The Biogeography of Peel’s Urban Forest: Patterns and Correlates of Species Diversity

by

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A thesis submitted in conformity with the requirement for the degree of Master of Science

Department of Geography
University of Toronto

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Abstract

The purpose of this research project is to identify the species-level diversity and distribution of trees within the urban setting of Peel Region (Ontario, Canada) and to determine how these characteristics change as a function of land use type. To address this, alpha diversity (species richness within a community), evenness, and beta diversity (species richness between communities) were calculated for eight distinct land use types within the study area. As well, the influence that a variety of socioeconomic and urban form variables have in determining urban forest composition was examined using regression techniques. Results indicate that significant relationships exist between land use type, species richness and overall tree abundance. Variables reflecting wealth and urban form are also shown to significantly influence tree abundance. The results of this study address issues pertaining to the adaptation, conservation, and management of the region’s urban tree species.
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Chapter One
Introduction

The ecological and social contributions that urban forests make to city environments are vital. Trees in municipal settings improve air quality, sequester carbon, affect storm water runoff, moderate temperature increases, and provide habitats for insects, birds, and mammals (Alberti 2005). Trees also reduce energy costs associated with heating and cooling, and provide numerous spiritual, recreational and restorative opportunities for city residents who may otherwise have no exposure to natural settings (Coley et al. 1997). Unfortunately, urban forests are coming under increasing threat. Factors such as climate change, invasive species, soil acidification, habitat fragmentation and urban densification are growing stressors for the urban forest (Alvey 2006).

Over the past four decades, the discipline of urban ecology has emerged specifically to address these issues. In the beginning, urban ecologists’ focus was largely restricted to the conditions of humans within urban environments and the environment itself was almost entirely neglected. However, a body of literature has steadily developed that attends to the issues that were ignored in the early and mid twentieth century. Modern urban ecologists now work to identify and explain the natural and anthropogenic processes and interactions that occur within urban environments.

Within the discipline, one prevalent stream of research focuses on urban vegetation studies. Different strategies have been employed to investigate the patterns that vegetation takes within cities; this work has included examining total vegetation abundance across a city, often via aerial photography and satellite imagery (Dweyer and Miller 1999; Iverson and Cook 2000; Heyen and Linsdey 2003; Grove et al. 2006; Conway and Urbani 2007), as well as conducting ground based counts of sample plots to yield detailed accounts of the different types of vegetation that exist (Sudha and Ravindranath 2000; Jim and Liu 2001; Zipperer
2002; Martin et al. 2004; Lundholm et al. 2006; Weifeng et al. 2006; Godefroid and Koedam 2007; Jim and Chen 2009; Luck et al. 2009; Walker et al. 2009; Zhao et al. 2010). As a result of this work, much is now understood about how city environments differ from natural ecosystem. For example, within urban tree ecology, it has been largely found that urban areas harbor amplified levels of species richness in comparison with their surrounding landscape (Wania et al. 2006; McKinney 2008). The number of invasive species is also often found to rise in tandem with urbanization (Turner et al. 2005; Alvey 2006).

In order to fully integrate ecological trends into the fabric of the city environment, researchers have recently begun to consider the relationship between vegetation and human influence. Though this stream of research has only begun to develop in the last decade, strong correlations have already been observed between patterns of human distribution, social stratification and plant community structure (Iverson and Cook 2000; Hope et al. 2003; Alberti 2003). It is therefore important that this avenue of urban ecological study be pursued.

Despite the insight into ecological processes that has been gained from the development of urban ecology, several areas have received less attention than others and gaps in the literature remain. Significantly, studies that consider urban trees at the species-level are lacking. Most studies examine the total canopy cover of an urban environment or broad vegetation classes (Weifeng et al. 2006; Zhao et al. 2010) and do not look at the individual species that compose the biomass. Identifying which tree species grow where and with what other species is an important component of understanding the urban ecosystem.

Another of the gaps that exists in the literature is at the scale of intra-city land use types. A large portion of studies that look at the relationships between land use and vegetation treat "urban areas" as a single unit. Alternatively, other studies focus exclusively on one very narrow land use within the larger city. The importance of connecting these two extremes was illustrated by Godefroid and Koedam (2006) in their study of the effects of six different urban land use classes on species
richness in Brussels. Their findings indicated that subtle differences in climate and surface cover between various urban land use types have notable consequences for species richness. As well, they discussed the increasingly important need for studies to combine land use (at a much finer grain that has previously been used) with socioeconomic data to advance current understandings of urban forests drivers.

In order to attend to these deficits, this project examines species-level data across different urban land uses to address two main objectives. The first is to determine alpha and beta diversity of tree abundance and species richness for the distinct urban land use types in Peel Region using plot level data collected following the UFORE Model protocol. The second objective is to quantify how tree abundance and species richness are related to specific urban form and socioeconomic variables.

Peel Region was chosen as the study area for this project because it is currently undergoing considerable urban development, a trend that is projected to continue into the future. For this reason, it is essential that the existing level of diversity within Peel’s urban forest be documented so that ecological changes that result from future urbanization may be monitored and quantified. As well, the four municipalities within Peel that this study focuses on embody different levels of urbanization and therefore present an opportunity to compare how each is impacted. The rapid speed and largely suburban form that development has taken in Peel Region is representative of much of the sprawling urban development occurring across North America. Therefore, findings obtained in this study can reasonably be used as approximations for the responses of other expanding municipalities under similar conditions.

This study provides insight into the relationship between land use and urban forest diversity for the Region of Peel. The results illuminate the impact that land use designation, urban form characteristics, and social factors have on Peel’s tree species richness and abundance. As well, the results can aid forest management decisions within Peel, and they provide an important baseline study for all future work in the region. This project contributes to a growing literature on urban ecology
by specifically addressing species-level analysis and social drivers in urban ecosystems.

This thesis is organized into six chapters. Following the introduction, a chapter addresses recent developments in urban ecology. The next three chapters present the research study area and address the two main objectives. The last chapter presents concluding remarks.
Chapter Two
Background Literature

2.1 Introduction

This chapter examines the urban ecology literature more closely; it discusses the development of the discipline, and the distinct methodologies and themes that are currently being explored within the field. As well, gaps within the literature are identified.

Cities are unique landscapes in many ways, notably they require enormous energy input, produce substantial amounts of waste, and the perturbations that occur to their biotic, microclimate, hydrological, and soil systems are often highly augmented (Alberti 2005). The specific properties of cities also create spatial signatures that are unique to urban environments, for example, high levels of distinctiveness between habitat patches and high numbers of alien species are often observed (Alberti 2005). The accelerating growth of urban environments worldwide has been well documented within the literature (McDonnell and Pickett 1990; Folke et al. 1997; Bolund and Hunhammar 1999; Grimm et al. 2000; McIntyre et al. 2001; Pickett et al. 2001; Nowak et al. 2002; Turner et al. 2005; Alberti 2008; Conway 2009). As urban areas have developed and expanded, the discipline of urban ecology has too. It is expected that as urban landscapes continue their exponential growth - in terms of the physical space they occupy, the amount of resources they consume and the degree of influence they have over surrounding areas - the field of urban ecology will continue to develop in tandem. The discipline is therefore an exciting frontier and an important area of research as no other environment can act as a proxy for how city ecosystems operate.

The following sections detail the history of urban ecology and discuss recent foci within the field.
2.2 Historical Development of Urban Ecology

Ecology, as a discipline, was established in the nineteenth century (Sukopp 1990). Originally, its practice was heavily influenced by natural history; discovery, description and categorization were the main objectives, with ecologists endeavoring to collect facts, rather than trying to observe conditions (Trepl 1990). Early ecology was about identifying where biological organisms lived and how, not why (Sukopp 1990).

Throughout the twentieth century, the study of ecology expanded and the landscapes of interest diversified. Rapid urban development at the beginning of the century in North America, as well as urban destruction during the Second World War in Europe prompted ecologists on both continents to focus their attention toward the response of vegetation to changes in city environments (Mucina 1990). However, urban ecology, as it was studied for most of the twentieth century, took on distinctly different foci in Europe and North America, with little collaboration between the two.

Within North America, the ecology of urban environments was focused largely on the humans who functioned within cities (Marzluff et al. 2008). In the early 20th century, social and biological scientists at the University of Chicago contributed to the production of some of the first works of North American urban ecology, with the discipline that emerged based primarily in the social sciences (Pickett and Cadenasso 2006). Specifically, the Chicago school of thought attempted to describe human functioning within a city through ecological mechanisms and metaphors. However, this early North American urban ecology was focused solely on human patterns rather than on examining the biophysical features in urban settings. Over time this social model failed to adapt to changes that occurred within ecological theory; the principals on which the Chicago School was built, such as deterministic succession, have since been abandoned in ecology (Pickett et al. 2001). The school of thought also quickly fell out of favour with social scientists (Picket and Cadenasso 2006). However, a lasting effect of the Chicago School and
other related work by North American social scientists was that the city was the
cervue of the social sciences, not natural or physical scientists.

At the same time in Europe, ecological studies of urban environments were
also occurring, but with a heavier focus on the physical processes and natural
environments that exist amidst human activities. European urban ecologists
produced detailed surveys of plants and animals, while characterizing their
surrounding urban landscape (Sukopp 1990). Over time, this branch of ecology
developed a close working relationship with urban planning in Europe (Pickett and
Cadenasso 2006). The social component, however, was largely ignored (Marzluff et
al. 2008).

It was not until the 1970’s that urban ecology began to develop as a
discipline in itself (Pickett et al. 2001). This can be attributed to two factors: (1) a
growing recognition by North American natural scientists that ecological features in
cities were important foci, and (2) a shift by ecologists towards systems theory
(Pickett and Cadenasso 2006). Whereas earlier, components of an environment had
been viewed individually, such that ecologists interested in plants looked only at
plants, by the 1970’s, ecologists were beginning to see those plants as part of the
larger ecosystem (Marzluff 2008). Subsequently, it began to be acknowledged that
elements needed to be studied and managed as part of their larger systems.
Ecologists were finally addressing the questions about why the changes they were
observing were occurring that had been ignored decades earlier.

One aspect of system theory that has been particularly influential on urban
ecology is patch dynamics. The theory of patch dynamics emerged in the late 1970s.
It considers the urban landscape as a heterogeneous mosaic, examining the flow of
energy and materials across those mosaic components (Pickett and Cadenasso
2006). A theory like patch dynamics, that stressed multiple and complex
connections within landscapes, was an important progression in the field of urban
ecology for several reasons. For one, it helped urban ecologists communicate the
heterogeneity they were finding in urban areas. It also provided a common theory to
capture the spatial distribution of both biotic and physical structures within the urban environment. In addition, the theory was equally applicable to urban and natural ecosystems and therefore helped to diminish the conceptual divide between the two (Pickett and Cadenasso 2006).

The influence of ecosystems theory continued to grow over the course of the 1980s and 1990s. In 2000, Grimm et al. (2000) identified two distinct approaches to urban ecology (ecology in cities and ecology of cities) as a result of the growing influence of system theory. Grimm et al. argue that up until the beginning of the twenty-first century, only the first of these approaches had been fully explored. Ecology in cities consists of comparing cities to other types of environments in order to address the question of how a city influences the components inside it. Each individual study, therefore, examines only a single component, such as: the occurrence of Culex mosquitoes across urban and rural sites (Reisen and Pfuntner 1987), the meteorological and quality conditions within a city (Yamashita 1974), the consequences of edge effects and habitat fragmentation on nest predation in urban environments (Matthews et al. 1999), or the interaction between native and introduced species (Fox 1990). These types of studies are refined in their focus and are important in documenting basic characteristics of urban environments, but they fail to consider the city as a cohesive unit. Pickett et al. (2001) reviewed this model and expressed that studies of ecology in cities form the base on which other urban ecology studies are built. Therefore, the significant amount of attention that they received early on was a necessary means of developing the field. While they provide a rich body of descriptive work, they usual stop short of considering why conditions existed.

