for determining the sample size or number of small quadrats”, a 1m^2 quadrat is the recommended size for density counts in the herbaceous layer of vegetation (Bonham, 1989). Herbaceous broadleaf plants were recorded if they were identifiable (i.e. more leaves than the cotyledon) and trees and shrubs were counted if they were greater than 30 cm in height. In the case of data discrepancies, where a species is recorded but no count is provided, a count of 1 was entered because it is assumed by the data interpreter that the species was found in the plot but the data recorder forgot to enter a count. Plant densities were counted rather than percent ground cover because plant densities also provide additional data about other invasive plants and will allow my analysis to be modified if necessary. One of the difficulties of the density method is the identification of an individual plant, particularly in the case of annual and perennial grasses, which may have multiple stems for each root system (Bonham, 1989). Grasses were counted but not identified beyond family level and therefore not included in the data analysis.

Consistent with many previous studies, Huebner (2007) found the stratified random sampling method to be the most accurate design for the detection of native and exotic plant species presence and patterns. Randomness is not always possible when sampling and is not convenient when samples need to be taken at regular intervals or at specific locations (i.e. at perimeter or interior sites) and systematic sampling provides an advantage in situations such as these. Additionally, systematic sampling is preferred because density measures can be mapped and used to compare plant densities to spatial data. Random sampling results in data with an increased variance that is easier to analyze but may result in larger errors than systematic sampling (Hill et al., 2005). Systematic sampling may introduce a bias if the vegetation sampled coincides with differences in microtopography (i.e. moisture, nutrients) and for this reason, systematic samples are not truly independent of each other. However, my statistical analysis will account for this relationship. Vegetation surveys were conducted in the spring of 2008 (May 7–May 27) to include the growth of spring ephemerals in the herbaceous layer species counts. Surveys were conducted by myself and at least one, but usually two to three, field assistants, all of whom were familiar with the identification of Ontario flora. Ancillary data collected and recorded for each plot included UTM location from a GPS.

Plants were identified to genus (or species if possible) and designated as native or exotic (Dickinson et al., 2004; Newcomb, 1977; Thieret et al., 2001; Kershaw, 2001; Soper & Heimburger, 1982; Farrar, 1995; Royer & Dickinson, 1999; Kaufman & Kaufman, 2007, Peterson & McKenny, 1968). Unknown species were verified and confirmed with experienced botanists.

3.4 Soil Data

Soil texture was measured to determine the percent clay, silt and sand and was always determined by the same individual (to ensure consistent results) using one or more of the following field tests: the feel test, moist cast test and ribbon test. Soil texture provides the ability to use a crude measure to determine if there is any trend or significant risk for one soil type to be invaded by certain exotic species or if native species are better able to resist invasion on that basis. Soil was identified using the *Field manual for describing soils in Ontario* (2003) from the Department of Land Resource Science (University of Guelph) and could not be compared against soil data for the Region because soil data only existed outside of city boundaries.

3.5 Spatial Variables

Landscape connectivity (or fragmentation) may be considered either an independent or dependent variable although it has been most commonly treated as an independent variable. Goodwin (2003) suggests that landscape connectivity be analyzed as a dependent variable when considering how movement behaviour relates to landscape structure. Alternatively, in determining how landscape structure influences ecological processes (e.g. population levels, species diversity, dispersal success), landscape connectivity is considered an independent variable (Goodwin, 2003).

My methodology required me to quantify how landscape patterns affect ecological processes, specifically how fragmentation influences invasive spread. Explanatory landscape variables were tested to determine their influence on the presence of exotic plant species using GIS spatial software and FRAGSTATS.

Shape files (orthorectified from digital air photos) were used in cooperation from the City of Waterloo's, City of Kitchener's and the Region of Waterloo's database (Woods_Waterloo_05, Parks_Waterloo_07.shp, Parks.shp, ESPA_Waterloo_03.shp, Region Natural Areas.shp, Waterloo Moraine.shp). Files were put into ArcView, and new polygons, which more accurately represented park edges were created by merging polygons from one or more of the above files. A new shape file was created which included my five study sites, and was then converted to a raster file (using signed integer with non-zero class values) before inputted into FRAGSTATS.

3.5.1 FRAGSTATS

FRAGSTATS is a program (available to the public) that uses spatial statistics to describe the characteristics of landscapes. FRAGSTATS has a literature embedded in ecological concepts and provides calculations that are fully integrated with GIS and easy to apply digitally (Raines, 2000). FRAGSTATS allows for the use of a built-in algorithm which uses mapped data inputted by a researcher and provides a user manual which fully explains the use and limitations of each and every metric and statistic involved (McGarigal et al. 2002). FRAGSTATS offers the ability to provide a new approach to a map analysis using easy-to-use, descriptive spatial predictions (Raines, 2000).

FRAGSTATS was used primarily with a focus on patch-level metrics. This means that the spatial character and context of individual patches were analyzed and these patch metrics then served as the computational basis for landscape metrics (McGarigal et al, 2002). The following metrics were measured at the patch level: patch size, patch shape complexity, edge: interior ratio, and core area.

Patch Area (and Core Area)

Patch size metrics were used as the basis for other patch and landscape indices, particularly for the proportion of edge effects which is dependent on both patch size and shape complexity. Core area (area within a patch which is not affected by edge effects) can be calculated using patch area and shape metrics. Although this metric is more important in identifying suitable habitat for species requirements, if vegetation surveys reveal distribution differences (of invasive plant species) between patch edges and the core, this will be a valuable metric.

The use of area metrics are mostly limited by the choice of scale. Because patch size metrics predetermine other patch and landscape computations it is essential that scale and resolution be kept constant. Core area metrics are also dependent on the chosen resolution, patch sizes and edge influences and are only useful if the dept-of-edge distance is measured and relevant to the study (McGarigal et al., 2002).

Patch Shape Complexity & Patch Edge

Total edge of each patch was calculated as well as edge density (edge as a per unit area) and both of these metrics are used to compute patch shape complexity. Patch shape complexity is a difficult measurement (due the infinite number of possible shapes) and is usually calculated as a perimeter–to-area ratio or fractal dimension then standardized using Euclidean geometry - geometry related to planes and geometric figures versus non-Euclidean, or hyperbolic, geometry - (McGarigal et al., 2002). Patch shape can also be measured using a medial axis transformation of the patch, linearity index, related circumscribing circle, and contiguity index (McGarigal et al. 2002). Shape metrics are also limited by the perimeter-to-area ratio method because patches with similar perimeter-to-area ratios may have different shape complexities and perimeter lengths may be biased higher in raster images (McGarigal et al., 2002).

3.5.2 FRAGSTATS constraints

There are certain constraints in the use and interpretation of metrics, such as those used in the FRAGSTATS program. First, the value of any one metric is a representation of the researcher's definition of landscape and chosen scale. The size of the chosen landscape can impact metrics computed from patch information within its boundary. Patch boundaries, for example, are dependent on scale and are artificially chosen (McGarigal et al, 2002). The format of data (raster vs. vector) may undervalue or overvalue metrics involving edge or perimeter as well as metrics based on cell adjacency information (McGarigal et al., 2002). The effectiveness of the FRAGSTATS program is also contingent on a proper understanding of the theory behind the use of metrics for the interpretation of landscape in addition to a choice of the proper metrics for the proper analytical reasons (McGarigal et al., 2002). Any metrics that are based on fractal geometry have limitations because applying fractal geometry to landscape patterns that are not truly fractal simplifies the pattern and may cause questionable results.

Metrics were analyzed by assigning each park as a separate class. This choice of analysis allows for shape and area metrics to be calculated for each park even if it comprised separate patches (e.g. separated by a road, stream or other barrier). This analysis, however, restricts the use of most of the landscape metrics which base their calculations on classes. Connectivity (which is measured using a cohesion matrix and a transversability index) determines the distance that can be reached from each focal patch to other focal patches of the same class (McGarigal et al., 2002). Proximity (or isolation) of focal patches is calculated similarly; the distance is calculated within an area of a defined radius between patches of the same class. Contagion is also a landscape metric and it provides a metric of the aggregation of patches but, again, it compares the clumpiness of patches of the same class and wasn't relevant.

3.6 Statistical Analysis

Parametric tests make the assumption that there is independence among samples but because of the spatial dependence inherent in ecological systems, most ecological data violate this assumption (Legendre, 1993; Anderson, 2001). For this reason, and because my data were non-parametric (confirmed using a Kolmogorov-Smirnov test), a Kruskal-Wallis test was used in this study. Nonparametric statistical procedures depend on a minimum of assumptions and therefore they are infrequently used improperly (unlike parametric tests), they can usually be computed quickly and easily and can be applied when the only data available for analysis is count or rank data (Daniel, 1978). Daniel (1978) explains the Kruskal-Wallis test as, “a weighted sum of squares of deviations of sums of ranks from the expected sum of ranks, using reciprocals of sample sizes as the weights.” Basically, the Kruskal-Wallis test (which is also known as a one-way analysis of variance) tests whether there is any difference in the mean ranks of independent samples of ranked data. Similar to ANOVA, Kruskal-Wallis can compare two or more samples but whereas the null hypothesis in ANOVA tests is that the means of populations (from the samples) are the same, the null hypothesis in Kruskal-Wallis tests is stochastic homogeneity (Ruxton & Beauchamp, 2008). Stochastic homogeneity means that, “each population is equal stochastically to a combination of the other populations,” (Ruxton & Beauchamp, 2008) but it can also be thought of as the population distribution functions being identical (Daniel, 1978). The use and interpretation of the Kruskal-Wallis test is correct only when it is used primarily as a test to evaluate the alternative hypothesis which is stochastic heterogeneity (Vargha & Delaney, 1998). Kruskal-Wallis makes certain assumptions of the data: that they consist of random samples, the observations are independent within and among samples, the variable of interest is continuous, the scale is ordinal and, the populations are only distinguished by a possible difference in location and are otherwise similar (Daniel, 1978).

An analysis of spatial pattern can be used to determine the effect of pattern on ecological process. In addition, although one cannot say with certainty that a particular pattern is responsible for a specific process (or vice versa), certain relationships can be deducted.