On the other hand, ecology of the city pushed urban ecology toward more meaningful, large-scale analyses of cities; this approach is commonly employed today in studies that consider, among other things, the metabolism (Warren-Rhodes and Koenig 2007) or ecological footprint (Bai et al. 2008) of entire urban systems, as well as the spatial distribution of amenities and human populations within cities (Landry and Chakraborty 2009). The methodology does not separate an urban
environment into components, rather all physical and biological aspects are treated as interconnected and are all thought to be influenced by changes that occur within any part of the city environment. Ecology of the city studies investigate the affect that changes - which are generally the result of some change to the intensity or frequency of an input to the city system - have on the city as a whole. Thus, this approach moves beyond description, seeking to identify explanation.

In their paper outlining the two methods of urban ecological study, Grimm et al. (2000) cited case studies from the project they were presently engaged in. The project was an extension of the US Long-Term Ecological Research network that had newly expanded to include urban areas, specifically, Baltimore, Maryland and Phoenix, Arizona. They claimed that the application of the ecosystems approach to these urban environments was novel at that time. Since then, the number of studies that have applied this methodology have increased substantially. It is currently employed at 26 sites within four countries (Long Term Ecological Research Network 2010), demonstrating the increased prevalence of urban ecosystem-oriented studies.

Grimm et al.’s (2000) paper did more than simply emphasize the necessity of considering the urban ecosystem as an assemblage. It also argued that in order to create truly successful studies of urban ecosystems, the natural sciences and the social sciences were going to have to work together. Up to this point, urban ecology had been notably deficient in significant collaboration between social and natural scientists and the capacity of studies to represent the true reality of urban systems had suffered as a result. Human structure and activity within urban spaces were acknowledged as being inseparable components of the urban ecosystem, however, the influence that cultural, economic and political forces have on ecosystem functioning had regularly been significantly simplified and underestimated in conventional urban ecology models in the physical sciences.

A similar sentiment has also been expressed within the social sciences. Braun (2005) conducted an analysis of the role of ecology in social urban studies and
found that, while discussion of natural ecosystems had occurred in the social science literature, it had never been a substantial focus. The natural component of urban systems had been just as overlooked in social studies, as social components had been ecological studies.

Alberti (2005), a strong advocate for collaboration between the two realms, reiterates that prior to the twenty-first century, little work was being done within the field of urban ecology to fill the gap that existed concerning the breadth of diversity, complexity and connectivity that operates within urban ecosystems, both social and natural. Her concern was that very often, studies that included social measures used only a single condensed variable, such as population density, to represent all social influence. She asserts that it was an entirely inadequate way of understanding an urban ecosystem.

Over the last decade, urban ecology has made a move toward more comprehensive urban analyses and ecosystem-level thinking. In addition, an increasing number of successful amalgamations of social and natural science now exist in the urban ecology literature. This has lead to many new and important findings about the inter-relationships within city systems. Researchers from both sides have recognized this shift as productive and vastly beneficial, though further room for development still exists (Alberti 2005; Braun 2005).

2.3 The Modern Discipline of Urban Ecology

This section considers recent urban ecological studies with an emphasis on those that focus on vegetation. Within the current literature, several patterns and themes are present. One pattern that emerges is that most urban vegetation studies tend to fall within one of three methodological approaches: urban to rural gradients, large scale canopy cover assessments using remotely sensed data, or ground based stem counts. Another pattern is that studies tend to either focus on one particular land use type within the urban environment (for example, park or private land), or else they group all land use types within a city under the single designation of
“urban” land and compare that land use against other broadly defined non-urban land use categories. Finally, studies that explore social drivers tend to focus on correlations between vegetation measures and a few key socioeconomic measures. This section will consider each of these patterns in turn.

2.3.1. Methodological Approach

There are many studies within the urban ecology literature that utilize the first methodological approach, examining the distribution of organisms along a gradient stretching from a high density urban centre to a point in the surrounding low density landscape (McKinney 2002; Zerbe et al. 2003; Conway and Hackworth 2007). Studies that employ an urban to rural gradient methodology test the assumption that the intensity of urban activity has a major impact on ecosystem components and processes. It is also useful for addressing hierarchy and disturbance theory and fluxes (McDonnell and Pickett 1990). Hierarchy theory is useful in quantifying the level of ecological impact that occurs at varying levels of urbanization in order to determine at which scales different effects become observable. Disturbance theory compares the ecological outcomes from different types of disturbances and investigates the consequences of changes in the frequency and intensity of impacts. Finally, fluxes – within community or species compositions, for example – may also be compared. Thus, urban to rural gradients provide a strong method for constructing comparisons between locales.

An example of an urban to rural gradient methodology is Zerbe et al. (2003) in Berlin, Germany, who tested for correlation between diversity of land use patterns and plant species richness and applied their findings to disturbance theory. Based on an urban to rural transect, they observed that diversity and richness peak in the middle urban densities, which they attributed to the types of land use present and historical trends of planting in the area. This somewhat surprising result of diversity peaking at middle densities in urban to rural gradients has been documented by others (McKinney 2002), representing a key finding of this approach.
However, certain limitations are associated with this methodology as transects are often too rigid and simplistic to truly reflect the distribution of urbanization levels that they encompass. For example, use of an urban-rural transect is based on the assumption that the underlying urban structure is monocentric, when often cities are not (Conway and Hackworth 2007). Another limitation of the method is that very often the units used to measure urbanization along a gradient are study-specific, therefore the results of different studies cannot be compared (McDonnell and Hahs 2008).

The second commonly utilized methodology to study urban vegetation involves measuring total vegetation or tree canopy cover using satellite imagery. As the technology advances, this method is being used in an increasing number of urban vegetation studies (Dwyer and Miller 1999; Iverson and Cook 2000; Heynen and Lindsey 2003; Grove et al. 2006; Conway and Urbani 2007). This is a particularly useful method for assessing the overall health or total abundance of an urban forest and for examining patterns across large areas. Results can obtained from this method at the state or nation level and still provide useful information at the local level. For example, Heynen and Lindsey (2003) employed advanced very high-resolution radiometer (AVHRR) data in a study of Central Indiana and were able to observe a positive relationship between canopy cover in urban areas and total forest cover in the surrounding county. Without the ability to conduct such large-scale spatial comparisons, this relationship would not have been noted.

Large-scale data collected remotely may also be used to identify variation in canopy cover between different urban areas and to address questions regarding how these variations can be explained. They can provide estimates of total attainable canopy cover for urban forest managers to help set goals and can be used to identify particular areas of concern within an urban forest (Heynen and Lindsey 2003). It is also possible to use aerial photographs and other remotely sensed spatial data in combination to achieve a range of spatial resolution values to reflect projects that are focused on large or small scale patterns (Dwyer and Miller 1999). A major limitation of this methodological approach is the inability to accurately
identify urban tree species using remotely sensed data, particularly when species richness is high (Voss and Sugumaran 2008).

The third methodology uses on-the-ground surveys of urban sites to obtain detailed stem counts of urban trees. While this method is more labour intensive than using remotely sensed data, for example, it yields data with much higher levels of detail, such as individual species counts, precise diameter at breast height (dbh) measurements, and can capture information about tree health that may not be obvious from canopy cover. Because of the specificity that is possible to obtain using this method, it is considered the best way to assess urban forest composition and health (Nowak 2008). However, because it is often not possible to assess every tree within a city, given time and financial limitations, a representative sample is often attained using a collection of small plots. The size, distribution and number of plots established vary from study to study depending on the objective of the project.

For example, in a study of Beijing City, Zhao et al. (2010) established 124 plots, 10 x 10 m each, distributed across six land use types, while in a study of Halifax, Nova Scotia, Turner et al. (2005) used 26 plots, approximately 25 x 25 m each, distributed across three land use types. Both studies were able to calculate and compare measures of species richness between land use types, as well as determine ratios of native to non-native species abundance within land uses despite employing different sampling methods. The ground-based stem count methodology allowed both studies to tailor their sampling methods to reflect their unique research goals and the number of land use types they wished to compare.

Because on-the-ground tree counts can be easily customized to the needs of individual landscapes and study objectives, many have been successfully employed in vegetation studies conducted around the world (Sudha and Ravindranath 2000; Jim and Liu 2001; Zipperer 2002; Martin et al. 2004; Lundholm et al. 2006; Weifeng et al. 2006; Godefroid and Koedam 2007; Jim and Chen 2008; Jim and Chen 2009; Luck et al. 2009; Walker et al. 2009; Zhao et al. 2010). However, the relative level of work required by this approach means that this method is typically limited to small
study sites, leading to a dearth of information on urban forest diversity, and more specifically, on the social drivers of such diversity across urban areas.

2.3.2 Specificity of Land Uses

Within the urban ecology literature, studies of vegetation tend to examine only a single land use type within the urban landscape or treat all urban land uses the same. For example, there has been an entire series of Urban Domestic Garden (UDG) studies conducted, which exclusively examine private residential yards, only one segment of the larger urban landscape (Thompson et al. 2003; Smith et al. 2006; Alison et al. 2008). Similarly, other land uses, including urban parks (Zipperer and Zipperer 1992; Weifeng et al. 2006), forest patches (Hobbs 1988), and golf courses (Tanner and Gange 2005), have also been examined in isolation from surrounding land uses within urban ecology studies.

Although this type of research neglects other important components of the urban landscape, there is significant merit in conducting such narrowly focused studies. In their UDG study, Alison et al. (2008) argue that even though individually gardens most often cover only a small area, in aggregate they make a substantial contribution to urban ecosystems. Therefore they conclude that limiting their study strictly to private gardens is justifiable. In this case, Alison et al. compared the composition and diversity of all plant species in their sample of domestic gardens between five cities in the United Kingdom. As well as providing important detailed site-specific information of these cities, the patterns they identified with respect to the habits of gardeners contribute to determining some of the factors that control species richness and tree abundance in urban residential areas. Moreover, they observed that species richness was substantially higher in garden sites than in surrounding natural sites; this pointed to the strong influence that human intervention has on determining garden species richness. This type of narrowly focused study produces an important level of detail that studies conducted across more heterogeneous land uses would miss.
At the other end of the spectrum, many studies classify all urban areas as a single land use type. Walker et al. (2009) used this method of classification in their study of the effects of urbanization on plant diversity in Phoenix, Arizona. They initially examined three land use types: desert, agriculture and urban. Their results, however, emphasized the importance of conducting analyses at different scales. They found that species richness values were higher on urban land than desert land. When they broke their urban land use category into more specific land use types (commercial, vacant, transportation and residential), they found that within each, diversity was actually quite low. From this they drew the conclusions that on the local scale, where human influence has a more direct impact, species homogenization occurs in response to landscaping trends and the voluntary introduction of exotic species contributes to greater evenness within species abundance. Without the use of more detailed land use classes, this insight would not have been accessible.

Another study that considered the urban environment as distinct land use types was conducted by Conway and Hackworth (2007). They used satellite imagery to examine the relationship between vegetation abundance (using the Normalized Difference Vegetation Index (NDVI)) and urban land use in the Greater Toronto Area. They employed six distinct land use classes: open land, parks and recreational land, residential, industrial, commercial, and government and institutional land. Of these six land use type classes, “parks and recreation land” consistently had the highest NDVI values along four transects (though in one of the four transects, open land has very similar NDVI values) and “commercial” land consistently had the lowest. This study was very useful in identifying the relationships between urban density, socioeconomic factors and NDVI however, the authors conceded that more detailed analysis at the species level is required to improve understanding of urban ecological processes.
2.3.3 Incorporating the Social

Significant attention has started to be given to the potential social, cultural and economic controls of urban ecosystems, which has a considerable potential to increase current understanding of urban vegetation patterns. The three most common ways researchers try to capture human influence on urban ecosystems is by focusing on population density, road structure, and the environmental equity hypothesis.

Human population density has been extensively investigated as a potentially strong control over urban tree abundance. It is an important measure that has been used to explore one of the fundamental questions regarding tree success in urban environments: is it possible for humans and trees to exist together at high densities? Perhaps contrary to initial assumptions, many studies have produced results indicating that it is possible (Iverson and Cook 2000, Luck et al. 2009). While these findings cannot be assumed to apply to all urban environments, they do provide an optimistic prognosis for urban forest success and rationalize further research and efforts to increase tree population levels in urban settings.

Despite this, population density is an inherently simple measure. In other studies, it has failed to identify trends in patterns of vegetation abundance where alternative socioeconomic measures, such as income and education, were successful (Heynen and Lindsey 2003; Conway and Hackworth 2007). Therefore, population density is most usefully employed in combination with other social indicators.

There are also a variety of studies that have focused on the relationship between road networks and tree health. While some studies focus on the type of pollution associated with roads and the direct mechanisms by which pollution affects tree health (Forman and Deblinger 2000; Eigenbrod et al. 2009; Avon et al. 2010), others focus more closely on patterns of tree distribution along city streets (Jim and Liu 2001; Jim and Chen 2009). Very often these streets are located in residential neighbourhoods and the studies incorporate aspects of the
socioeconomic conditions attributed to those residential neighbourhoods (Grove et al. 2006; Landry and Chakraborty 2009). However, road networks alone cannot be considered responsible for shaping their surrounding vegetation. Intrinsically, roads expose nearby vegetation to numerous other sources of pollution and interference, both human and not. Therefore, additional factors must be considered to fully understand the controls that shape road-side vegetation.

Finally, the environmental equity hypothesis focuses on the uneven distribution of access to environmental resources and urban amenities within a city (Landry and Chakraborty 2009). The hypothesis draws on urban political ecology research that has considered the location of harmful elements such as toxic storage and disposal facilities (Pastor, Jr. et al. 2001; Harner et al. 2002) or disruptive amenities, such as transportation systems (Chakraborty 2006) within urban environments. The distribution of favourable amenities, such as city parks (Solecki and Welch 1995; Talen 1997) and trees (Pedlowski et al. 2002; Heynen 2003; Perkins et al. 2004; Landry and Chakraborty 2009), have also been investigated. These studies largely consider measures of access across gradients of race and wealth. In most cases marginalized groups, racial minorities and low-income residents experience the highest levels of exposure to negative elements and the lowest levels of exposure to beneficial amenities (Landry and Chakraborty 2009).

For example, Pedlowski et al. (2002) conducted a study on the relationship between levels of wealth and the number and species diversity of trees in gardens and along residential streets in Campos dos Goytacazes, Rio de Janeiro, Brazil. Their findings indicated that in Campos dos Goytacazes, street trees were indeed unevenly distributed across income levels, leaving mid to low- income level residence at a disadvantage. They surmised that because of the direct positive influence that tree presence has on quality of life, street trees constitute an area of research that deserves a great deal of focus, as do all elements of the urban environment that similarly impact the health of its residents.
At this point in time, the distribution of positive elements across urban environments have received less attention in the literature than have negative elements. Highlighting uneven access to positive environmental elements is a key step to ensuring urban trees and forests can be distributed equally amongst all citizens.

2.4 Conclusion

This chapter has outlined the development of the field of urban ecology and described key themes found in the recent urban ecological literature, with a focus on urban vegetation studies. As a relatively new discipline, there are still substantial gaps in the literature as it stands.

One of these gaps concerns the lack of studies that investigate urban tree composition and distribution at the species-level. In order to optimize the ability of urban areas to adapt to imminent changes in population, built-structure density and climate, studies must view tree communities and stands as assemblages of distinct species, rather than simply homogeneous canopy. A second gap in the literature is the lack of consideration that currently exists between urban land use types. The distribution of individual species must be examined across the full variety of distinct land use types that constitute an urban environment in order to further understand how planning practice can directly impact species diversity and distribution in urban areas. Present studies that focus on either one land use type or group all urban land uses together overlook an important level of resolution.

Finally, a gap exists in the literature that results from insufficient collaboration between the social and physical sciences. Although significant progress has been made in the past decade concerning this deficiency, there is still a lot of ground to cover.

Chapters Four and Five in this thesis attempt to primarily address the first two of these gaps, and to a lesser extent the third gap. Chapter Four investigates the specific tree species composition of eight land use categories within the region of
Peel. Measures of species richness, diversity and evenness are identified and patterns within land use types, between land use types, and across the region as a whole are investigated. Chapter Five relates measures of species richness and tree abundance within residential neighbourhoods in Peel to nine socioeconomic and urban form variables. In addition, it examines the effect that the proximity of trees to roads has on species richness and tree abundance.
Chapter Three
Study Area and UFORE Data

3.1 Peel Region Study Site

Peel Region occupies 1,254 km² of southeastern Ontario, falling within the bounds of 43°35’N to 43°52’N and 79°37’W to 80°0’W (Appendix 1). On its eastern edge it is bordered by Toronto, Canada’s largest city, and on its southern edge it is bordered by Lake Ontario. Peel is situated within the ecological boundaries of the Great Lakes-St. Lawrence Forest Region, which is characterized by species such as *Pinus strobus, Pinus resinosa, Tsuga canadensis,* and *Betula alleghaniensis* (Hosie 1979). As well, Peel contains Carolinian Forest species such as *Fagus grandifolia, Acer saccharum* and *A. rubrum, Tilia americana, Quercus rubra, Q. alba* and *Q. macrocarpa* (Waldron 2003). Interesting geological features within Peel region include the Niagara Escarpment, a UNESCO World Biosphere Reserve, and the Oak Ridges Moraine; both of which are part of Ontario’s protected Greenbelt land. Peel also contains a significant riparian ecosystem, the Credit River.

Because of it’s proximity to Toronto, Peel has experienced strong urban expansion pressure as urbanized land uses spread further from Toronto’s core (Conway and Hackworth 2007). Much of the urban land use transformation that has occurred in Peel Region has occurred within the last 40 years. The region also contains several major transportation corridors that experience high volumes of commuter traffic, and an extensive local road network.

This study looks at four specific municipalities within Peel Region: Bolton, Brampton, Caledon and Mississauga. The municipality of Caledon technically encompasses the town of Bolton, however, it is the urban township of Caledon East (hereafter referred to as Caledon) rather than the entire municipality that this study examines. Thus, the Caledon and Bolton plots occupy geographically discrete areas.
Table 1: Population and Land Area for Peel Region and Municipalities

<table>
<thead>
<tr>
<th>Region/Municipality</th>
<th>Human Population</th>
<th>Land Area (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bolton</td>
<td>26,478</td>
<td>19</td>
</tr>
<tr>
<td>Caledon</td>
<td>2,604</td>
<td>5</td>
</tr>
<tr>
<td>Brampton</td>
<td>433,806</td>
<td>267</td>
</tr>
<tr>
<td>Mississauga</td>
<td>668,806</td>
<td>289</td>
</tr>
<tr>
<td>Peel</td>
<td>1,159,405</td>
<td>1,254</td>
</tr>
</tbody>
</table>

As Table 1 indicates, Mississauga and Brampton are the largest, most populated of the four municipalities. Mississauga contains one of Ontario’s largest shopping centres, the University of Toronto Mississauga campus, as well as extensive residential and industrial land uses. Brampton is also densely populated with a number of Big Box Store shopping complexes, which are a common form of development throughout the GTA, as is its extensive suburban-style development. Bolton and Caledon are much smaller and are located further from Toronto. Bolton has a more substantial percentage of non-urban and agricultural land than either Mississauga or Brampton. Caledon is also situated in a more rural landscape, serving as a rural town centre.

3.2 UFORE Data

The tree and plot data used in this project were obtained from the Toronto and Region Conservation Authority (TRCA), originally compiled for use in the Urban Forest Effects (UFORE) Model. The Model was developed by the United States Department of Agriculture Forest Service Northeastern Research Station with the intention of quantifying urban forest structure and functioning, as well as identifying environmental effects and habitat values (Nowak and Crane 1998). The UFORE Model methodology is often-used and well-trusted; it is currently being employed in more than fifty cities, a third of which are outside the United States.
(Nowak 2008). Most municipalities within the Greater Toronto Area have now completed UFORE data collection and analysis (Nelson, personal communication). Outside of the United States and Canada, studies that employ UFORE data have examined Mexico City (Escobedo and Chacalo 2008), Milan and Florence, Italy (Paoletti 2009), and London, England (Tiwary et al. 2009), among others.

The model has been used to address a wide range of research topics, including identifying the effect that green roofs and walls have on air pollution in Toronto (Currie and Bass 2008), designing a strategy to optimize tree planting initiatives in New York City (Morani et al. 2010), quantifying the carbon storage of urban trees in Syracuse, New York (Myeong 2003), and modeling urban hydrological processes in Baltimore, Maryland (Wang et al. 2008). As well, UFORE data has been applied to more diverse topics ranging from optical engineering (Heisler et al. 2003) to identifying potential fire behavior for various vegetation types and the associated societal benefits in the eastern Sierra Nevada Mountains (Dicus et al. 2009). Studies employing UFORE data have also covered a wide range of scales, spanning from city-specific (Nowak et al. 2006; Keen 2010) to continental-scale (Cumming et al. 2008). However, despite the wide application of UFORE data in the literature, the methodology has not yet been use in conjunction with socioeconomic data.

Most commonly, the UFORE Model is used to create city-specific profiles and address questions about the forest structure, biogenic emissions, carbon storage and sequestration, air pollution removal and building energy effects associated with that city (Nowak et al. 2008). It addresses these topics through four modules (Table 2). The UFORE model effectively allows for cities around the world to be easily compared based on values such as tree density, percent tree cover, net carbon sequestered, and hourly pollution removal (in both mass and dollars).
Table 2: Component Outputs of the UFORE Model

<table>
<thead>
<tr>
<th>Module</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>UFORE – A</td>
<td>Anatomy of the urban forest</td>
</tr>
<tr>
<td>UFORE - B</td>
<td>Biogenic volatile organic compound emissions</td>
</tr>
<tr>
<td>UFORE – C</td>
<td>Carbon storage and sequestration</td>
</tr>
<tr>
<td>UFORE – D</td>
<td>Dry deposition of air pollution</td>
</tr>
</tbody>
</table>

Each of these modules depends on detailed data collection from ground based sites and local hourly pollution and meteorological data. The UFORE data that was implemented for this thesis project relates only to the UFORE-A module therefore it will be the only one to be discussed further.

The aim of UFORE-A is to quantify the structural components of the urban forest through species composition, tree density, tree health, and leaf and biomass measurements (Kenney et al. 2001).

To complete UFORE-A for Peel Region, ground surveys were conducted on sample plots, including a complete census of each tree within those plots (Appendix 2). A total of 279 ground based plots were established during the UFORE data collection. Each plot was 11.3m in diameter (a total circular area of 400m²). Plot sites were selected throughout the study area using a random sampling technique stratified by land use. Each land use type was assigned a number of plots in proportion to its total amount of canopy cover (the land use descriptions and resultant distribution of plots within them can be found on page 34 of this thesis). Plots were then distributed randomly within each land use type. This is the standard methodology for UFORE data collection.

Field data was collected by the TRCA in 2006. After the field data was collected, two datasets were produced. The first included the land use type and land cover type (both organic and inorganic) at each site. The second had characteristics of the individual trees contained within each plot. Each plot was assigned a unique
ID for cross-referencing between the two datasets. The information that was pertinent to this study was the land use associated with each plot as well as the number of trees and species of each tree located within each plot.

Given the extent of detailed information pertaining to the location of individual tree species and the associated land use of their plots, this thesis is able to significantly expand upon the UFORE Model output by identifying patterns and correlates of urban forest diversity specific to Peel. Because UFORE data has not been used in relation to urban socioeconomic drivers in previous studies, this research presents a new and powerful application for the Model output.
Chapter Four
Diversity Measures

4.1 Introduction

There are many reasons why the preservation of natural biodiversity is important in an urban ecosystem. In general, a high level of biodiversity provides a high level of security against environmental changes and stochastic events by creating greater potential for adaptation and survival (Jim and Chen 2009). Greater urban biodiversity also increases the amount of exposure that city residents have to a wide spectrum of nature, which is an important element in stimulating people’s desire to support conservation (Goddard et al. 2010). Finally, greater levels of biodiversity allow for more complex ecosystem functioning, higher productivity, and create more niche opportunities that positively feed back to further increase biodiversity (Jim and Liu 2001).