Chapter 4.0 Results

This chapter presents results of my study. Five parks, (a total of 98 20x20 m plots) were surveyed within the city of Kitchener-Waterloo (Table 4.1 and see Appendix A for site locations) and the goal of the vegetation survey was to provide a case study of the abundance of native and exotic plant species in natural areas within Kitchener-Waterloo. All upland wooded areas of each park were sampled. Huron Natural Area is comprised of variety of ecosystems other than upland forest (i.e. riparian, marsh and meadow), which were not sampled

Results have been subdivided into three main hypotheses. First, the relationship between site location (edge or core site) and diversities/abundances of native and exotic herbaceous, tree and shrub species richness was analyzed. Secondly, the relationship between soil texture and the diversity/abundances of native and exotic species was analyzed. Lastly, the relationship between landscape (spatial) attributes of each park and the diversity/abundances of native and exotic species was analyzed. Kruskal-Wallis tests were performed for each of these variables and carried out separately based on vegetation diversities and abundances. Spatial metrics were calculated based on an edge width of 20 m which was determined by the literature (see Gates & Mosher, 1981; Matlack, 1993; Ranney, 1981; Wales 1972) and, which also corresponds with the width of the sampling unit. Landscape measurements were also performed on patch sizes greater than 0.25 ha (see Levenson, 1981 and Holl & Crone, 2004) and an edge contrast set at 100%. This percentage was chosen because the parks are surrounded by urban landscape, which I deem to be inhospitable for plant dispersal. Most plants can only disperse a short distance even though the occasional long distance dispersal has been documented (Whigham, 2004; Cain, 1998). Distance metrics (and any metrics that were calculated based on landscape-level metrics) were based on eight nearest neighbor measurements and the landscape (the Region of Waterloo) was considered an open system.

Table 4.1: Number and type of vegetation sampling sites at each of the five parks sampled.

Park	Total # Edge Sites	Total # Core Sites	Total # Herb Plots	Total # Tree/Shrub Plots	Park Size (ha)
Breithaupt	9	7	80	80	16
Doon	8	9	85	85	29.1
Homer Watson	22	6	140	140	70
Huron Natural Area	4	4	40	40	107.5 (15ha upland section)
Forested Hills	18	11	145	145	146.1

4.1 Vegetation Survey

Total diversity (species richness) counts were calculated for native and exotic herbaceous, tree and shrub species for each park (Table 4.2, Table 4.3). In total, 19 exotic and 39 native herbaceous species and 11 exotic and 43 native tree and shrub species occurred in the 490 sampling plots at 98 sites. See Appendix B for a list of all species surveyed.

Table 4.2: Native and exotic herbaceous species diversities for each of the five parks sampled.

	All Sites			**Edge Sites**			**Core Sites**		
PARK	**Native**	**Exotic**	**Total**	**Native**	**Exotic**	**Total**	**Native**	**Exotic**	**Total**
Breithaupt	16	6	22	12	4	16	11	4	15
Doon	9	13	22	7	8	15	3	8	11
Homer Watson	24	9	33	23	9	32	11	3	14
Huron Natural	14	5	19	13	5	18	9	2	11
Forested Hills	16	6	22	15	6	21	14	1	15

Table 4.3 Native and exotic tree and shrub species diversities for each of the five parks sampled.

	All Sites			**Edge Sites**			**Core Sites**		
PARK	**Native**	**Exotic**	**Total**	**Native**	**Exotic**	**Total**	**Native**	**Exotic**	**Total**
Breithaupt	20	6	26	18	3	21	16	4	20
Doon	32	10	42	23	8	31	26	5	31
Homer Watson	32	5	37	29	4	33	23	3	26
Huron Natural	18	4	22	17	3	20	9	1	10
Forested Hills	18	4	22	16	2	18	15	3	18

Results of the Kruskal-Wallis test showed that when all 98 sites were analyzed collectively, native herbaceous, tree and shrub species richness was significantly greater in all core and edge sites. When each of the five parks was tested independently, this relationship was weaker; native plant species diversities were not significantly higher than exotic plant species diversities at core sites. However, native plant species diversities were significantly higher than exotic plant species diversities at edge sites, with the exception of Site 2 (Doon) which had greater diversities of exotic herbaceous species at its edge sites (Table 4.4). Additionally, there was no significant relationship found between any of the spatial variables tested and diversities/abundances of native or exotic plants (Table 4.4, Table 4.5). When species abundances were tested with spatial variables, the strongest relationships were found between exotic plant species and the # of disjunct core areas as well as total park area. However, when species diversities were tested with spatial variables, the strongest relationships were found between exotic plant species and total edge of the park as well as the shape complexity of the park. The probability threshold for significance was 0.05.

4.2 Soil Results

Results of the soil texture survey are found in Appendix C. Kruskal- Wallis tests also showed that there was no significant (probability threshold for significance was 0.05) relationship between soil texture and the diversities/abundances of native or exotic plant species (Table 4.4). The relationship between exotic herbaceous species and soils with sandy textures was stronger than the relationship between exotic tree and shrub species and soils with sandy textures.

Table 4.4. Kruskal-Wallis test results of hypothesis testing.

VARIABLES	HYPOTHESIS	P VALUE
Edge vs. Core : HERBACEOUS species	H1a: Native herbaceous species richness will be higher than exotic species richness in **core** sites	***P = 0.001 native >exotic in core sites.**
	H2a: Native herbaceous species richness will be higher than exotic species richness in **edge** sites	***P = 0.01 native > exotic in edge sites.**
	H3a: Native herbaceous species richness will be higher than exotic species richness in core sites at each park.	P= 0.276
	H4a: Native herbaceous species richness will be higher than exotic species richness in edge sites at each park.	***P = 0.05; except for Doon (exotic > native in edge sites)**
Edge vs. Core: TREE & SHRUB Species	H1b: Native tree and shrub species richness will be higher than exotic species richness in **core** sites	***P = 0.001 native > exotic in core sites.**
	H2b: Native tree & shrub species richness will be higher than exotic species richness in **edge** sites	***P = 0.001 native > exotic in edge sites.**
	H3b: Native tree and shrub species richness will be higher than exotic species richness in core sites at each park.	P = 0.443
	H4b: Native tree and shrub species richness will be higher than exotic species richness in edge sites at each park.	P = 0.497
Soil Texture vs. Exotic Species	H5a: There will be higher species richness of exotic herbaceous species in soils with sandy textures.	P = 0.552
	H5b: There will be higher species richness of exotic tree and shrub species in soils with sandy textures.	P = 0.761
Spatial Attributes* vs. Exotic Species.	H4a: Spatial attributes will predict the presence/absence of exotic herbaceous species.	P = 0.266 – 0.503
	H4b: Spatial attributes will predict the presence/absence of exotic tree and shrub species	P = 0.172 – 0.481

*Spatial Attributes are independently tested in Table 4.4

Table 4.4: Kruskal-Wallis test results testing the relationship between spatial attributes and exotic tree/herbaceous species abundances/diversities. (Note: Not all spatial metrics shown due to metric redundancy).

FRAGSTATS SPATIAL METRIC	DESCRIPTION	P VALUE*	P VALUE**
Total Area (ha)	Total area of each park	0.412; 0.281	0.168
Total Core Area (ha)	Total core area of each park[a]	0.284; 0.347	0.272
% of Landscape	Proportional area of each park to the landscape	0.291; 0.274	0.204
# of Disjunct Core Areas	No. of distinct core areas contained within each patch that comprises a park.	0.223; 0.266	0.153
Mean Core Area Index	Percentage of each park that is core area	0.285; 0.269	0.268
Total Edge	Total edge[b] length of each park	0.172; 0.376	0.294
Shape Complexity	Patch perimeter divided by the minimum perimeter possible for a maximally compact patch (correction of the perimeter: area ratio index)	0.197; 0.261	0.216
Mean Radius of Gyration	Mean distance (m) from between each cell patch and the patch centroid.	0.252; 0.462	0.274
Mean Frac	The logarithm of the patch perimeter divided by the logarithm of patch area, multiplied by 2.	0.333; 0.503	0.281
Perimeter: Area	The ratio of the patch perimeter to area.	0.226; 0.489	0.189
Mean Contiguity	A measure of the spatial connectedness of each patch that comprises a park	0.342; 0.368	0.196

The threshold for P = 0.05

*P Values calculated based on exotic tree; herbaceous plant species diversities

** P Values calculated based on exotic tree/herbaceous plant species abundances

[a] Core area is determined based on an edge width of 20 m.

[b] All edge metrics are calculated based on an edge contrast of 100%.

Chapter 5.0 Discussion

5.1 Spatial Attributes

No significant relationship was found between exotic plant species diversities (or abundances) and any of the spatial attributes tested. And, at all parks except for one (Doon), native plant species abundances and diversities were significantly higher than exotic plant species at both core and edge sites. Several studies have previously compared exotic plant species diversities and patch characteristics but a review of their approach and findings (Table 5.1) revealed a number of disparities in results and a need for a more standardized methodology. Although comparing similar studies is beneficial, it is important to recognize that comparing landscape studies across regions is difficult due to differences in geomorphology, disturbance regimes, climate, and management techniques.

5.1.1 Fragment Area

The results of my study are consistent with those of Pysek et al (2002) and Ross et al. (2002) who found that there was no significant difference in exotic species richness at different fragment sizes and to reference sites. The literature is divided on the exotic species-area relationship: Timmins and Williams (1991), Duguay et al. (2007) and Hobbs (1988) found that exotic species richness increases with increased fragment area while Ohlemuller et al. (2006) found that smaller fragments had higher exotic species richness at each plot (Table 5.1). Timmins & Williams (1991) suggest that “smaller reserves may have weeds throughout but larger reserves may have as many or more weeds because they tend to have a greater range of habitats available for weed invasion.”

Exotic species richness may also increase with native species richness and there have been studies that support this positive relationship (Pysek et al. 2002, Robinson et al. 1995, Levine & D'Antonio, 1999, and Wiser et al. 1998) and those that have found either no relationship or a negative relationship (Ohlemuller et al. 2006, Elton, 1958). If resources are not limiting, it is possible that exotic species do not have to outplace (or compete with) native species, they can simply co-exist (Huston & DeAngelis, 1994).

5.1.2 Connectivity/Isolation

Due to FRAGSTATS limitations (previously discussed in the methodology section), I wasn't able to qualitatively test for additional matrix effects such as connectivity and isolation and it is uncertain whether these factors would contribute to species diversity in an urban setting. Additionally, because all five of these parks are no longer interconnected with old forest, they may be considered isolated by several definitions (see Peterken, 2000 and Jacquemyn et al, 2003). In a similar study, Hobbs (1988) measured isolation (from aerial photographs) as the distance from each forest patch to the next patch that could contain at a minimum, a 2.3 ha circle (based on Levenson, 1981). Hobbs (1988) found that isolation was unimportant in determining species richness in urban forest patches because although urban forests may be isolated from other forest patches, they still have access to seed sources in the urban matrix.

The two connected parks, Doon and Homer Watson, had the highest total and native species diversities of tree/shrub species compared to the remaining three isolated parks (Table 4.3). However Doon and Homer Watson also had the highest number of exotic herbaceous species (Table 4.2). These results indicate that connectivity increases dispersal success (therefore increases species diversity) but that it also promotes the spread of exotic species.