Land use change and climate change exacerbate changes in urban species composition and overall biodiversity (Grimm et al. 2008; McKinney 2008). With respect to tree species, these changes have been shown to increase the proportion of exotic species, which can have strong negative impacts on the success of native trees (McKinney 2006; Goddard 2010). Exotics have also been shown to increase disease transmission, and can have both economic and conservation consequences (Goddard 2010). As well, an increase in the proportion of exotic species is thought to lead to an increase in biotic homogenization globally. This is reflected in a decrease in the distinctiveness of local vegetation, and a reduction in local and global species richness (McKinney 2004).

In order to protect biodiversity, urban landscapes must be studied and a better understanding of how they function obtained (Dearborn and Kark 2009). Because urban areas often portend the future environmental conditions of the landscape that surrounds them, a thorough comprehension of their ecosystem
dynamics can positively affect an area much greater than the city itself (Dearborn and Kark 2009). Knowledge of how a landscape is likely to operate in the future can be applied to facilitate floral adaptation within urban and future-urban areas and can help mitigate some of the negative consequences associated with a changing climate. Thus, the benefits that come from understanding the species structure of an urban ecosystem cover large temporal and spatial scales.

The goal of this chapter is to explore the tree species composition of Peel to understand general patterns in the distribution and composition of Peel’s urban forest across different land use types. The UFORE data collected by the TRCA was used to calculate several measures of diversity, methods commonly applied in all branches of ecology (Magurran 1988). Specifically, species richness, evenness, and alpha and beta diversity are quantified, with the latter measure focusing on differences between land uses and municipalities. The results contribute to an understanding of the functioning of Peel’s urban trees and point to particular management issues.

This chapter begins by discussing the specific indices that are used to describe each measure of diversity. The best metrics are then identified and applied to the Peel UFORE data. All measures were calculated by land use type and municipality. The results of the analysis are then discussed to highlight differences as a result of specific human activities or set of municipal policies. The discussion focuses on what these values reveal about the urban biogeography of Peel.

4.2 Literature and Background

In a textbook outlining the principles of applying diversity measures to ecological studies, A.E. Magurran (1988) asserts that diversity is absolutely central to the study of ecology, and supports this by noting that diversity measures are very often used as indicators of ecological wellbeing. She also notes that a considerable amount of debate has occurred over past decades in association with the best measurement of diversity. The debate stems from the proliferation of measures
that have derived from R.H. Whittaker’s (1960) original definition of the units of alpha, beta and gamma diversity. Which of the new measures is best remains to be decided as each performs differently under varying circumstances. Frustration over the lack of convention regarding which methods to apply exists because it makes comparison between studies very difficult (Magurran 1998; Southwood and Henderson 2000). The following sections describe the metrics commonly used to measure species richness, alpha diversity, species evenness and beta diversity.

### 4.2.1 Species Richness

Species richness is a description of the number of unique species found within a given study area. It can be measured in several ways. The most straightforward way involves simply examining the species numbers tallied in field counts. However, in sampled surveys, the observed species richness rarely provides an accurate estimation of total population richness (Walther and Moore 2005). Improving this value requires increasing sample size, which can defeat the cost and time saving purposes of taking samples. As well, variation in sampling intensity between separate communities limits inter-community comparisons. To overcome this, a number of estimation methods can be used to assess the validity of species richness derived from samples.

Species accumulation curves are an effective way of communicating whether sampling effort has been exhaustive or not. They plot the number of samples taken in a survey against the cumulative number of unique species found within that many samples (Magurran 2004). When the curve reaches an asymptote it is assumed that maximum species saturation has been reached and that the sampling effort reflects the true species composition of the entire population (Seaby and Henderson 2006).

Walker et al. (2009) used species accumulation curves to distinguish levels of plant species richness across landscape types in Phoenix, Arizona and found that they could also be used effectively to account for sampling biases attributed to scale. The technique will be used in this study to detect any bias that may occur as a result
of differences in sampling intensity when comparing species richness counts of all land use types and municipalities.

4.2.2 Alpha Diversity

Alpha diversity is a heterogeneity measure, meaning it combines the influence of richness and evenness to produce a single, community-specific, diversity value. It is commonly described as providing a measure of “within community diversity” (Southwood and Henderson 2000). Two metrics of alpha diversity will be used in this study to quantify the amount of tree species diversity contained in each of Peel’s distinct land use types and each of its municipalities.

The most common measure of alpha diversity found in the literature is the Shannon-Wiener index, $H'$. It is a non-parametric index based on the proportional abundance of species taking into account both species richness and evenness. It can be determined using the equation:

$$H' = -\sum p_i \ln p_i$$

Where $p_i$ represents the proportion of the population that belong to the $i$th species. Typical values for $H'$ range from 1.5 to 3.5 when measurements are obtained from empirical data (Margarren 1988). Not particularly robust on its own, Southwood and Henderson (2000) describe the Shannon-Wiener Index as a “distraction” because of how strongly it is influenced by species number. However, because of its prevalence in the literature, it provides a useful comparison against other studies. None of the many attempts that have been made to improve upon the measure have so far succeeded in replacing it with something as widely accepted. However, as a result of these attempts there is an overwhelming lack of convention found in the literature’s use of alpha metrics. Thus, use of $H'$ is justified as it is one of the only means for comparison between most studies.

For example, $H'$ was calculated in studies of Taipei (Jim and Chen 2009) and Bangalore City (Sudha and Ravindranath 2000). Using this metric the level of
diversity within unique land use types within each particular city was determined (i.e. in Bangalore, the diversity of trees on institutional land ($H' = 1.475$) is less than on commercial land ($H' = 1.549$) and on residential land ($H' = 1.605$)). This then makes it possible to determine how the levels of diversity within similar land uses compare between the two cities. Jim and Chen found park land to have a $H'$ value of 3.76, for example, while Sudha and Ravindranath found $H'_{park}$ to be 1.697. From this it can be concluded that Taipei possesses greater diversity within its urban parks than does Bangalore City.

Waite (2000) places value on the Shannon-Wiener index's sensitivity to rare species. Because of this sensitivity, $H'$ can be effectively employed in tandem with another measure of alpha diversity, Simpson's $D$, which is much more sensitive to change in the abundance of common species (Waite 2000).

Simpson's Index, $D$, is the probability of any two individuals drawn at random from an infinitely large community belonging to the same species (Magurran 1988). It is a dominance measure and is much more sensitive to the most abundant species in a sample than it is to species richness. Southwood and Henderson (2000) note that this can be a helpful quality for considering inter-specific encounters. $D$ is calculated using the following formula:

$$D = \Sigma p^2_i$$

The relationship between the diversity of a population and the value of $D$ is inverse. As diversity increases, $D$ decreases from 1 to 0, somewhat counter intuitively (Waite 2000). Because of this, the measure is often employed in the more practical forms of $1-D$ or $1/D$. Here, the reciprocal form will be used, denoted as $C$. The lower bound for $C$ is 1 (minimum diversity) and the maximum value for each sample is the number of observed species (maximum diversity). Together, $H'$ and $C$ illustrate the distribution of species abundance within a community and contribute to a comprehensive analysis.
4.2.3 Evenness

Measures of evenness consider the relative abundance of species within a given community. High levels of evenness are obtained when the number of observed individuals is distributed uniformly across all the species in a population. It is an important measure to consider as it has been suggested that the relative abundance of species can have a greater impact on the diversity and ecological function of a landscape than does species richness (Valery et al. 2009).

Evenness can be easily visualized using rank-abundance graphs. These graphs plot species in order from most abundant to least along the x-axis and species abundance on a log scale along the y-axis. Most species abundance data has been found to take one of four main shapes when graphed this way: geometric, log, log normal or broken-stick (Magurran 1988). Each model infers different ecological conditions.

Data fit to the geometric model depicts few very common species and a large proportion of rare species. This often reflects harsh environments or environments in early successional stages where a few species have assumed control of the available niches. The log model depicts species distributions where mid-rank abundances are fairly common. It is often representative of habitats where the ecosystem success is strongly controlled by a small number of factors, such as light availability for example. The log normal distribution fits an even greater degree of commonality in mid-rank abundances than does the log model. It is the most commonly observed distribution in nature and often reflects well-established and diversified communities. The broken-stick model is the closest that naturally occurring species distributions come to being equal. It is not observed to occur very often though it could result from niche overlapping or if there is even sharing of resources among species (Magurran 1988).

To obtain a quantifiable measure of the distribution of diversity within a community, it is possible to manipulate the Simpson's Index of diversity, D. The
complimenting evenness measure, $E_{1/D}$, that this produces is achieved by using the following formula:

$$E_{1/D} = (1/D)/S$$

Where $D$ is the Simpson’s Index and $S$ is the number of species in the sample. The resultant evenness measure ranges from 0 (no species equitability) to 1 (total species equitability) This measure has been applied at least once in an urban tree analysis, finding evenness of 0.74 for urban parks and 0.70 for street trees (Jim and Chen 2009).

### 4.2.4 Beta Diversity

Beta diversity is the measure of the difference in diversity that exists between communities. It will be used in this study to quantify the amount of species diversity that exists between each of the distinct land use types and municipalities. Beta diversity increases (from 0 to 1) as the number of shared species between communities decreases (Magurran 1988).

The original measure of beta diversity, $\beta_w$, was developed by Whittaker (1960). Other metrics of beta diversity have since been proposed, however $\beta_w$ has maintained its value. This was illustrated by Wilson and Shmida (1984), who conducted a test to evaluate six competing measures of beta diversity. Based on the amount of influence that community change, additivity, independence from alpha, and independence from sample size had on each beta measure, they concluded that $\beta_w$ remains the strongest beta diversity metric (Magurran 1988). It is calculated using the following equation:

$$\beta_w = (S/\alpha_{\text{mean}})-1$$

$S$ represents the total number of species in the system and $\alpha$ represents the average sample diversity assuming that all samples are a standard size and diversity is species richness (Magurran 2004). Because of this dependence on standard sample size, $\beta_w$ is not a valid measure to use for the Peel data. In order to overcome this
dependency, an adaptation on $\beta_w$ that was introduced by Harrison et al. (1992) may be used. Harrison's measure, $\beta_{H1}$, is calculated as:

$$\beta_{H1} = \{[(S/\alpha) - 1]/(N-1)\}*100$$

Where N is the number of plots in a sample population.

4.3 Methods

This study was conducted at the species scale to examine the variation in species composition for various land use functions within an urban environment. UFORE data, collected during the summer of 2006, was used to provide tree counts, species descriptions and land use descriptions for each of 279 plots within Peel (Appendix 2). The land use categories assigned by the UFORE survey differ slightly from the categories used by the Peel Regional Municipality, however it was necessary to use the Peel Regional system in order to calculate the total area of each land use across Peel. The associations between each source's classification system and the percentage of the study area that each land use occupies are listed in Table 3.

Table 3: Land Use Classification Systems and Percentages of Total Land Area

<table>
<thead>
<tr>
<th>Peel Land Use Designation</th>
<th>UFORE Land Use Designation</th>
<th>Percent of Total Study Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Commercial</td>
<td>Commercial/Industrial*</td>
<td>1</td>
</tr>
<tr>
<td>Government and Institutional</td>
<td>Institutional</td>
<td>6</td>
</tr>
<tr>
<td>Open Area</td>
<td>Agriculture, Vacant</td>
<td>31</td>
</tr>
<tr>
<td>Parks and Recreation</td>
<td>Park, Golf</td>
<td>6</td>
</tr>
<tr>
<td>Residential</td>
<td>Residential</td>
<td>39</td>
</tr>
<tr>
<td>Resource and Industrial</td>
<td>Transportation, Commercial/Industrial*</td>
<td>16</td>
</tr>
<tr>
<td>Water body</td>
<td>N/A</td>
<td>1</td>
</tr>
<tr>
<td>Total</td>
<td></td>
<td>100</td>
</tr>
</tbody>
</table>

*Some UFORE land use types fall under multiple Peel land use categories
First, the data were organized by land use type (according to the UF plausible classification system: agriculture, commercial/industrial, golf, institutional, park, residential, transportation, vacant). Simple species richness and total tree abundance were then calculated. Accumulation curves for each land use type were then created to test for exhaustive sampling.