5.1.3 Scale of Measurement

The scale of measurement may influence the results of exotic species richness assessments and a lack of consistent sampling methods may render a comparison of the findings from Table 5.1 ineffective . Both Ohlemuller et al. (2006) and Ross et al. (2002) suggest that total species richness for entire fragments may not be reflective of the species richness results per unit area (i.e. per 20x20m sampling plot). For example, Stohlgren et al. (1998) found a negative relationship between exotic species richness at the $1m^2$ level but positive relationships between exotic species richness and native species richness at 1000 m^2 and landscape scale. Honnay et al. (1999) examined the influence of patch area on species diversity and although they didn't consider exotic species, they suggested that instead of comparing species abundance with fragment area, z-values should be compared. However, this analysis is only applicable to normalized data, which wasn't the case with my findings.

There appears to also be an inconsistency in fragment size classifications, as well as sampling methodology in related studies (Table 5.1) Several authors fail to mention the area of their fragments (Pysek et al. 2002; Honnay et al., 1999; Dzwonko & Loster, 1992; Ohlemuller et al. 2006) or apply a subjective area classification range (Ross et al. 2002; Timmins and Williams, 1991). Of those studies that do include size of fragments studied, area ranges include 8-23 ha (Brothers & Spingarn, 1992). These inconsistencies make it difficult to accurately compare findings across studies. The lack of a significant relationship detected in my study between exotic species richness and fragment area may be due to a lack of spread of data. The smallest park sampled was 29 ha (ranging to 146 ha) and it is possible that more pronounced edge effects and species –area relationships would have been seen if smaller parks (some of which are essentially mostly edge) had been sampled.

5.2 Parks Analysis

My results lead me to infer two non-exclusive possible conclusions. The first is that native plant species in the parks/natural areas in Kitchener-Waterloo are successfully either outcompeting or keeping exotic plant species population levels manageable (based on results revealing that collectively native plant species diversities were higher than exotic plant species diversities at both edge and core sites). The second is that there has been enough time since the fragmentation of these parks that their core areas have become as equivalently invaded as their edges. This conclusion is supported by the results which showed that two of the five parks (Breithaupt and Doon) had exotic herbaceous species diversities in core sites equivalent to exotic herbaceous species diversities in edge sites (Table 4.2). Two of the five parks (Forested Hills and Breithaupt) also had higher exotic tree and shrub species diversities in core sites than in edge sites (Table 4.3). These are relatively old fragments and enough time may have passed to see the theoretical species-area curve equilibrate. Without previous years' vegetation data, only assumptions can be made regarding the current status of these parks.

Macdonald et al (1988) suggested a simple formula to calculate a relative index of invasion and Usher (1988) modified this equation to include all introduced species (not just invasive).

$$W = 100\ N/S$$

Where N = the number of all introduced species and S = the total number of all species.

Although my goal was not to obtain a complete species inventory but rather to gain a representation of the exotic species richness, this formula is limited but it is still relevant for comparisons. I found 19 exotic and 39 native herbaceous species and 11 exotic and 43 native trees and shrubs in my survey. Applying Usher's equation to my findings, results in a 33 % invasion rate of herbaceous species and a 20% invasion rate of tree and shrub species. Collectively, the invasion rate for all species was, 26.5 % which is lower than the 36% invasion rate found by Hobbs (1988) but higher than the 18% invasion rate found by Howard et al. (2004) who used similar sampling methods to compare forest invasibility in southeastern New York (a true comparison cannot be made due to differences in sampling methodology). Duguay et al. (2007) and Pysek et al. (2002) also found comparable invasion rates of 16% and 17% respectively (see Table # for sampling methodologies). In 2004, according to the Invasive Alien Species Strategy for Canada, alien species comprised 27% of all vascular plants in Canada. The report does not clarify how this figure was calculated and whether both developed (urbanized) and undeveloped (intact forest) areas were included but based on this percentage alone, the invasion rate of Kitchener-Waterloo lies slightly below the national average (Environment Canada, 2004).

5.3 Edges and Disturbance

Edges are the first point of contact for a dispersed invasive species and can serve as barriers that discourage the invasion of exotic plants into the interior (Brothers & Spingarn, 1992). Cadenasso

& Pickett (2001) performed a study where they experimentally altered the vegetation within 20 m of the forest edge and compared its effect on seed dispersal to an intact edge. The authors found that that more seeds crossed the altered (thinned) vegetation layer than the intact forest edge and were able to travel further into the forest interior. Of all the parks sampled, except for one, there was no significant difference in exotic species diversities in edge sites compared to core site, i.e. native species diversities were significantly higher in edge and core sites (Table 4.4). These results may indicate that the edges of the surveyed fragments are relatively intact and as a result are functioning as an effective barrier. However, if these parks were resistant to invasion, you would see a sharp decline in exotic species richness inward from the forest edge. Two of the five parks had exotic herbaceous species diversities in core sites equivalent to exotic herbaceous species diversities in edge sites (Table 4.2). In the future, additional attention should be paid to the role forest edges play as barriers to invasion.

Although I did not directly study the effects of disturbance, it has been shown to be an important factor in invasion success and several studies have shown that the greater the anthropogenic disturbance (visitation rate) of a site, the higher the invasion of exotic weeds (Hobbs & Huenneke, 1992; Ross et al, 2002; Pysek et al. 2002). In a study which looked at urban stressors on avian communities, Schlesinger et al. (2008) found that disturbance from human activity (activities the authors identify as an increased flow of people, vehicles and domestic animals) was more important than habitat loss from urban development. According to its Regional Forest Management Plan, Doon suffers from frequent dumping of wastes (from surrounding households) and has high trail usage (Region of Waterloo, 2009a). Therefore, it is possible that a higher level of disturbance in this park makes it more susceptible to invasion.

5.4 Invasion Bias

It can be argued that because there is a lack of objectivity in studies reported in the invasion literature (Brown & Sax, 2004, Larson 2007, Larson 2005), there may be a bias towards positive results (i.e. the detrimental effects of exotic species) and as a result, negative results are less often published or acknowledged. In a highly publicized and debated field like invasion ecology, not only can publication bias result in a loss of scientific credibility but it undermines our ability to search for scientific truth and protect our native biodiversity from potentially detrimental exotic species.

Table 5.1: Review of studies that have compared exotic plant species diversities and landscape characteristics

Reference	Fragment Area & Exotic Plant Species	Fragment Shape (Edge: Interior) & Exotic Plant Species	Methodology	Additional Findings
Brothers & Spingarn (1992)	n/a	Frequency of alien species decreased from 86% at the exterior to 22% at the interior;	Rural, upland and mesic forest patches (8 – 23 ha); transect sampling	Increased invasion on warm vs. cool edges.
Ohlemuller et al. 2006	Negative (smaller fragments have more exotic species per plot).	Edge plots had higher abundances of alien species than interior plots.	Temperate patches, predominantly agricultural landscape; 20x20 m plots (143 total sampling plots)	No relationship found between native species richness and exotic species richness
Timmins & Williams (1991)	Ambiguous	As E:I increased, exotic plant species richness increased.	234 reserves in agricultural, urban and natural settings (based on literature review not survey). Classified as small – medium sized (<500 ha).	Proximity to town and anthropogenic disturbance significantly predicted invasion success.
Ross et al. (2002).	No correlation	No correlation	50 forest fragments; 3 class sizes. Small <1 ha, medium 1-10 ha, large 10-100ha. 25m2 sized plots (with 4 5x5 m subplots).	Disturbance affected exotic species richness more significantly than fragmentation.
Pysek et al. (2002).	Weak correlation	n/a	Species lists from 302 nature reserves compiled from published records.	Exotic species richness positively corresponded with native species richness; suggested that the two groups do

				not compete.
Hobbs (1988)	Positive	n/a	30 stands in residential areas (1-8.6 ha); Stratified random sampling - each site searched thoroughly and plant species listed.	Exotic species richness not influenced by isolation.
Duguay et al. (2007)	Positive	n/a	44 small remnant forest fragments (3- 7.5 ha); A grid of 1m2 quadrats covering the entire fragment (average of 21 quadrats per forest fragment).	40% more introduced species in urban fragments than agricultural or forested fragments.

5.5 Soil

Soil texture was tested because if a significant relationship was discovered, this would allow for a relatively easy monitoring protocol. In theory, exotic plant species should have better success in finer soils rather than coarser soils because soil texture determines the infiltration rates of water and nutrients and finer soils are able to retain more water and subsequently more nutrients. Stohlgren et al. (1998), Goslee et al. (2001) and Lundgren et al. (2004) all found a positive relationship between soil percent silt and exotic species richness.

My results showed that there was no significant relationship between a particular soil texture and the abundance or diversity of exotic tree, shrub or herbaceous plant species. In an invasibility study to mine, comparing the factors affecting the invasion of exotic plant species, Ohlemuller et al. (2006) measured soil pH, fertility, and texture (gravel content) and also found no detectable relationship between soil texture and exotic plant species richness. Howard et al. (2004) also found that soil texture was not a good predictor of invasion success.

Soil texture (specifically sand content) may influence the success of exotic conifer trees more than deciduous trees because this has been shown to be the case with some native conifer woodland species (Dzwonko & Loster, 1992). This may be related to the increased acidity of the soil in conifer woodlots and the positive relationship that both Chaberie et al. (2007) and Ohlemuller et al. (2006) found between exotic species diversity and acidic soils. This relationship may also explain why I found the highest proportion of exotic herbaceous and exotic tree species at Doon, compared to the four other woodlots. Doon, unlike the other natural areas was comprised predominantly of coniferous trees (Doon was acquired in 1949 by the Waterloo County and a conifer plantation was established after acquisition) and it is possible that the acidic soils of this area encouraged the invasion of exotic plant species. Interestingly, the conifer plantations also included several non-native species, such as Jack Pine (*Picea banksiana*) and Norway Spruce (*Picea glauca)* (Region of Waterloo, 2009a). The planting of these species may have contributed to higher diversities of exotic species in Doon compared to the other parks studied. This signifies the importance of previous land-use history in invasion studies.

Soil fertility may be a better indicator of invasion success than the physical characteristics of the soil and many studies have shown that exotic species are better able to establish and spread in high fertility or nutrient enriched soils (Lake & Leishman, 2004; Wiser et al, 1998; Howard et al, 2004; Timmins & Williams 1991; Ohlemuller et al. 2006). Lake & Leishman (2004) specifically found that there were significantly higher numbers of exotic plant species at sites that received nutrient and water enrichment although they believe soil enhancement to be the more significant factor. Exotic plant species may create a positive feedback as they change soil nutrient availability to promote their proliferation and growth, as has been shown to be the case in the invasive species, *Alliaria petioloata* (Rodgers et al. 2008).

Soil classifications were kept consistent with the texture classes I used as a reference. However, if I were to assign soil texture based on less specific classifications (i.e. designated soil by its predominant texture - clay, sand or silt), I may have seen a stronger correlation between texture and exotic plant species presence. The soil surveyed in this thesis was almost always a combination of two or more textures and, I think it would be a difficult and misguided choice to change the classification scheme in order to look for trends.

5. 6 Remnant Trees in the Urban Matrix

The role of remnant (isolated) trees within the urban matrix is often ignored but they should be integrated into landscape–level management and monitoring policies and programs. According to Manning et al. (2006), their ecological importance in human-modified landscapes has yet to be acknowledged and appreciated. Their role in the connectivity of the matrix is two-fold: they promote the succession and re-establishment of forested areas of plant species but also serve as habitat (perching structures), and a food sources for mammals, and birds dispersing from one patch to another (Herrera & Garcia, 2009). Herrera & Garcia (2009) found that remnant fruit trees facilitate the succession of forested areas by promoting dispersal of plant species (by frugivores) throughout the matrix. And, although Herrera & Garcia (2009) make this case for remnant trees, planted native trees would serve the same purpose. Hobbs (1988) believes that urban forest fragments, although not connected or distant to remaining forest are not isolated from seed sources, and these trees may compensate for the lack of connectivity. “They have a disproportionate effect on the ecosystem relative to the small area occupied and low biomass of any given tree and the low density of scattered trees collectively,” (Manning et al., 2006). Similarly, backyard habitat and gardens are also important as corridors of connectivity throughout urban areas and provide dispersal linkages for plant species (Rudd et al., 2002). Rudd et al., (2002) performed a connectivity analysis of green spaces in the Greater Vancouver Area in British Columbia and determined the total combined area and the number of linkages as well as identifying important dispersal nodes. It would be beneficial to see a similar case study performed on the cities of Kitchener and Waterloo to identify the overall connectivity and isolation of natural areas in the cities.

The important role that isolated trees play in the urban matrix has implications for the City of Kitchener-Waterloo and their urban forest policies. In 2007, Weaver gauged the condition of the street tree population in the City of Kitchener and found that 52% of the street tree population was exotic species but only 9% of the forest (in nearby Breithuapt Park) species was exotic. At the time of study, Weaver (2007) concluded that *Acer platanoides* was not currently an invasion threat but may be one in the future. And, according to my results (Table 4.3) the percentage of exotic tree species in Breithaupt Park has increased since the time of her study. In the City of Waterloo's Urban Forest Policy (1998), it recommends that trees indigenous to the region be planted in areas that are in close proximity to natural areas. This certainly is a step in the right direction but my results lead me to believe that the invasion success of these street trees is increasing and that the City should plant native street trees not just in close proximity to natural areas but everywhere in the city.

5.7 Beyond Island Biogeography Theory

Hobbs (1988) concluded that the IBT theory is irrelevant in an urban setting because it ignores patch shape and focus3s only on area and isolation, the latter of which she proved to be unimportant. IBT provides a good theoretical understanding of dispersal techniques on insular islands but it may be irrelevant when it comes to the management, policy and monitoring of natural areas (Boecklen & Gotelli, 1984). Gilbert (1980) believes that the IBT has "little support for its tenants" and its application to "nature conservation and ecology is therefore premature."

Brown & Sax (2004) suggest a unique approach when applying the theory of island biogeography and metapopulation theory to the colonization of new exotic species. They suggest that the theory of island biogeography can be thought of as an intermediate view between two extreme views of the carrying capacity of ecosystems. The first is that biodiversity is nearly at its carrying capacity (species saturation approach) and according to this view, ecosystems should resist invasions or a dynamic immigration/extinction rate should be reached (Brown & Sax, 2004). The second is that most ecosystems are below their carrying capacity and according to this view, ecosystems should be susceptible to invasion (Brown & Sax, 2004). The theory of island biogeography then provides an intermediate theoretical approach because it predicts that invasion rates will be higher in isolated habitats as a result of human introduction which has caused increased colonization rates (Brown & Sax, 2004).

It is possible that fragmentation is too simple a concept especially in an urban setting because it tends to ignore additional possible landscape configurations and does not take into consideration multiple types of habitat. Another way of viewing fragmentation in an urban matrix might be a continuum of fragmented, variegated, and relict landscapes, as suggested by McIntyre & Hobbs (1999).

5.8 Biological Characteristics and Dispersal Traits

The literature is inconclusive regarding the relevance and accuracy of comparing life history traits and invasion success but it still may be beneficial to look for trends. The success of invasive plant species cannot be explained by a single characteristic but a combination of traits may be responsible for increased invasion success. Based on a literature search, six main invasion traits have been identified. These traits are: vegetative reproduction (Reichard & Hamilton, 1997; Baker 1974), invasion history (Reichard & Hamilton, 1997), dispersal type (Graae, 2000; Baker 1974), dispersal distance (Komuro & Koike, 2005; Baker 1974), smaller relative seed mass size (Rejmanek & Richardson 1996; Cadotte & Lovett-Doust, 2001), and larger specific leaf area size (Lake & Leishman, 2004, Baruch & Goldstein, 1999, Grottkopp et al. 2002; Smith & Knapp, 2001). Of these six, two of the most useful traits are invasion history (previous invasion success) and the ability to produce vegetatively (Kolar and Lodge, 2001; Williamson, 1999).

It is difficult to draw correlations between certain biological characteristics and invasion potential because generally there is a lack of information about these attributes for most plant species. For example, in an attempt to predict invasiveness of plant species (165 invasive species and their non-invasive paired species) based on biological information, Goodwin et al. (1999) reviewed published flora characteristics, and were only able to find information for three biological characteristics: life form, stem height and flowering period. Although, it's generally agreed that an intensive data set of species or genera specific biological information (used to derive invasion traits) would be beneficial, the cost : benefit ratio of such an undertaking is highly debated (Daehler & Strong, 1993; Lodge 1993). Additionally, the characteristics of exotic species populations may be more useful than individual characteristics (Williamson, 1999). I agree with Daehler & Strong (1993), that "predictions from these approaches may overestimate invasion success, but the low cost of generating these predictions (compared to the potentially high cost of losing endemic communities to exotics) may make them the best available for molding management policies."

5.8.1 Identifying New Invaders

We shouldn't only focus on potential exotic plant species outside of Canada's borders; species can just as easily move across a provincial border or into a new ecosystem and become invasive (Environment Canada, 2004). Additionally, with climate change and ecosystem fragmentation, we should also be attentive to current native or indigenous species that have the potential to become invasive. This has already been shown to be the case in Southern Ontario with native species like *Parthenocissus quinquefolia* (L.), *Toxicodendron radicans* (L.), and *Ambrosia artemis* (L.).

Canada desperately needs the development of a strong invasive species prevention strategy. According to the Alien Species Strategy for Canada, “taxonomic benchmarks and expertise for the proper identification of invasive alien or native species are essential, but currently lacking in Canada, a gap that undermines Canada’s insecurity.” (Environment Canada, 2004)

5.9 Additional Indicators of Fragmentation

Plants are valuable indicators of fragmentation but non-vegetation variables should also be used to determine the influence of fragmentation on the biodiversity of urban natural areas. Birds, for example, are very usual indicators because they are particularly sensitive to habitat changes. Frieson et al (1995) found that as the size of the forest fragment increased, so did the number of neotropical migrants and the authors found that the number of neotropical migrants decreased with increasing urban development (determined by the number of homes bordering forest fragments) regardless of the size of the fragment. In another Ontario study, Zajc (2005) also found that a 10% increase in urbanization (or a 10% reduction in habitat) would result in a 20% decrease in native avian species diversity.

5.10 Scale

There was no temporal dimension to this study; the vegetation was surveyed during the course of one spring season. Temporal and spatial scales should be studied together at a variety of scales as it’s probable that different species respond uniquely to different scales. Haila (2002) cautions that ecological monitoring is problematic because it’s an attempt to sample an ecological process at a particular temporal and spatial scale and most (if not all) ecological processes occur at different spatio-temporal scales. Also, one of the limitations of current invasion studies is that they have only focused on one (often local) spatial setting even though ecological changes caused by plant invasions are scale-dependent (Pauchard & Shea, 2006; Radosevich et al. 2003). More studies need to take a landscape scale or multi-scale approach if spatial ecology is to provide efficient answers to the patterns of invasive spread (Pauchard & Shea, 2006).

Chapter 6: Conclusions and Recommendations

6.1 Conclusions

Exotic plant species have the potential to disrupt ecosystem structure and function and once they have established or spread, their control or elimination is difficult. Prevention, rather than reaction is always the better scenario when dealing with potential new exotic plant species. The objective of this study was to examine the spatial distribution of exotic plant species at five parks in Kitchener-Waterloo, and to use these data to test the feasibility of using landscape attributes as predictors of exotic plant species diversities.. Soil texture was also tested to determine whether it was a useful predictor of exotic plant species presence. Results from my study (based on the five parks sampled) showed that soil and spatial attributes were not statistically significant predictors of exotic plant species diversities and abundances. However, of the spatial attributes tested, smaller park size and higher edge: interior ratio were the strongest predictors of increased numbers and diversities of exotic plant species. Results also showed that native species outnumber exotic plant species at core and edge sites at all parks except for Doon, which may be due to decreased soil pH at the Doon site. Longer term and large scale studies of natural areas (of varying shapes and sizes) should be conducted before spatial attributes as predictors of invasion can be ruled out and I believe that more significant results may emerge with a larger sample size which includes more parks of smaller areas. Additionally, soil fertility should be tested to determine its usefulness as a predictor of invasive plant species presence.

This research does raise concern over the use of species richness as a positive ecological indicator which is common among ecological studies. My results showed that Doon had total herbaceous species diversity counts equal to two other parks but more than double the amount of exotic herbaceous species (Table 4.2). Doon also had the highest total tree and shrub species diversity but had more than double the amount of exotic tree/shrub species as three of the other parks (Table 4.3). A recent study by Christian et al. (2009) cautions that increases in species richness should not always be interpreted as conservation improvement as they could be a result of a transition from species specialists to generalists, or in the case of my study, an increase in exotic species.

This study was conceptualized and developed on a number of assumptions. The first is that native biodiversity in urban natural areas is valuable and worthy of conservation efforts. The second is that exotic plant species have the potential to become invasive by disrupting ecosystem function or outcompeting native plant species. The third is that the five parks chosen, Breithaupt Park, Doon, Homer Watson Park, Huron Natural Area and Forested Hills ESPA were all representative parks of the cities of Kitchener and Waterloo, and the associated management and monitoring techniques of these cities.

6.2 Management Implications and Future Directions

The City of Kitchener is currently developing a Parks Master Plan and invasive plant species prevention should be a priority. Kitchener and Waterloo need an increased commitment to invasive species management with a significant focus on monitoring, which is in my opinion, an essential part of prevention. Limited financial resources will surely constrain effective invasive species management but there are ways around this, aside from increased funding. Citizen Scientist (volunteers who participate as field assistants in ecological studies) movements are growing and as citizens recognize the importance of ecological conservation, they seek to become more educated and involved. Citizen Scientist groups are not new (the Audubon Society has relied on volunteers for its Christmas bird count since the 1900's) but with financial restraints, more municipalities and research groups are relying on these volunteers for the collection of scientific data (Cohn, 2008). The reliability of the data is debated by opponents but once a group of individuals is trained (in this case, familiar with botany identification or equipped with good field guides and led by an expert botanist), there is no reason why they can't successfully carry out vegetation surveys.