An analysis of variance (ANOVA) was then applied to these datasets and used to investigate the strength of the relationship between both species richness and overall tree abundance for land use type and municipality. This was done using SPSS.

The results indicated that species richness and tree count are not distributed evenly across land use types or municipalities, justifying a much more in-depth analysis of these relationships. Therefore, the study moved on to explore more robust metrics of alpha diversity, species evenness, and beta diversity. Each of these measures was calculated using the software Species Diversity and Richness III (Seaby and Henderson 2006).

Alpha diversity was calculated using both the Shannon-Weiner Index and the Simpson’s Index. Evenness for each land use type and municipality was examined first by producing rank-abundance curves, allowing for assumptions to be made about the ecology of each community, and then using Simpson’s measure of evenness. Beta diversity was calculated between each land use type and municipality, resulting in 32 and 6 pair-wise comparisons, respectively. The pair-wise values for each land use type were then averaged and ranked according to what land use has the most to least in common with other land uses. Beta diversity was then calculated for each community as a measure of the diversity between plots of a common land use type.

The results are organized so that land use type-specific analyses are discussed first, followed by municipality-specific analyses. The distribution of tree abundance when broken down by both classification systems is then addressed.
4.4 Results

In total, the 2007 UFORE survey observed 4122 trees within Peel Region across 279 plots. This sample population was comprised of 179 different species. The ten most abundant species (and their percentage of the total tree population) are: *Thuja occidentalis* (15.1%), *Rhamnus cathartica* (10.7%), *Fraxinus americana* (8.2%), *Acer saccharum* (5.0%), *Rhus typhina* (4.2%), *Crataegus* (3.5%), *Populus aurea* (3.2%), *Acer negundo* (2.8%), *Picea glauca* (2.4%) and *Acer platanoides* (2.3%). Thus, three species account for 34% of all tress and ten species for 58%. On average, each plot had 15 trees and 5 species.

4.4.1 Species Richness and Tree Abundance by Land Use

Table 4 illustrates the total trees and species counts associated with different land uses:

**Table 4: Tree Counts and Species Richness by Land Use Type**

<table>
<thead>
<tr>
<th>Land Use Type</th>
<th>Raw Tree Count</th>
<th>Raw Species Count</th>
<th>Number of Plots</th>
<th>Average Number of Trees per Plot</th>
<th>Average Number of Species per Plot</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>22</td>
<td>10</td>
<td>7</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td>Commercial</td>
<td>126</td>
<td>43</td>
<td>37</td>
<td>3</td>
<td>2</td>
</tr>
<tr>
<td>Golf</td>
<td>116</td>
<td>15</td>
<td>3</td>
<td>39</td>
<td>6</td>
</tr>
<tr>
<td>Institutional</td>
<td>96</td>
<td>26</td>
<td>8</td>
<td>12</td>
<td>5</td>
</tr>
<tr>
<td>Park</td>
<td>450</td>
<td>48</td>
<td>22</td>
<td>21</td>
<td>5</td>
</tr>
<tr>
<td>Residential</td>
<td>1312</td>
<td>145</td>
<td>146</td>
<td>9</td>
<td>6</td>
</tr>
<tr>
<td>Transportation</td>
<td>212</td>
<td>29</td>
<td>10</td>
<td>21</td>
<td>5</td>
</tr>
<tr>
<td>Vacant</td>
<td>1788</td>
<td>88</td>
<td>46</td>
<td>39</td>
<td>6</td>
</tr>
<tr>
<td>Peel</td>
<td>4122</td>
<td>179</td>
<td>279</td>
<td>15</td>
<td>5</td>
</tr>
</tbody>
</table>
4.4.2 Raw Counts by Land Use

The first two columns of Table 4 present the raw values obtained from the TRCA’s survey. Immediately, residential and park land stand out as having substantially more trees and species than the other land uses, and agricultural land substantially less. However, because the number of plots differs so significantly for each land use, these values cannot be used to make comparisons between land use types. This problem is addressed further in upcoming sections using accumulation curves. In the meantime, the raw counts are used to identify intra-land use trends. Appendix 3 identifies the most common species for each land use; it is clear that between uses the dominant species vary considerably. This section will consider the species composition of each land use type individually.

Agricultural land plots possess the lowest values for both the raw tree abundance and species richness counts. Though there were very few sampled plots, both the average number of trees per plot and number of species per plot are the lowest for all land use types. The most prominent species in agricultural land, *Rhamnus cathartica* (European buckthorn), comprises 45% of its total sampled population. The remaining nine species were each observed only once or twice within the sample.

The most abundant tree species that grows on commercial land is *Picea pungens* (Blue Spruce) despite it having been observed only 12 times out of the total 126 trees. Within Peel as a whole, *Picea pungens* was sampled a total of 45 times and is its 17th most abundant species. Slightly greater than one quarter of all *Picea pungens* in Peel (27%) grow on commercial land. The second most abundant species on commercial land is *Picea glauca* (White Spruce), which is the ninth most abundant species in all of Peel. Together, both spruce populations comprise 18% of all the trees that were sampled in commercial land. Notably, half of the *Picea pungens* population was observed in a single plot and the rest of the population was distributed across four other plots.
Like agricultural land, golf courses include relatively large populations of *Rhamnus cathartica*, as it comprises 44% of the total sampled trees. *Pinus resinosa* (Red pine) is also an abundant species on golf courses (24% of the sampled population). The other 13 of the total 15 species comprise less than 6% of the total population each.

*Pinus resinosa* is the most abundant species that was sampled on institutional land. It comprises 22% of the total tree abundance and is one of 26 total species present. Approximately a quarter of all *Pinus resinosa* that was counted in Peel grows on institutional land, only golf courses harbor more (24% and 32% of the red pine population respectively). In fact, the only land use type on which the species was not found is agricultural land.

Park land is unique amongst the eight land use types for its large proportion of *Rhus typhina* (Staghorn sumac). Ninety-one percent of the total *Rhus typhina* population exists in park land, the rest grows on only four other land types (residential, 6%; vacant, 1%; transportation, 0.6%; commercial 0.6%). Within park land *Rhus typhina* comprises 35% of the total sampled tree population. The second most abundant species is *Acer saccharum* (Sugar maple), it makes up 13% of the population. Interestingly, while *Rhus typhina* is almost entirely exclusive to parkland, *Acer saccharum* is one of only three species in Peel that exists in all eight land use types.

The most common species in residential land, *Thuja occidentalis* (Northern white cedar), is also the most common species in vacant land and in the Peel sample as a whole. Despite this, the TRCA survey did not find any occurrence of it in agricultural, golf or institutional land. Within residential land, *Thuja occidentalis* comprises 14% of the total sampled tree population.

*Thuja occidentalis* is a useful species for demonstrating the difference between abundance and relative abundance. Though in absolute terms it has more trees on residential land (179 trees) than it does in transportation land (42 trees), the species comprises a lesser proportion of the residential land sample (14%) than
it does in transportation (20%). Therefore, although its raw abundance is greater in residential land, its relative abundance is greater in transportation land. Such distributional patterns may lead to misinterpretation of data. By simply considering the relative abundance data it may be concluded that northern white cedar is more successful in transportation areas than in residential areas. However, because the white cedar population is considerably more evenly distributed in residential land than it is in transportation land, it may actually be considered a representative component of the residential landscape while that may not be true of transportation land.

Transportation has the fifth highest observed species richness. Its most abundant species is *Populus aurea*, comprising 37% of the total sample population despite being only 3% of the total species composition. Sixty-nine percent of the total sampled *Populus aurea* in Peel was found in transportation land use areas. As noted above, *Thuja occidentalis* also comprises a significant component of its population.

Finally, vacant land has the highest total observed tree count and second highest observed species richness. Its most populous species, *Thuja occidentalis*, was found to occur 385 times in vacant land, which more than twice the population of the next most populous species in another land use (which is *Thuja occidentalis* in residential land with 179 trees). Even the second and third most abundant species in vacant land (*Rhamnus cathartica* and *Fraxinus americana*, respectively) have higher abundances than the most abundant species in other land uses. However, because of the high total number of species found in vacant land and the high abundance at which they also occur, the relative abundance of *Thuja occidentalis*, *Rhamnus cathartica* and *Fraxinus americana* are weaker than in other land use types.
4.4.3 Per Plot Averages by Land Use

The last two columns in Table 4 illustrate the variation between the average number of species and trees that exist on a per plot basis across all land use types. It is immediately obvious that plots with the highest tree density are found on vacant land and golf courses and that plots with the lowest tree density are found on agricultural and commercial land. The species richness is much more evenly distributed within a very narrow range of values. All land use types average species richness per plot values fall into one of two categories: 2 species, or 5 or 6 per plot. Golf courses, residential and vacant land are clearly the highest rank and agricultural and commercial land the lowest. However, by observing the raw data it becomes obvious that this value does not accurately represent the reality of species densities across plots. Interestingly, while vacant land has the highest average tree count per plot, it ranks much lower in terms of average number of species. Opposite to that, institutional land has a relatively low tree density but the second highest species diversity.

4.4.4 Species Accumulation Curves by Land Use

In order to address the problem introduced by differences in the intensity of sampling between land uses, species accumulation curves (Appendix 4) were used to identify which land use types had been sampled exhaustively and which were likely to have revealed greater species richness had they been sampled more thoroughly. The resultant curves can be grouped into four general categories: graphs that increase linearly, graphs that curve slightly, graphs that curve and graphs that come close to reaching an asymptote.

Land use types that fall into the first category, graphs that increase linearly, include: agriculture, golf and transportation. These land use types contained 7, 3 and 10 plots, respectively. It can be concluded from this that sampling so few plots is simply inadequate to accurately represent the species structure of the entire community. The second category, graphs that curve slightly, contains only
institutional land use. Although institutional land contains only 8 plots, its species accumulation curve suggests that this class was sampled more representatively than the three above. This finding reiterates the difference in composition of varying land use types and the necessity of examining each individually. The land use classes that appear to have been more thoroughly sampled, although not exhaustively so, are commercial land and park land. Both of these land uses contained 37 and 22 plots, respectively. The two land use types that came closest to be exhaustively sampled were vacant and residential land (46 and 146 plots each). Though neither of these graphs completely levels off, they do show indications of nearing an asymptote; therefore, the diversity values that they produce can be assumed to well represent the reality of their population.

4.4.5 Rarity of Species Across Land Use Types

Of the 179 total species that occur in the Peel Region, no single land use type accommodates all of them. In fact, 48% of species exist on only one land use type in the sample.

Figure 1: The Frequency of Species Dispersion Across Land Use Types

![Graph](image)

Figure 1 presents the relative abundance of species that occur for a given number of land use types. There is a very clear exponential decrease in frequency as the number of land use types a species is found in increases. Almost half of all the tree
species in Peel exist in only one land use type, however there is no consistency or
evenness as far as which land use type. Of those species that occur in one land use
and nowhere else, 69% were found in residential land, 20% in vacant land, 6% in
park land, 3% in transportation land, and 2% in commercial land. The samples for
agricultural, golf, and institutional land were not found to contain any species
unique to them. The average number of land use types that a species occupies is 2.
It is evident that land use type-specific species are much more common in Peel than
generalist species. The distribution of land use is thereby an important factor in the
distribution of species.

Of the two species that grow in all but one land use type (Fraxinus americana
and Pinus resinosa), neither grows on agricultural land. The three species that grow
in all eight land use types are Acer negundo, Acer saccharum, and Rhamnus
cathartica. Four of these five general species are native to the landscape, only
Rhamnus cathartica is not.

Thus, the majority of the trees in Peel belong to a narrow group of species
with high populations; the remaining trees are distributed across a wide group of
species, all with very low populations. Figure 2 (below) depicts the relationship
between the number of these rare species (defined as having only one or two trees
per species within the entire Peel Region sample) in each land use type and its
observed species richness. There is a very clear linear relationship between the two
variables: as the number of rare species increases, so too does the observed species
richness. The only land use type that deviates from the linear trend is golf land; it
has fewer rare species per number of observed species than the trend would
predict.
4.4.6 ANOVA by Land Use

An ANOVA was then used to determine if the difference in species richness and tree abundance varied significantly in relationship to land use type (Table 5).

<table>
<thead>
<tr>
<th>Null Hypothesis</th>
<th>Significance</th>
<th>Test Statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>The distribution of SR is the same across categories of land use types</td>
<td>.000</td>
<td>-11.08</td>
</tr>
<tr>
<td>The distribution of TC is the same across categories of land use types</td>
<td>.000</td>
<td>3.85</td>
</tr>
</tbody>
</table>

Significance level is .05

The relationship was deemed to be significant at the 0.01 level. The results of the ANOVA substantiated an in-depth analysis of influence that both land use type and municipal boundaries exerts on species richness and tree abundance.
Had sampling in each land use type been exhaustive, it would have been possible to compare each land use type based on the raw counts alone. However, the results of the accumulation curves indicated that that is not the case. Therefore, metrics of diversity will be considered to compare the composition of species within and between land use types and municipalities.

### 4.4.7 Alpha Diversity by Land Use

Table 6 provides the outcome of both alpha diversity indices, the evenness measure, and the between-plot diversity for each land use type:

<table>
<thead>
<tr>
<th>Land Use Type</th>
<th>$H'$</th>
<th>$C$</th>
<th>$E$</th>
<th>$\beta_{\text{HI}}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>1.855</td>
<td>4.813</td>
<td>0.4812</td>
<td>56.25</td>
</tr>
<tr>
<td>C</td>
<td>3.388</td>
<td>26.88</td>
<td>0.625</td>
<td>49.22</td>
</tr>
<tr>
<td>G</td>
<td>1.796</td>
<td>3.88</td>
<td>0.2587</td>
<td>75</td>
</tr>
<tr>
<td>I</td>
<td>2.686</td>
<td>10.36</td>
<td>0.3986</td>
<td>63.91</td>
</tr>
<tr>
<td>P</td>
<td>2.606</td>
<td>6.355</td>
<td>0.1324</td>
<td>41.37</td>
</tr>
<tr>
<td>R</td>
<td>4.082</td>
<td>28.85</td>
<td>0.1989</td>
<td>17.31</td>
</tr>
<tr>
<td>T</td>
<td>2.239</td>
<td>5.256</td>
<td>0.1813</td>
<td>56.02</td>
</tr>
<tr>
<td>V</td>
<td>2.948</td>
<td>9.528</td>
<td>0.1083</td>
<td>28.27</td>
</tr>
</tbody>
</table>

$H'$ = Shannon-Weiner Index; $C$ = Simpson's Index; $E$ = Simpson's Evenness Measure; $\beta_{\text{HI}}$ = Harrison's Beta Diversity

The majority of the results of the Shannon-Weiner Index, $H'$, fall within the normal range (1.5-3.5) reported in the literature for all ecosystems (Magurran 1988). Only residential land exceeds this range, indicating that it has the highest degree of within community diversity, and significantly so.

It is notable that the results of both the alpha diversity measures ($H'$ and $C$) rank diversity by land use in the same order (only vacant and institutional land alternate third and fourth rankings between the indices). For the large part, the simple richness and tree count rankings also rank land use similarly to each other.
(vacant and institutional land are the anomalies again, however; vacant land ranks first with respect to tree count and fifth with respect to species count, while institutional land ranks fifth with respect to tree count and second with respect to species count (Table 4)). However, there are substantial differences between the alpha rankings and the simple count rankings. Excluding vacant and institutional land, the two types of metrics are generally inverse. Four land use types exemplify this reversal particularly well: residential, golf, commercial, and transportation. Residential land, for example, possesses the highest alpha diversity values and has the highest species count per plot value, though it also has the third lowest tree count per plot value.

The three lowest $H'$ and C values are for agricultural, golf and transportation land use. These are notably the three land use classes that produced linear species accumulation graphs. Although under-sampling does not appear to directly bias tree count values (golf has the second highest trees per plot average, transportation the third highest, and agricultural land the lowest), it does appear to produce very low values relating to the species diversity within those tree communities.

Commercial land and park land produced curving, though not asymptotic, species accumulation graphs. Despite the similarity of these graph lines, commercial land has the second highest alpha values and park land the fifth. This illustrates that once sampling nears a comprehensive level, the unique compositional characteristics of the communities are observable.

The highest tree density for any of the land use types occurs in vacant land. However, even with the highest density and largest tree count values, vacant land returned lower $H'$ values than both residential and commercial land and lower $C$ values than residential, commercial, and institutional land. The proportional abundances of species that grow on vacant land largely reflect the proportional abundances at which they grow in Peel.
4.4.8 Evenness by Land Use

All of the rank-abundance graphs that were produced to illustrate the distribution of species abundance within each of Peel’s eight land use types (Appendix 5) fit one of two distribution models. These are the log or the log normal. Because of the limited number of sample plots obtained for each land use, it is difficult to determine precisely which model best applies to the data; this is a commonly observed phenomenon when using small datasets (Magurran 1988). However, both models represent species distributions that are uneven and that have few abundant and many rare species, though data better fit to the log normal distribution have a greater number of species at an intermediate abundance and therefore have slightly higher levels of evenness. By observing Appendix 5, it can be inferred that the urban forest in Peel Region is mature and diverse and is controlled by a narrow range of factors, although to varying degrees according to land use. As rank-abundance graphs are most useful for identifying general trends between species assemblages and ecological conditions, greater detail into the distribution of species within each of Peel’s land uses was obtained using the Simpson’s Index evenness measure.

The Simpson’s E values for all the land use types in Peel region were relatively very low (Table 6), which supports the findings that a relatively small number of species dominates the composition of Peel’s urban forest. This trend is least clear, but still is observable, for commercial land, where strong human influence through tree clearing and landscaping may help to curb the abundance of some trees. It is most strong for vacant land. Despite its high diversity, the distribution of trees within vacant land clearly favours a certain few. In each of the various land use types, patterns of evenness are manifest in several different ways; some of the more notable patterns are discussed here.

In total, only ten tree species were sampled in agricultural land. Of those, *Rhamnus cathartica*, by far the most abundant species, comprised 10 of the 22 trees in the sample and occurred in 5 of the 7 plots. Each of the other 9 species were
observed only one or two times in agricultural land. It is because of this consistency in (very low) abundance of 90% of the sample that the Simpson's E value is high for agricultural land.

The Simpson's E value for golf courses (0.2587) reflects the land use type's inequitable distribution. This inequitability is observable both within individual species distribution and with respect to total tree abundance. Fifty of the 51 Rhamnus cathartica trees present on golf courses existed in a single plot. Similarly, 26 of the 28 Pinus resinosa trees occupied one plot while the remaining 2 were found in another. Tree abundance has considerable unevenness; 106 of the 116 golf course trees exist in a single plot while the other two plots contain only 8 and 2 trees.

Commercial is the most even of all the land use types in the sample. It also has relatively high alpha diversity. The total number of species found in commercial samples is 43. Although twenty-one of those species were found to occur only once within the sample the maximum abundance value of any commercial species is 12. The relatively high species richness, combined with the narrow range of abundance values justifies the relatively high evenness value.

What is interesting about park land is that its evenness is expressed across both the landscape and plot level scales. As previously stated, 91% of the observed population of Rhus typhina throughout all of Peel exists in park land. Within the park land use category specifically the species is very unevenly distributed; 154 of the 159 Rhus typhina trees in parks exist in one plot. The remaining five plants are distributed across four plots.

The low Simpson's E value for vacant land (0.1083) reflects the fact that 65.2% of its total tree abundance is comprised of only 5 species and that 25 other species occur only once in the sample. The lack of evenness is also evident in the distribution of abundance among plots; 44.3% of all trees in vacant land grow in 11.0% of the plots.
4.4.9 Beta Diversity by Land Use

The pair-wise beta diversity between communities was calculated for all 32 combinations of land use types. Harrison’s beta diversity values are presented in Table 7.

Table 7: Pair-wise $\beta_{H_j}$ values of all Land Use Types

<table>
<thead>
<tr>
<th></th>
<th>A</th>
<th>C</th>
<th>G</th>
<th>I</th>
<th>P</th>
<th>R</th>
<th>T</th>
<th>V</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0</td>
<td>72</td>
<td>52.38</td>
<td>66.67</td>
<td>67.35</td>
<td>87.1</td>
<td>61.11</td>
<td>78.38</td>
</tr>
<tr>
<td>C</td>
<td>0</td>
<td>76.47</td>
<td>51.52</td>
<td>46.84</td>
<td>56.76</td>
<td>57.58</td>
<td>53.85</td>
<td></td>
</tr>
<tr>
<td>G</td>
<td>0</td>
<td>67.57</td>
<td>72</td>
<td>85.9</td>
<td>56.76</td>
<td>70.67</td>
<td></td>
<td></td>
</tr>
<tr>
<td>I</td>
<td>0</td>
<td>44.62</td>
<td>69.59</td>
<td>57.69</td>
<td>57.78</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>P</td>
<td>0</td>
<td>57.61</td>
<td>44.62</td>
<td>33.98</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>R</td>
<td>0</td>
<td>69.59</td>
<td>38.76</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>T</td>
<td>0</td>
<td>53.33</td>
<td></td>
<td></td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>V</td>
<td>0</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

A = agriculture, C = commercial, G = golf, I = institutional, P = park, R = residential, T = transportation, V = vacant

The $\beta_{H_j}$ values for Peel’s land uses cover a broad range. The pairs of land use types with the smallest difference between their species composition are vacant-park land and vacant-residential land, with both sharing approximately two thirds of their total species composition. The pairs of land use types that contains the most diversity between their species composition are residential-agricultural land and residential-golf course land, with only 15% of the species the same.
Table 8: Average Pair-wise $\beta_{H1}$ Values for Each Land Use Type

<table>
<thead>
<tr>
<th>Mean Pair-wise $\beta_{H1}$ Value</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>69.28</td>
</tr>
<tr>
<td>C</td>
<td>59.29</td>
</tr>
<tr>
<td>G</td>
<td>68.82</td>
</tr>
<tr>
<td>I</td>
<td>59.35</td>
</tr>
<tr>
<td>P</td>
<td>52.43</td>
</tr>
<tr>
<td>R</td>
<td>66.47</td>
</tr>
<tr>
<td>T</td>
<td>57.24</td>
</tr>
<tr>
<td>V</td>
<td>55.25</td>
</tr>
</tbody>
</table>

Table 8 presents each land use type’s average $\beta_{H1}$ values. The land use type that shares the greatest proportion of its species composition with other land use types is park land, while the land use type that shares the least is agricultural land.

Residential land experiences the third greatest diversity from the other land use types (mean $\beta_{H1} = 66.47$). It is also one component of both the most diverse and second most diverse land use pairings: residential-agricultural and residential-golf. Residential land also experiences the greatest range of pair-wise $\beta_{H1}$ values of all the land use types.

Transportation land is the third most similar land use to all the others. The land use type to which it is most similar is park land and it is least similar to residential land. This might indicate a lack of compatibility between the two land use types (transportation and residential) and should be taken into consideration from an urban planning standpoint. As urbanization proceeds and urban populations increase, roadways will unavoidably become more widespread, particularly to accommodate the subsequent increase in housing. The impact of
more roadways intersecting residential areas may have a negative effect on surrounding tree diversity.

The mean $\beta_{hl}$ value places vacant land second to last in the list of all other land use types. However, it does possess the second largest range of $\beta_{hl}$ values. It is most similar to park land and least similar to agricultural land.

Overall, high beta values generally correspond to midrange alpha values and medium to low evenness. Low beta values correspond to low alpha values and medium to high evenness. Midrange beta values correspond to medium to high alpha values and inconsistently ranked evenness.

4.4.10 Species Richness and Tree Abundance by Municipality

Table 9 shows the results by municipality, rather than by land use type.