Yearly monitoring of natural areas in the region should be implemented, and those natural areas that are smaller in size or with a larger edge: interior ratio should be prioritized. Lake & Leishman (2004) also suggest that monitoring be focused on non-invasive exotics with higher specific leaf areas because they may be able to outcompete native species in nutrient-enriched sites. Planting new native species with high specific leaf area may also be a successful tactic in outcompeting invasive exotic species. Monitoring can be carried out by volunteers and the data should be stored in an accessible database for reference in future years. Of course, citizen scientist groups are only effective if the public is aware and educated on the need for monitoring natural areas for the establishment of invasive plant species and the presence of new exotic plant species.

Based on the results of this study, I believe that natural areas within the cities of Kitchener and Waterloo are well-connected and are currently under good management. However, connectivity should remain a focal point of importance in terms of research, management, and development. Corridors and buffers should be protected and their establishment (using only native plant species) should be encouraged. Both cities recognize the significance of the protection of natural areas and their connectivity. The Region of Waterloo has designated over 25,000 acres of protected land, and there are 80 Environmentally Sensitive Policy Areas where activities and development are restricted (Region of Waterloo, 2009b). The City of Kitchener's Municipal Plan also encourages the protection of significant stands of trees and hedgerows in the city and Kitchener's Tree Management Strategy was recognized in 2003 by the Canadian Society of Landscape Architects "for its foresight and its role in fostering environmental stewardship" (City of Kitchener, 2009b). A multi-regional study, comparing the connectivity of natural areas (independently or in combination with exotic plant species abundances) in other regions would be interesting. Although the cost and time of such an undertaking would be grand, the results may yield beneficial comparisons.

Invasive plant species that have effectively established and spread should still be controlled locally using whichever techniques are most effective for the species in question (mulching, hand removal, herbicide treatments) but large-scale infestations may require different eradication techniques (see McCartney (2005) for a review of control techniques for *Alliaria petiolata* and *Vinca minor).* Fragmentation of current natural areas and urbanization in close proximity to natural areas should also still be minimized or avoided to decrease the risk of new invasions and establishment and exotic plant species