<table>
<thead>
<tr>
<th>Municipality</th>
<th>Tree Count</th>
<th>Species Count</th>
<th>Number of Plots</th>
<th>Number of Trees per Plot</th>
<th>Number of Species per Plot</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bolton</td>
<td>480</td>
<td>74</td>
<td>26</td>
<td>19</td>
<td>7</td>
</tr>
<tr>
<td>Brampton</td>
<td>1435</td>
<td>96</td>
<td>109</td>
<td>13</td>
<td>4</td>
</tr>
<tr>
<td>Caledon</td>
<td>1079</td>
<td>65</td>
<td>26</td>
<td>42</td>
<td>7</td>
</tr>
<tr>
<td>Mississauga</td>
<td>1127</td>
<td>108</td>
<td>118</td>
<td>10</td>
<td>5</td>
</tr>
</tbody>
</table>

Across the four municipalities, tree count, observed species richness and the number of plots vary. Tree count in Bolton is less than half what it is in the other three. However, it also has substantially fewer plots than do Brampton and Mississauga. Caledon, on the other hand, has just slightly fewer trees than Brampton and Mississauga and yet an equivalent number of plots to Bolton. This makes Caledon’s average tree density per plot much higher (between two and four times higher) than the other municipalities. Despite this, Caledon has the lowest number
of observed species. Brampton has the highest number of trees and the second highest number of species. Mississauga has the second highest number of trees but the highest number of species.

The ten most abundant species found in each municipality are listed in Appendix 6. *Thuja occidentalis* dominates both Bolton and Caledon and is a prominent component of Brampton and Mississauga’s tree composition also. *Rhamnus cathartica* is the most dominant species in Brampton and it also ranks high for Bolton and Caledon. *R. cathartica* comprises only two percent of Mississauga’s sampled trees, however. Mississauga is also unique in that its most dominant species is *Rhus typhina*, a species that is not found in the sample in any of the other three municipalities.

The species accumulation curves that were produced for the municipal samples (Appendix 7) indicate that for each municipality, sampling was much closer to being exhaustive than it was for many land use classes when broken down by land use type. Bolton and Caledon, having only 26 plots each, produced curving graphs (most similar to the park and commercial curves). Brampton (109 plots) and Mississauga (118 plots) produce species accumulate curves that come closest to reaching an asymptote. Therefore, diversity measures taken from these municipalities can be assumed to accurately represent the composition of tree species within them.

**4.4.11 ANOVA by Municipality**

The ANOVA process was repeated to test for significant relationships between species richness/tree abundance and municipality. The results (Table 10) illustrate that a significant relationships were found at the 0.05 level.
To further explain these relationships, alpha, evenness and beta metrics were, again, employed.

4.4.12 Alpha Diversity by Municipality

Table 11 provides the outcome of both alpha diversity indices and the evenness measure:

Table 10: ANOVA Results for Municipalities

<table>
<thead>
<tr>
<th>Null Hypothesis</th>
<th>Significance</th>
<th>Test Statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>The distribution of SR is the same across categories of municipalities</td>
<td>.018</td>
<td>1.49</td>
</tr>
<tr>
<td>The distribution of TC is the same across categories of municipalities</td>
<td>.015</td>
<td>-5.14</td>
</tr>
</tbody>
</table>

Significance level is .05

4.4.12 Alpha Diversity by Municipality

Table 11: Municipal Alpha and Evenness Values

<table>
<thead>
<tr>
<th>Municipality</th>
<th>H’</th>
<th>C</th>
<th>E</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bolton</td>
<td>3.483</td>
<td>16.33</td>
<td>0.2206</td>
</tr>
<tr>
<td>Brampton</td>
<td>3.332</td>
<td>11.42</td>
<td>0.1189</td>
</tr>
<tr>
<td>Caledon</td>
<td>2.604</td>
<td>5.731</td>
<td>0.0882</td>
</tr>
<tr>
<td>Mississauga</td>
<td>3.761</td>
<td>23.09</td>
<td>0.2138</td>
</tr>
</tbody>
</table>

H’ = Shannon-Weiner Index; C = Simpson’s Index; E = Simpson’s Evenness Measure

The results of the Shannon-Weiner index were higher on average than they were for the individual land use types. This is not unexpected as each municipality comprises a contiguous area composed of many different land use types. Therefore, relatively high within-community diversity makes sense. The H’ value for all the municipalities fall within the standard range expected from empirical data, as provided by Magurran (1988) with the exception of Mississauga, which slightly
exceeds the upper bound of 3.5. This might be explained by the fact that Mississauga contains the largest number of plots. Previous studies have found a positive relationship between land area and species richness (Angold et al. 2006).

Both H' and C rank the municipalities in the same order from greatest to least diversity: Mississauga, Bolton, Brampton, Caledon. Interestingly, Caledon, with the highest trees per plot density, consistently has the lowest alpha diversity and Mississauga, with the lowest trees per plot density has the highest alpha diversity. The number of species observed during sampling supports this finding. Caledon's urban forest may be more densely distributed but it is composed of a smaller variety of species whereas Mississauga's forest is more sparsely distributed and composed of a wider variety of species.

With respect to sampling intensity, the least thoroughly sampled municipalities, Bolton and Caledon, produced the lowest number of observed trees and species; however, the Bolton samples has the second highest alpha values and Caledon fourth. Therefore, sampling appears to have been sufficient throughout all municipalities to differentiate between their unique compositions.

4.4.13 Evenness by Municipality

Across all four municipalities evenness is very low. Bolton and Mississauga have the highest E values, though they are only 0.2206 and 0.2138. Caledon has lowest E value (< 0.1). Therefore, the proportional abundance of Caledon’s dense, non-diverse (in relation to the other municipalities, at least) urban forest very heavily favours a few species. The raw data supports this claim: \textit{Thuja occidentalis} comprises 37 percent of Caledon’s sampled trees, \textit{Fraxinus americana} comprises 17 percent, while 74 percent of Caledon’s tree species comprise only 10 percent of it’s total sampled population. Even Bolton, which has the highest evenness value of the municipalities has two species, \textit{T. occidentalis} and \textit{Populus tremuloides}, that comprise 31 percent of the entire sample population while another 72% of it’s species contribute only 10 percent.
For the most part, higher species richness seems to predict higher within
municipality diversity and evenness. Mississauga ranks the highest in these
categories (except for in evenness, where it ranks second), Bolton ranks second
(again, except for evenness, where it ranks first), Brampton is consistently third and
Caledon consistently fourth.

4.4.14 Beta Diversity by Municipality

The beta diversity between each pair of municipalities are listed in Table 12:

<table>
<thead>
<tr>
<th></th>
<th>Bolton</th>
<th>Brampton</th>
<th>Caledon</th>
<th>Mississauga</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bolton</td>
<td></td>
<td>48.02</td>
<td>37.93</td>
<td>44.97</td>
</tr>
<tr>
<td>Brampton</td>
<td></td>
<td></td>
<td>46.43</td>
<td>49.06</td>
</tr>
<tr>
<td>Caledon</td>
<td></td>
<td></td>
<td></td>
<td>51.11</td>
</tr>
<tr>
<td>Mississauga</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

The two municipalities with the greatest diversity between them are Mississauga
and Caledon; approximately half of their populations are mutually exclusive. The
two with the least diversity between them are Caledon and Bolton; only 38 percent
of their populations are mutually exclusive. Geographically, Mississauga is the
furthest municipality from Caledon and Bolton is the closest and it follows from
geographic principle that municipalities that are closer together will be more alike
than municipalities that are farther apart.
4.4.15 Tree Abundance by Land Use Type and Municipality

<table>
<thead>
<tr>
<th></th>
<th>Bolton</th>
<th>Brampton</th>
<th>Caledon</th>
<th>Mississauga</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agricultural</td>
<td>0</td>
<td>3</td>
<td>4</td>
<td>0</td>
</tr>
<tr>
<td>Commercial</td>
<td>3</td>
<td>3</td>
<td>6</td>
<td>3</td>
</tr>
<tr>
<td>Golf</td>
<td>0</td>
<td>39</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Institutional</td>
<td>6</td>
<td>4</td>
<td>12</td>
<td>20</td>
</tr>
<tr>
<td>Park</td>
<td>21</td>
<td>9</td>
<td>78</td>
<td>20</td>
</tr>
<tr>
<td>Residential</td>
<td>11</td>
<td>8</td>
<td>17</td>
<td>8</td>
</tr>
<tr>
<td>Transportation</td>
<td>80</td>
<td>6</td>
<td>0</td>
<td>8</td>
</tr>
<tr>
<td>Vacant</td>
<td>29</td>
<td>27</td>
<td>108</td>
<td>20</td>
</tr>
</tbody>
</table>

Table 13 displays the distribution of the average number of trees per plot across land use type per municipality. Some interesting distributional patterns can be observed from this breakdown. Vacant land possesses the most densely populated plots (108 stems/plot) in Caledon, while the other three municipalities have vacant land with medium tree densities. The most sparsely populated plots in Peel exist on commercial land in Bolton, Brampton and Mississauga, and on agricultural land in Brampton. However, even in Caledon, commercial land is very sparsely populated. Residential land exists at consistent (medium) densities across all land use types. Park land and institutional plots vary across municipalities. Overall, the table illustrates inconsistencies in the distribution of trees across land use types and municipalities and therefore, the importance of analyzing species composition and distribution with respect to both components.

The various measures of diversity are distributed differently across municipalities than they are across land use types. Municipalities adhere to stricter rankings and there is far less mixing of order than there is with land use types in this study. This is understandable as municipalities cover distinct, isolated, and
contiguous areas, while land use types are dispersed across the Region of Peel in a much less uniform pattern. However, the range of H’ values is larger across land use classes (1.796 – 4.082) than it is for municipalities (2.604 – 3.761). This may indicate that large unified areas promote greater within community diversity on average, but does not account for the distribution of that diversity.

4.5 Discussion

The 179 unique species of tree that are found in Peel Region are not evenly distributed across land use types or municipalities. Because of this, certain patterns emerge that can be used to distinguish individual classes. On top of that, one single overarching trend can be observed that reflects the species composition and distribution of Peel as a whole, that is, very low evenness. Many of the trends observed in Peel replicate those noted in the literature, while some deviate from them. In this discussion, each land use type will first be considered individually and then as one cohesive region.

4.5.1 Agricultural Land

Structurally and compositionally agricultural land is the most unique land use class identified in this study. With respect to tree density and number of species on a per plot basis, agricultural land has the lowest value, which is consistent with the literature. For example, Wania et al. (2006) found in a study of plant richness in Halle, Germany that rural areas experienced lower tree abundances and species richness than did their urban counterparts. They attributed this trend to the high number of exotic species that exist in urban settings that are absent in agricultural settings, and the fact that native tree counts were also higher in the urban settings due to the increased heterogeneity of those areas. It is likely that both of these factors are contributing to the low species richness and tree abundance in Peel’s agricultural land uses. Because of Peel’s location within Canada’s largest urban centre, the General Toronto Area (GTA), the agricultural land that persists within the Region is likely to be intensely managed resulting in controlled tree populations
and little chance that natural variability in the landscape has been left unaltered and un-homogenized.

4.5.2 Commercial

Commercial land as a specific land use designation is not considered in the literature to a great extent. One study that does examine commercial land use was conducted by Sudha and Ravindranath (2000). They found that in Bangalore City, India, commercial land has a species richness of 104 species over 46 plots and Shannon-Weiner index of 1.549. They observed very low tree density and noted that within commercial areas trees are primarily confined to very specific areas, for example, along sidewalks or in small squares. While the results from Peel’s commercial areas produced only 43 species and a much higher alpha diversity value (H’ = 3.388), a very low tree density was consistent with the results from Bangalore. Commercial land in Peel is the second least dense land use type of the eight examined in this study.