APPENDIX A
PLOT LOCATIONS

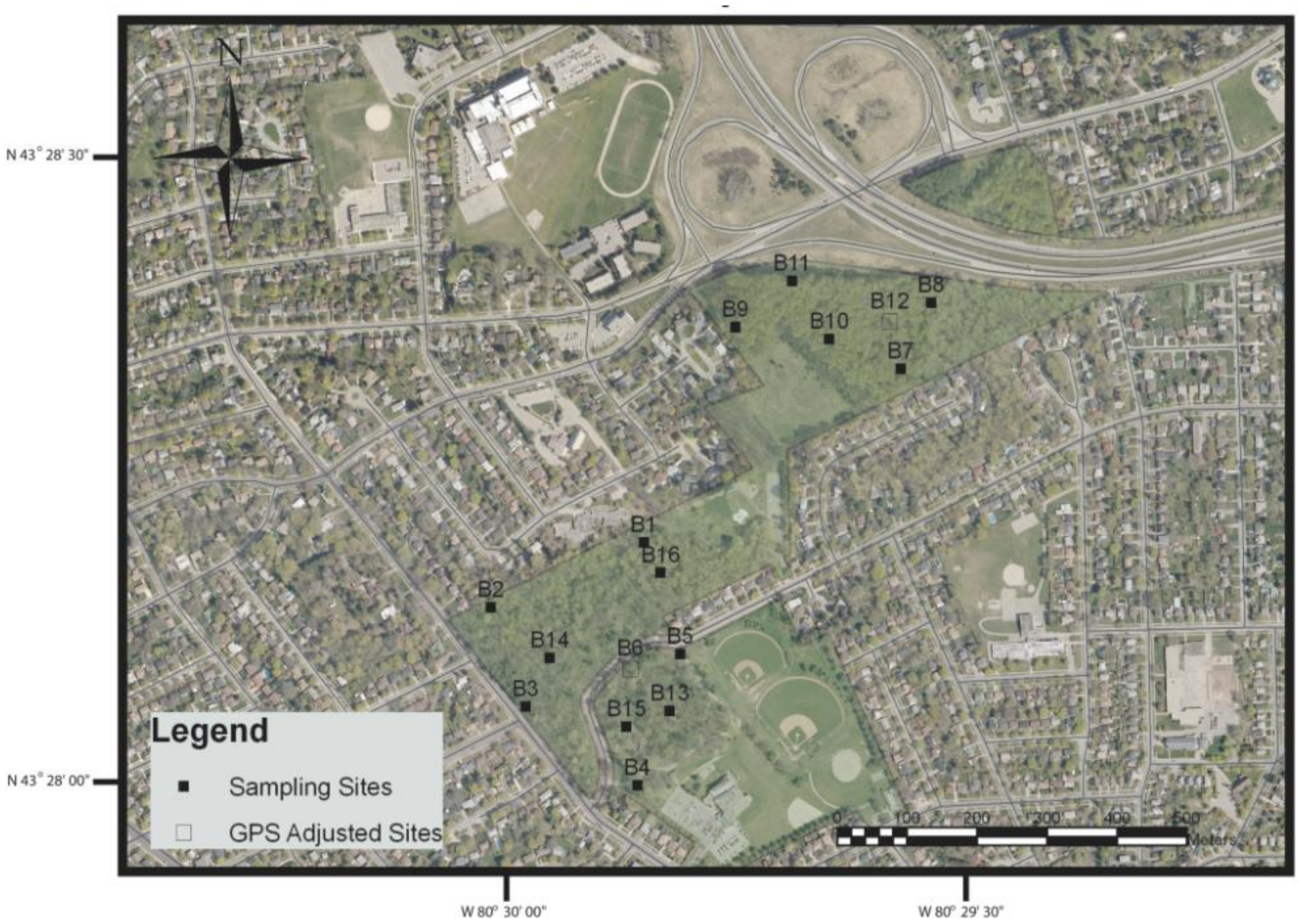

Figure A.1 Plot locations at Breithaupt Park

Figure A.2 Plot locations at Doon Village

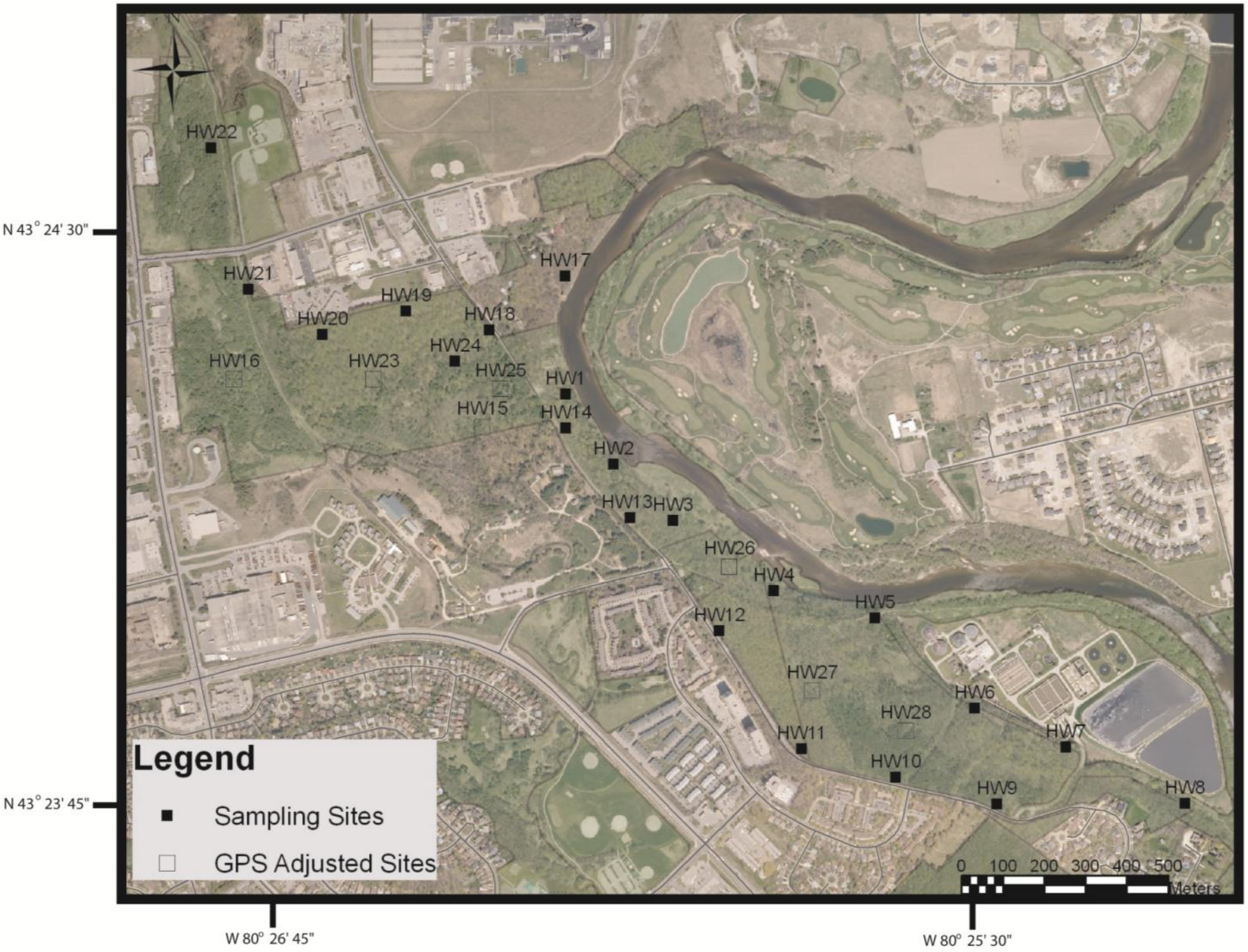

Figure A.3. Plot locations at Homer Watson Park

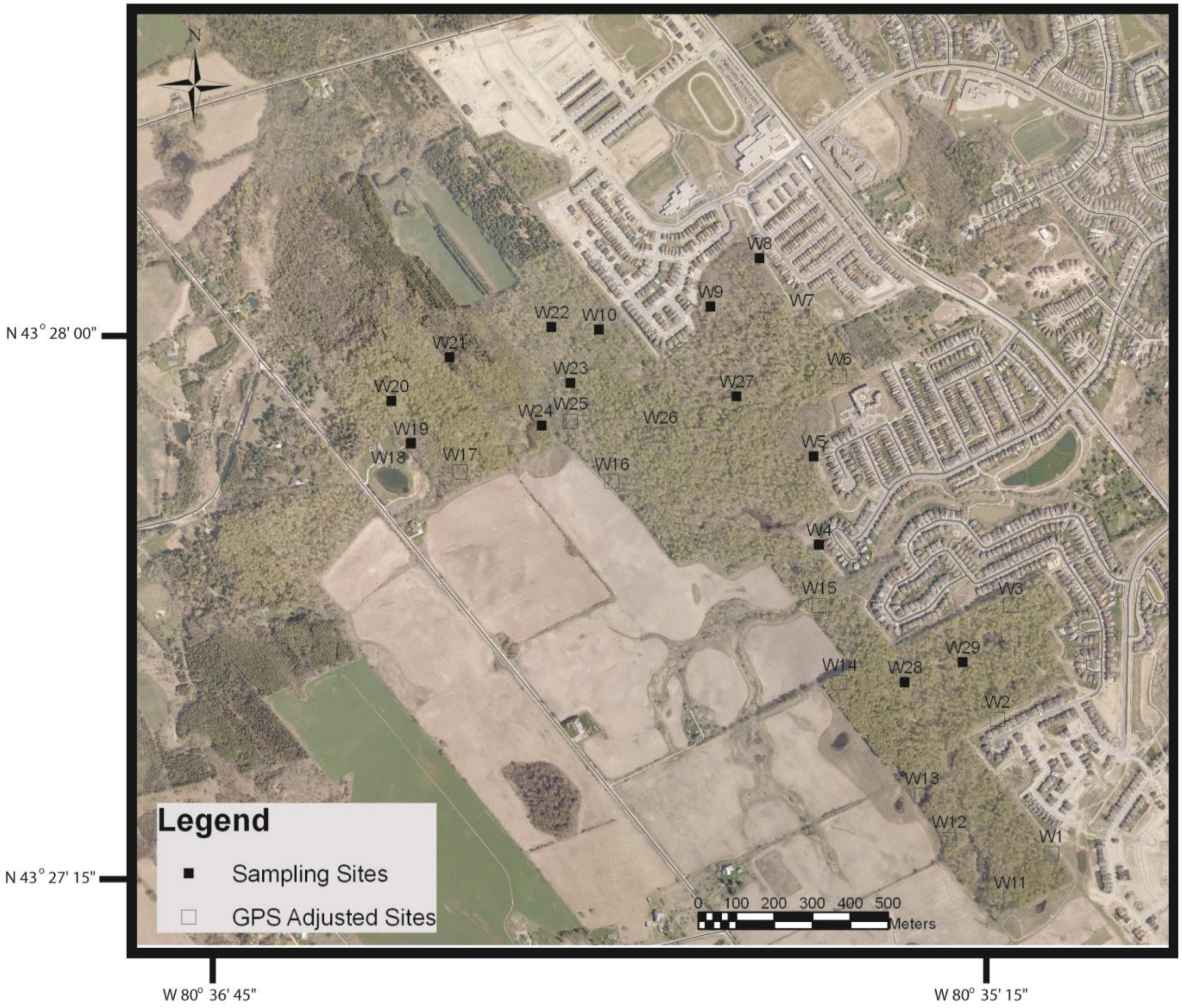

Figure A. 4 Plot locations at Forested Hills ESPA

Figure A. 5. Plot locations at Huron Natural Area

APPENDIX B
SPECIES LIST

Table B.1. Herbaceous species list

Scientific Name	**Common Name**	**Native/Exotic/Weed**	**Edge Sites**	**Core Sites**
Abutilon Mill.	Indian Mallow	E/W	HW, F	-
Actaea sp. L. *(Actaea rubra, Actae pachypoda)*	Baneberry	N	B	-
Alliaria petiolata (M.Bieb.) Cavara & Grande	Garlic Mustard	E	All	All
Allium tricoccum Ait.	Wild Leek	N	HW, F	HW, F
Ambrosia artemisiifolia L.	Common Ragweed	N/W	HW	-
Arisaema triphyllum L.	Jack-in-the-pulpit	N	D, HW, HN, F	B, HW, HN, F
Asarum canadense L.	Wild Ginger	N	B, HW, HN, F	HN, F
Boehmeria cylindrical L.	False Nettle	N	F	B
Cardamine diphylla	Two-leaved Toothwort	N	HW	-
Caulophyllum thalictroides (L.) Michx.	Blue Cohosh	N	B, HW, F	HW, HN, F
Chelidonium majus L.	Celandine	E	F	D
Chenopodium album L.	Lamb’s-quarters	E/W	-	D
Circaea lutetiana L.	Enchanter’s Nightshade	N	All	All
Convallaria majalis L.	Lily of the Valley	E	D	-
Dicentra eximia (Ker Gawl.) Torr.	Wild Bleeding Heart	N	F	HN, F
Duchesnea indica (Andrews) Focke	Indian Strawberry	E	B	-
[a]*Echinochloa crusgalli* (L.) P. Beavuv.	Barnyard Grass	E/W	HW, HN, F	-
Elodea nuttallii (Planch.) H. St. John	Water Thyme	N	HW	-
Equisetum arvense L.	Common Horsetail	N/W	D, HN	-
Equisetum sp. L.	Equisetum sp.	N/W	HW	-
Erythronium americanum Ker Gawl.	Trout Lily	N	All	All
Euonymus fortunei (Turcz.) Hand.-Maz.	Wintercreeper	E	D	-
Euphorbia cyparissias L.	Cypress Spurge	E/W	D	-
Fragaria virginiana Duchesne	Wild Strawberry	N	D, F	B, HW
Galium mollugo L.	White Bedstraw	E	D, HN	-
Geranium maculatum L.	Wild Geranium	N	D	-
Geranium robertianum L.	Herb Robert	E	HW, HN, F	D, HW, HN, F
Geum rivale L.	Water Avens	N	HN, F	HW, F
Glechoma hederacea L.	Creeping Charlie	E	B	-
[a]Grass sp.	Grass sp.	*	All	All
Helianthus divaricatus L.	Woodland Sunflower	N	HW	D

Hepatica acutiloba DC.	Sharp Lobed Hepatica	N	B	-
Hydrophyllum virginianum L.	Virginia Waterleaf	N	D, HW, HN, F	F
Ilex verticillata (L.) A. Gray	Winterberry	N	HW	D
Impatiens capensis Meerb.	Jewelweed	N	HW, HN, F	HW, HN, F
Lysimachia nummularia L.	Money Wort	E	D	-
Maianthemum canadense Desf.	Canada Mayflower	N	B, HW, HN	B, HW, F
Nepeta cataria L.	Catnip	E	-	D
Plantago major L.	Common Plantain	E/W	-	D
Podophyllum peltatum L.	Mayapple	N	-	B
Polygonum hydropiper L.	Common Smartweed sp.	E/W	B, D, HW, HN	D
Polypodium vulgare L.	Common Polypody	N	HW	-
Prenanthes altissima L.	Tall White Lettuce	N	HW	-
Ranunculus sceleratus L.	Cursed Crowfoot	N	F	F
Sanguinaria Canadensis L.	Blood Root	N	B, HW	HW
Smilacina racemosa (L.) Desf.	False Solomon’s Seal	N	D, HW, HN, F	B, HW, HN, F
Stylophorum diphyllum (Michx.) Nutt.	Wood Poppy	N	B	B
Taraxacum officinale F.H. Wigg.	Common Dandelion	E/W	D, HW	D, HW
Thalictrum sp. L.	Meadow Rue sp.	N	B, HW	B, HW
Trillium sp. L. (*Trillium grandiflorum* (Michx.) Salisb.; *Trillium lancifolium* Raf.)	Trillium	N	B, HW, HN, F	B, HN, F
Tussilago farfara L.	Coltsfoot	E	D	-
Uvularia grandiflora Sm.	Large flowered Bellwort	N	-	HW
Vinca minor L.	Periwinkle	E	B, HW, F	B
Viola blanda Willd.	Sweet White Violet	N	-	HN
Viola sororia Willd.	Common Blue Violet	N	HW	-
Violet sp.	Violet sp.	N	D, HN	HN
Waldsteinia fragarioides (Michx.) Tratt.	Barren Strawberry	N	-	B

*B = Breithaupt, D = Doon, HW = Homer Watson, HN = Huron Natural Area, F = Forested Hills ESPA

[a] Grasses were counted in the survey but not identified and not included in analysis.

Table B.2. Tree/Shrub species list

Scientific Name	**Common Name**	**Native/Exotic**	**Edge Sites**	**Core sites**
Acer platanoides L.	Norway Maple	E	B,D,HW	-
Acer rubrum L.	Red Maple	N	All	All
Acer saccharum Marsh.	Sugar Maple	N	HN	HW
Aesculus hipposcastanum L.	Horsechestnut	E	D	B
Berberis sp. (*Berberis thunbergii* DC; *Berberis vulgaris* L.)	Barberry sp.	E	D	-
Betula alleghaniensis Britton	Yellow Birch	N	HW	HW, HN, F
Castanea dentate (Marsh.) Borkh.	American Chestnut	N	-	D, HW
Clematis occidentalis (Hornem.) DC.	Purple Clematis	N	HW	-
Cornus alternifolia L. f.	Alternate-Leaf Dogwood	N	All	All
Fagus grandifolia Ehrh.	American Beech	N	B,HW,HN,F	B, HW, HN, F
Fraxinus americana L.	White Ash	N	D, HW	D, HW
Fraxinus nigra Marsh.	Black Ash	N	All	All
Gaultheria procumbens L.	Wintergreen	N	HW	-
Hamamelis virginiana L.	American Witch-Hazel	N	HW	D
Horticultural prunus species	Horticultural species	E	-	F
Larix laricina (Du Roi) K. Koch	Tamarack	N	D	
Lonicera periclymenum L.	European Honeysuckle	E	D, HN	B, D
Lonicera dioica L.	Limber Honeysuckle	N	B, D	B
Lonicera tatarica L.	Tartarian Honeysuckle	E	D, HW, HN	D
Malus coronaria (L.) Mill.	Wild Crabapple	N	B,D, HW	B, D, HW
Ostrya virginiana (Mill.) K. Koch	Hop-Hornbeam	N	All	All
Parthenocissus quinquefolia (L.) Planch.	Virginia Creeeper	N/W	HW	D, HW
Physocarpus capitatus (Pursh) Kuntze	Ninebark	N	-	D
Picea abies (L.) Karst.	Norway Spruce	E	-	F
Picea glauca (Moench) Voss	White Spruce	N	D, HW	D
Picea mariana (Mill.) Britton, Sterns & Poggenb.	Black Spruce	N	D	D, F
Pinus resinosa Ait.	Red Pine	N	D, HW	D
Pinus strobus L.	Eastern White Pine	N	D, HW	D
Populus deltoides Bartram ex Marsh.	Eastern Cottonwood	N	B	D
Populus tremuloides Michx.	Trembling Aspen	N	HN, F	F
Prunus pensylvanica L.f.	Pin Cherry	N	B, HN, F	D, F
Prunus serotina Ehrh.	Black Cherry	N	All	B,D, HW
Prunus virginiana L.	Choke Cherry	N/W	All	All

Quercus alba L.	White Oak	N	HW	-
Quercus muehlenbergii Engelm.	Chinquapin Oak	N	HN	-
Quercus rubra L.	Red Oak	N	B, HW, F	B, D
Rhamnus cathartica L.	Common Buckthorn	E/W	All	B, D, HW, F
Rhus glabra L.	Smooth Sumac	N	HW, HN	-
Rhus typhina L.	Staghorn Sumac	N/W	HN	-
Ribes cynosbati L.	Prickly Gooseberry	N	B, HW, F	B, HW, F
Ribes hirtellum Michx.	Wild Gooseberry	N	D, HW, F	D, HW, F
Rubus ostryifolius Rydb.	Highbush Blackberry	N	B	-
Sambucus nigra L.	Common Elder	N	-	D
Sambucus sp.	Elderberry sp.	N	All	All
Saxifragaceae family	Currant Family	N	D, HW, F	D, HW, F
Solanum dulcamara L.	Climbing/Bittersweet Nightshade	E/W	D	B, HW, HN
Sorboria sorbifolia L.	False Spirea	E	D	HW
Symphoricarpos albus (L.) S.F. Blake	Common Snowberry	N	D	B
Syringa vulgaris L.	Common Lilac	E	-	D
Thuja occidentalis L.	Eastern White Cedar	N	HW	HW
Tilia americana L.	American Basswood	N	All	B, HW
Toxicodendron radicans (L.) Kuntze	Poison Ivy	N/W	HW	HW
Tsuga canadensis (L.) Carriere	Eastern Hemlock	N	HW, HN	D, HW, HN
Vitis sp. L.	Wild Grape	N/W	B, D, HW, F	HW

*B = Breithaupt, D = Doon, HW = Homer Watson, HN = Huron Natural Area, F = Forested Hills ESPA

Native/exotic status designated by Brouillet, et al. (2006) and the Integrated Taxonomic Information System (IT IS) of the Canadian Biodiversity Information Facility. Weed status designated by the Ontario Ministry of Agriculture, Food & Rural Affairs Publication 505: Ontario Weeds

APPENDIX C
SOIL SURVEY RESULTS

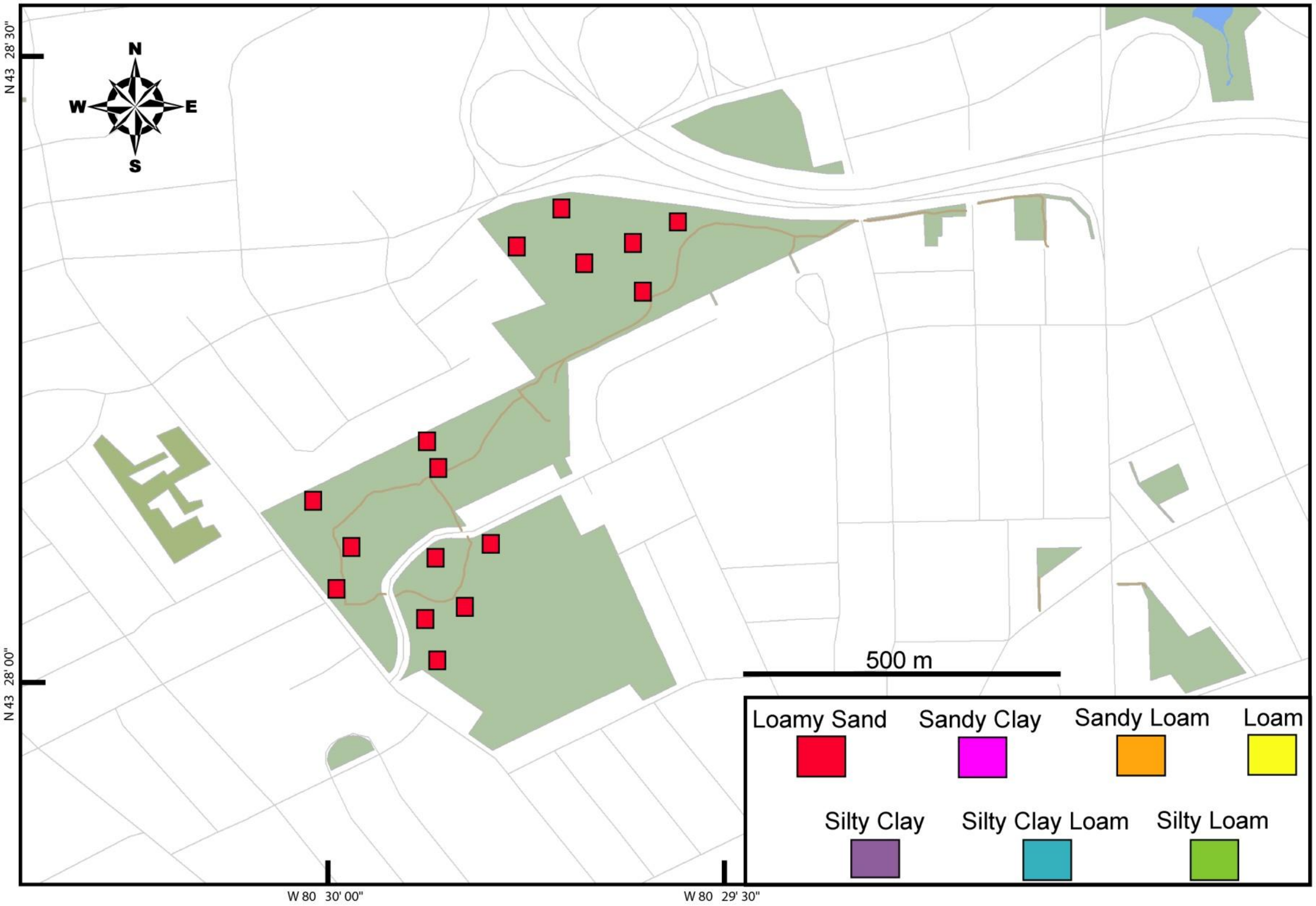

Figure C.1 Soil texture at plot locations at Breithaupt Park

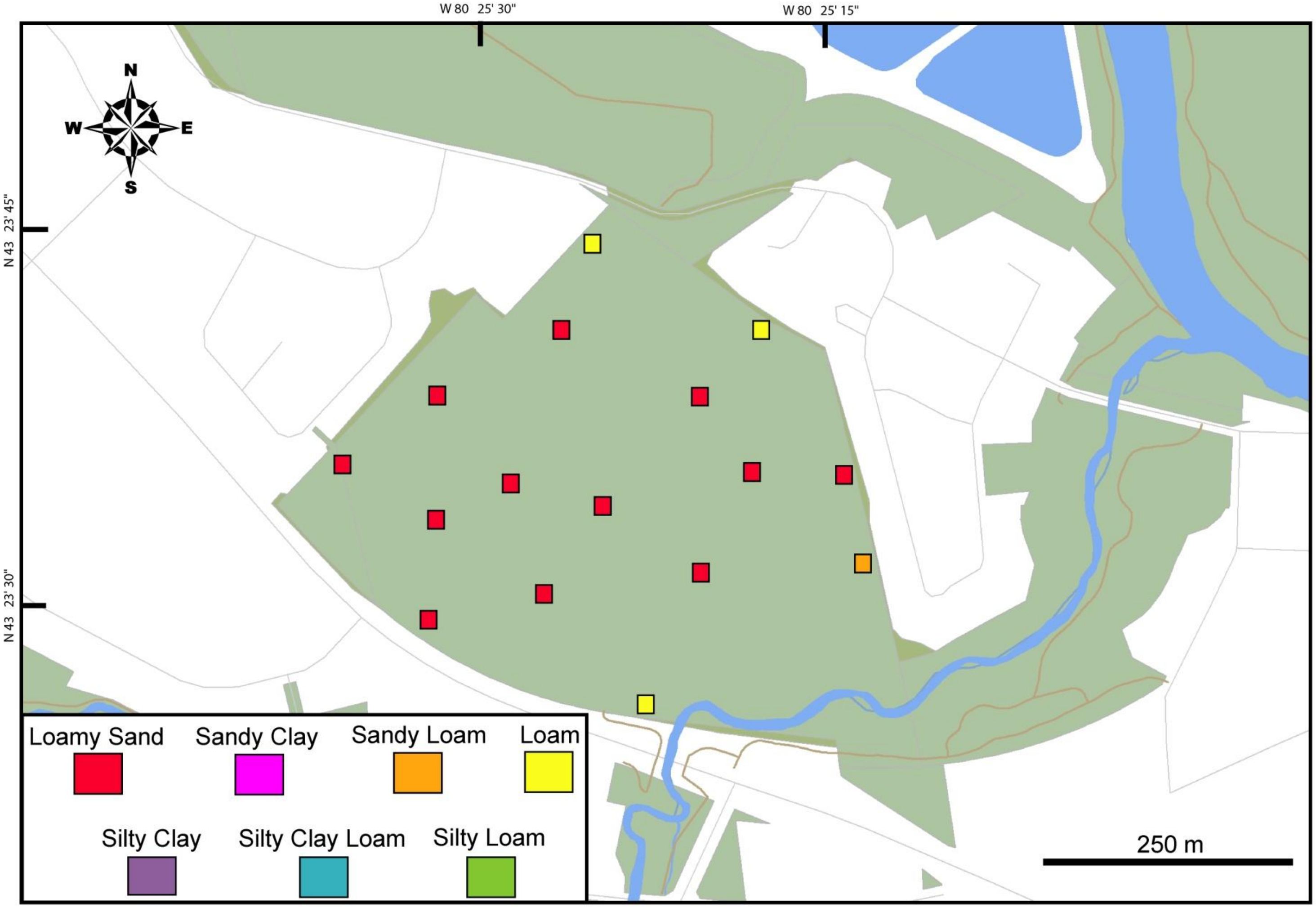

Figure C.2 Soil texture at plot locations at Doon Village

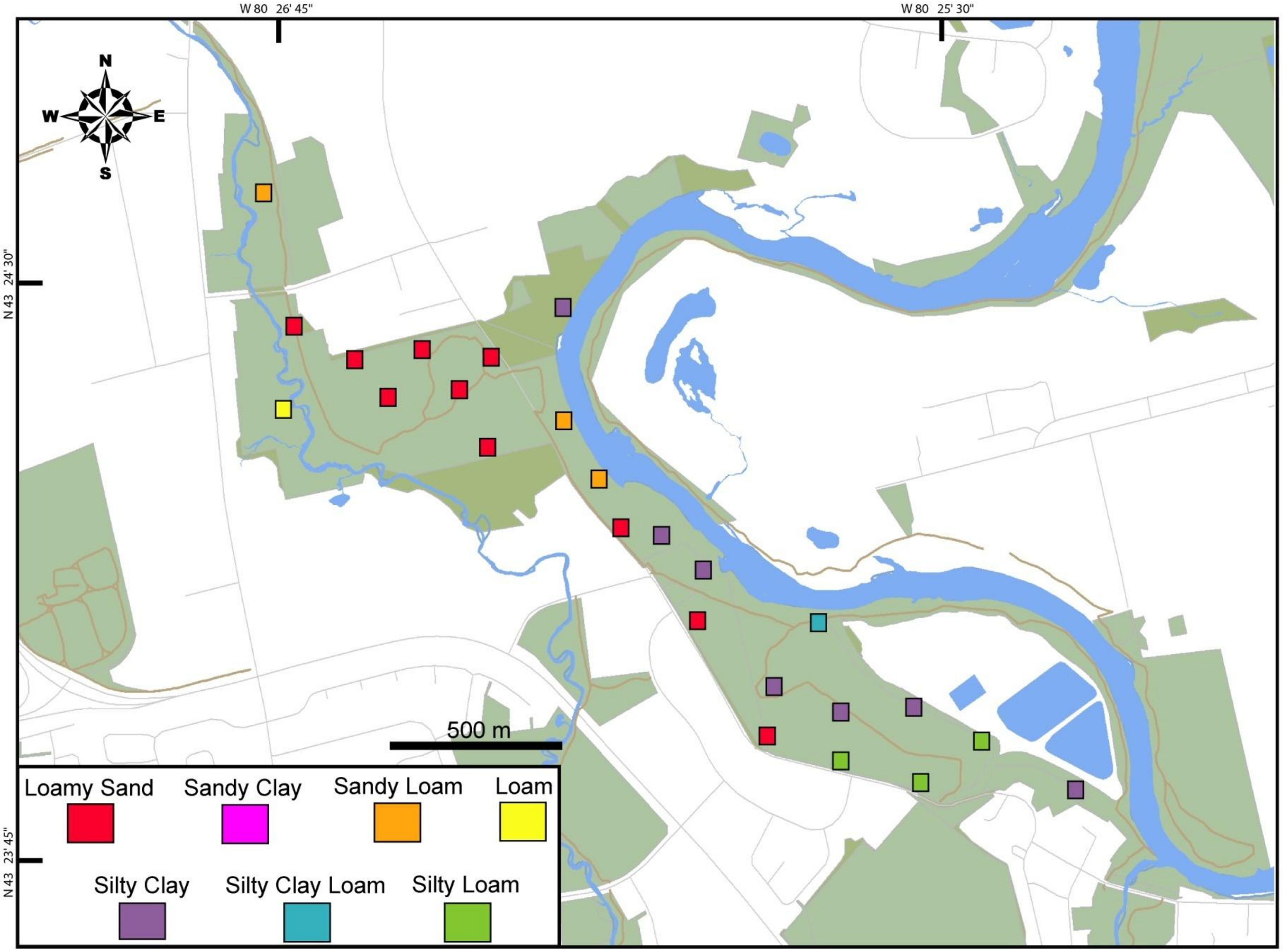

Figure C.3 Soil texture at plot locations at Homer Watson Park

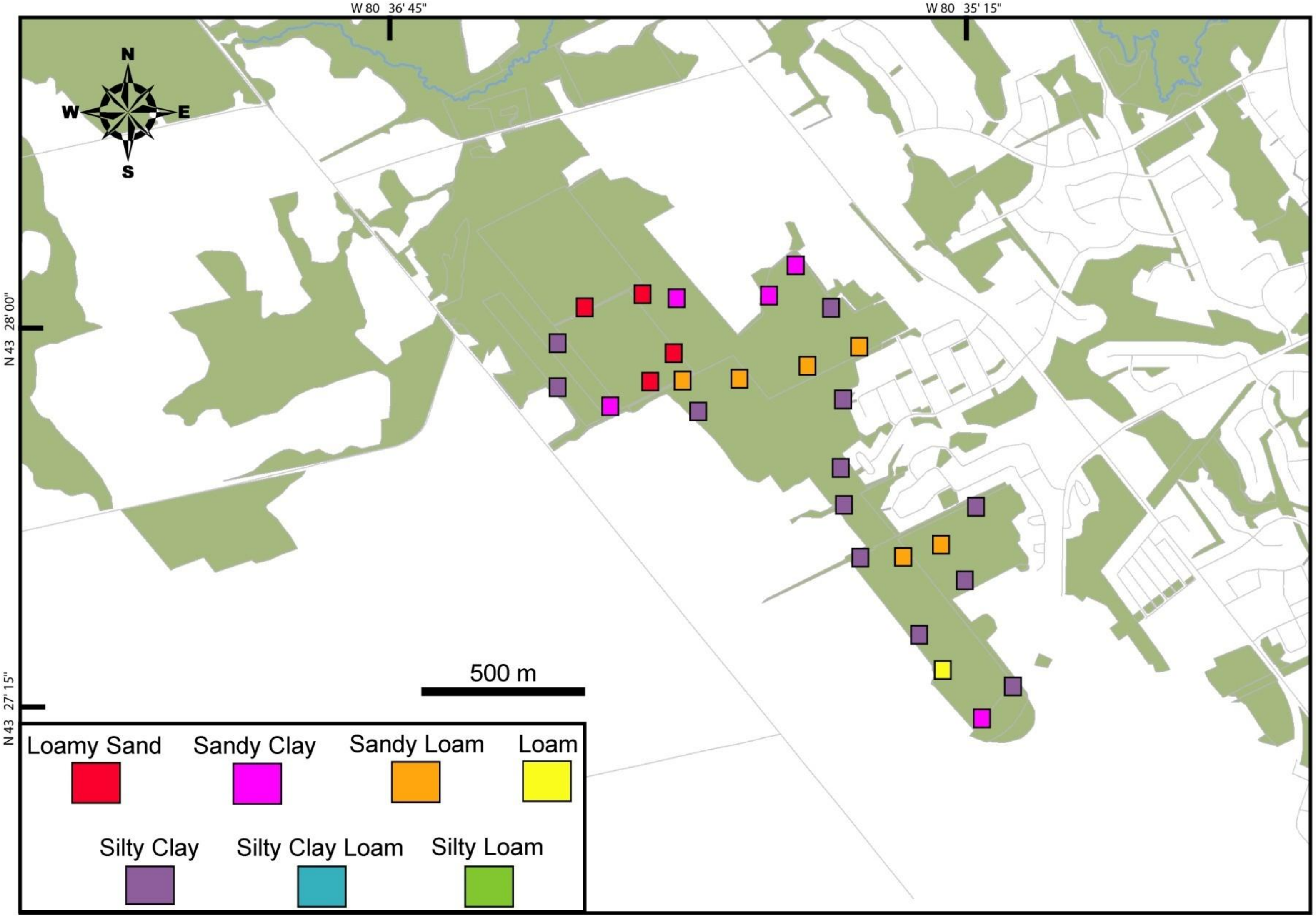

Figure C.4 Soil texture at plot locations at Forested Hills ESPA

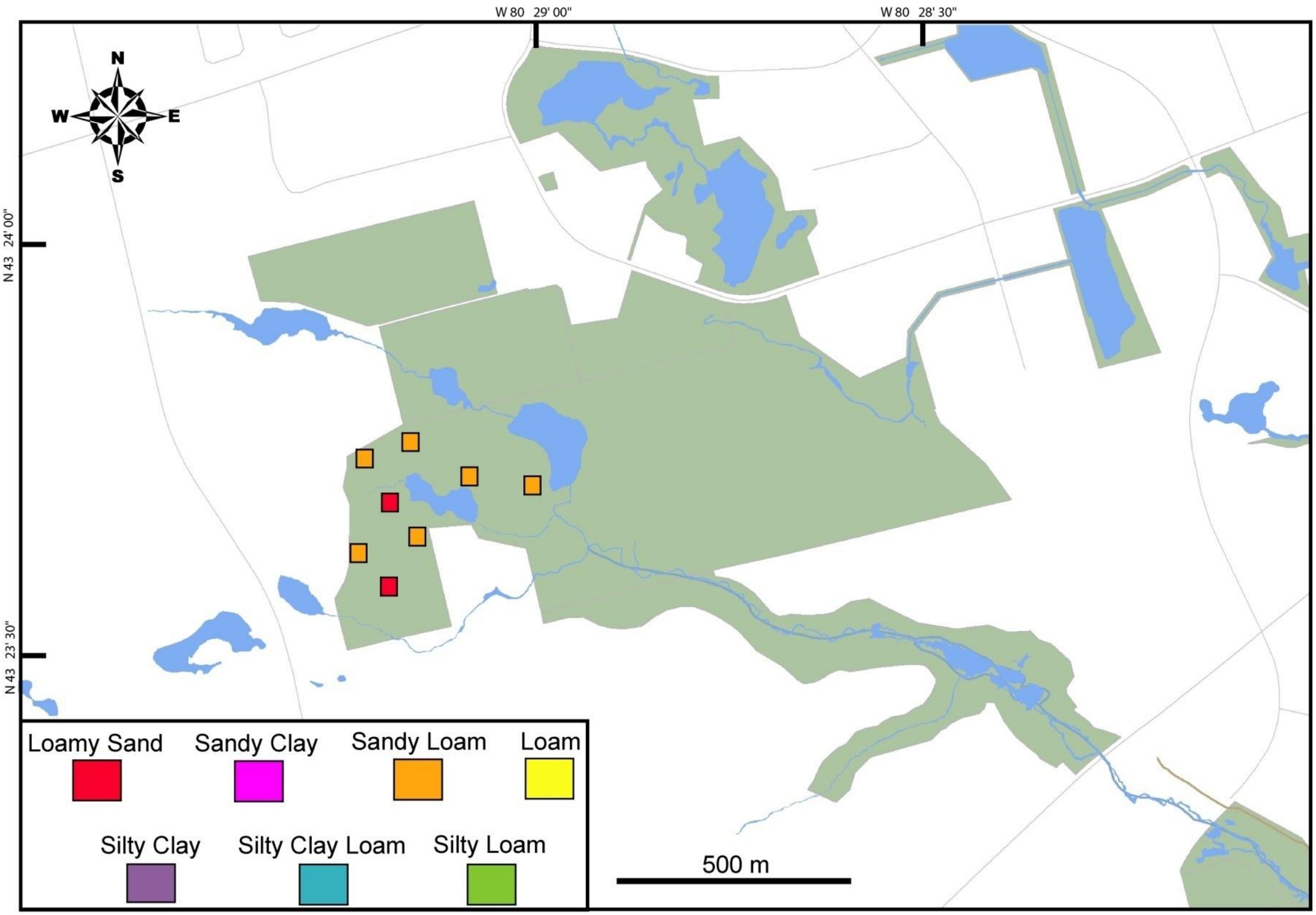

Figure C.5 Soil texture at plot locations at Huron Natural Area

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