In another study that considered commercial land, Zipperer (2002) found that most commercial areas in Syracuse, New York were associated with remnant, rather than regenerated, patches. It is likely that Peel’s commercial land also shares this characteristic as it supports the fact that it has the highest evenness level of all the land use types. If Peel’s commercial land is maintaining the remains of an older tree population (established under a different land use), within a spatially constrained area, it is not surprising that it has high evenness. There is not an opportunity for any of the remaining species to dominate the landscape and there is very little room for the recruitment and propagation of new, more aggressive species. The data support this; commercial land’s most abundant species, Picea pungens, has only twelve trees. The other 42 species must all have populations between 1 – 11, therefore, the evenness of the community is high.
4.5.3 Golf

Extreme differences in the abundance of trees in plots on golf courses can be inferred somewhat from the function of the land use. Trees are concentrated into narrow and restricted areas to allow for uninterrupted, open playing space. However, the results of this study indicate significant clumping of trees by species within those restricted areas, evidence of the manicured nature of golf courses. With respect to diversity, golf courses possess the lowest H’ and C values of all land use types. This is counter to the findings of Tanner and Gange (2005), which indicated that biodiversity was greater on golf courses than on surrounding farmland. However, the diversity measures for golf courses in this study were significantly hampered by under sampling, future work should more closely examine Peel’s golf course tree diversity.

4.5.4 Institutional

The distribution of tree abundance across institutional plots is bimodal; plots are either relatively dense (containing greater than 20 Pinus resinosa, Fraxinus americana or Acer saccharum trees per plot) or relatively sparse (containing fewer than 6 trees per plot, not dominated by one species in specific). This trend likely speaks to the planned nature of institutional land whereby trees are organized to serve either one discrete function or another. Jim and Liu’s (2001) survey of institutional land use sites in Guangzhou City, China support this finding. Their results indicated a large degree of variation between species composition, which was contributed to by site age (younger institutional sites contained more trees), area (larger sites contained more trees) and tree frequency (the greater the number of trees, the greater the number of species). Because both age and area are outside the scope of this project it may be speculated that these are controlling factors in Peel’s institutional lands, although no concrete conclusions may be drawn.

With respect to within community diversity, between community diversity and evenness, institutional land ranks toward the middle of the pack. These results
speak to the adaptability of institutional land; it can serve multiple functions and accommodate a variety of species, however, it rarely does either of these to any exceptional extent. This is likely because the principle function of institutional land is served by the buildings that are on the site and the trees that are placed around them serve a secondary purpose.

4.5.5 Park

Because the vast majority (88.5\%) of Rhus typhina is isolated to a single locality within Peel, it should not be assumed that it is a recurrent component of the landscape in Peel or even in park land despite it being the fifth most abundant species in Peel. Whether or not to consider Rhus typhina a tree or a shrub has been debated (Meagham Eastwood, TRCA, personal communication). It was decided for this study that it should be considered a tree because of its interesting distributional pattern, the fact that it commonly grow to heights of 6 metres (Farrar 1995) and its classification as a tree in many reference guides (Hosie 1979; Farrar 1995). However, when considering the density of the species compared to others, it should be taken into account that in reaching maturity, stems may be substantially smaller than their maximum thickness of 10 cm (Farrar 1995).

This pattern of unevenness in the distribution of a single species repeats itself for both the second and third most abundant park species as well, resulting in an extremely low evenness measure. This result is much lower than the evenness measures obtained by Jim and Chen (2009) for Taipei’s urban parks, who found values three to six times higher (0.4 to 0.69). This may be reflective of a difference in the sampling methods employed in both studies; Jim and Chen’s results are based on a complete census of Taipei’s urban trees rather than on sampling methods.

The Shannon-Weiner values obtained for urban parks by Jim and Chen (H’ = 3.493) are more congruent with the results of this study (H’ = 2.606). However, the value for Peel is still lower than would be expected given the well documented trend
for the natural heterogeneity of activities within parks to yield high diversity values (Jim and Chen 2009; Jim and Liu 2001).

4.5.6 Residential

The alpha indices indicate that residential land is the most diverse land use type, which is supported in the literature. This is likely due to the variety of social factors that do not affect the other land use types to such a degree (Troy et al. 2007). This contributes to high numbers of exotic species, as possessing unique and rare species is often used as a way of showcasing wealth or specific cultural values. It also results in a high level of unevenness as certain species are clearly favoured by many households according to local trends.

Many other studies consider the effect that private, domestic gardens have in shaping the species composition of residential land. For example, in their survey of residential gardens in Sheffield, UK Smith et al. (2006) concluded that gardens contain high levels of species richness (they observed 90 tree species, which is far surpassed by Peel’s 145 residential species), a high percentage of exotics, and are strongly shaped by the behaviour of their owners.

Residential land is also the land use type that is likely to reflect cultural diversity most directly, as homeowners are able to choose tree species according to their personal tastes and values. Especially within the GTA, a strongly multicultural area, this diversity is likely to be expressed.

On the other hand, residential land has very low evenness (E = 0.1989). This finding is also supported by the literature; in Bangalore City, Sudha and Ravindranath (2000) found that the ten most dominant species in residential land comprised 58% of the tree population. In Peel, this value is 44%, still nearly half the entire sampled population. Inequitability can be observed in the distribution of abundance across all the species in residential land and in the distribution of abundance across residential plots. The interesting relationship between residential land’s high diversity and low evenness is considered in much greater detail in
Chapter Five, by examining how several socioeconomic factors contribute to shaping this pattern.

4.5.7 Transportation

Transportation is ranked third in the trees per plot and species per plot categories. The literature predicts that tree communities in transportation land use have high species richness as a result of high levels of exposure to sunlight and drainage (Forman and Alexander 1998). However, in Peel, tree communities in transportation land were found to have mid to low alpha values and low evenness. This may be a result of the fact that trees within transportation land uses, most often along roadsides, experience a unique group of additional stressors. Such stressors may be attributable to salt, sand and nutrients from road dust (Forman and Alexander 1998), as well as to the direct effects of road salt, heavy metals and the microclimate (Forman and Deblinger 2000).

4.5.8 Vacant

Vacant land in Peel possesses mid range alpha diversity values, but has the highest ranking in both the average species richness per plot and average tree abundance per plot categories. These high rankings are contrasted by the results of a study conducted in Beijing by Zhao et al. (2010) but they are supported by the results of the UFORE Model applied to Chicago (Nowak et al. 2010). Zhao et al. found evergreen and deciduous trees in vacant land to have the lowest relative species richness compared to park, protective (deciduous trees only), institutional, residential and street green space. However, Nowak et al. (2010) found their “open space” land use category to rank first with respect to both number of trees per area and tree leaf area index. Zhao et al. explain that such differences between findings in Chinese and North American urban areas may be attributable to differences in management. They emphasize the importance that is placed on ornamentation in Beijing, which leads to large areas being covered in single, decorative species. Whereas in North America, such areas are more likely to be allowed to pursue more
natural succession. Vacant land in Peel Region should be expected then to fit in with the North American model, and that is what this study finds.

4.5.9 Municipalities

Despite being located within the same upper-tier municipality (Peel Region), each of the four constituent municipalities that were examined in this study are subject to slightly different environmental factors. Mississauga is unique amongst the other municipalities in that it directly borders Lake Ontario, as well it is subdivided by a major highway. Both of these components act as conduits for new species. It is therefore not surprising that Mississauga experiences the largest alpha diversity of any municipality. However, both of these elements also expose the surrounding tree communities to a variety of harsh environmental effects including higher wind speeds, colder temperatures and more direct exposure to pollutants. The culmination of these effects may be responsible, in part, for the fact that Mississauga experiences the lowest average number of trees per plot of any municipality.

Conversely, Caledon experiences the lowest alpha diversity values but the highest average number of trees per plot. Of the four municipalities, Caledon exists in the most rural setting and it therefore buffered somewhat from the environmental stresses associated with high levels of development, traffic and more direct effects from the city of Toronto and Lake Ontario.

Conway and Hackworth (2007) examined the NDVI of transects, two of which run through the municipalities in this study. They observed NDVI increase along these transects as distance from the urban centre increased. This supports the position that Bolton and Caledon experience higher average plot densities because of their geographic location. However, Conway and Hackworth found only a weak correlation between NDVI and population density. Based on this, it cannot be concluded that population contributes significantly to the distribution of tree abundance between municipalities though this is a tempting relationship as
population size is also disparate between the Bolton/Caledon and Brampton/Mississauga pairs.

The form and pace that urban growth has taken within each of the municipalities are also likely to have influenced the noticeable differences between urban forest composition and structure. As will be expanded upon in Chapter Five, neighbourhood design pattern and development age have both been found to influence tree density and distribution (Southwood and Owens 1993; Troy et al. 2007). Therefore, urbanization patterns appear to be the controlling factor determining tree abundance, and urbanization can be directly linked to land use.

4.5.10 Large Scale Patterns

All of Peel Region’s urban tree communities, irrespective of scale, can be characterized as having one thing in common: low evenness. This has been observed within land use types, within municipalities and across the total population.

One effect that this widespread unevenness has in conducting surveys of distinct communities (for example, land use types) is that it emphasizes the importance of using an alpha diversity statistic. This is because the alpha statistic encompasses both species richness and evenness. A simple measure of species richness would provide an incomplete picture with no insight into how those species relate to each other.

There is a global concern that as the structure of cities become increasingly similar around the world, urban biodiversity will begin to reflect this uniformity. If all urban environments are dominated by the same invasive species, gradually native species and local habitats will become obsolete and global diversity will be negatively affected in a powerful way (Lundholm and Marlin 2006). Fortunately, the results of this study do not suggest that Peel has become subject to global homogenization. Studies from multiple urban centres around the world --- i.e. Syracuse, New York (Zipperer 2002), Taipei City, Taiwan (Jim and Chen 2009), Beijing, China (Zhao et al. 2010), Bangalore City, India (Sudha and Ravindranath
2000) --- were reviewed. While trends in diversity and evenness were shared between many geographically distant cities, specific species profiles were not. Encouragingly, native species comprise the majority of the ten most abundant species for each land use type in Peel (Appendix 3). They make up 100% of the top ten species on institutional land, 90% on golf and vacant land, 80% on agriculture, transportation and park land, 70% on residential land, and 40% on commercial land.

4.6 Conclusion

Based on the findings that there are a large number of species that exist in very low numbers in Peel Region and that there are significant differences between the species compositions of distinct land use types, four management strategies should be employed.

The first strategy should be aimed at increasing the evenness of species within each land use type and across the entire region so that diversity and robustness are promoted by maintaining high population levels of many (if not all) species. In planting new trees, a wide variety of species, rather than a few popular species, should be encouraged. As well, more native species have to be introduced. This is important for several reasons. For one, it creates an urban ecosystem that is more reflective of the true diversity of the natural habitat. As well, high evenness values that correspond with high abundance provide the urban forest with some level of insurance against stochastic events. For example, in the case of pest (i.e. the emerald ash borer) infestation, an urban forests with a greater number of stable species populations is less likely to be devastated by the outbreak (Alvey 2006). A higher number of species means a larger percentage of them are likely to be unaffected by the pest - or similarly, by disease - and higher populations within each species suggests that even those effected are less likely to become locally extinct.

The second management strategy should allow vacant land, parks, and other unexploited areas, such as some transportation corridors, to naturalize. In his study
of regenerated and remnant patches in Syracuse, New York, Zipperer (2002) emphasized the importance of having regenerated areas in urban settings. Peel would also benefit from allowing certain, vacant areas to re-grow naturally, without any direct management. While this no-management approach could not be applied to the Peel region as a whole, Zipperer did stress that as this form of growth does not expend energy, there are ultimately net benefits associated with it, and it may contribute positively to certain areas in Peel. For example, consideration should be given to empty plots to allow them to develop this way rather than by immediately clearing them for further construction. Predominantly passive, re-naturalizing plans with minimal human interference have been found to increase species richness in Osaka, Japan and increase native species in Christchurch, New Zealand (Alvey 2006).

Another potentially beneficial management option involves a shift in the way tree functions are viewed. Rather than allocating tree planting sites and choosing species based on only a single functional consideration, planting practice should account for the multiple functions that trees can fulfill. Sudha and Ravindranath (2000) concluded that Bangalore City in India would benefit from increased planting of trees that have multiple functions. The functions they placed heavy value on were production of edible fruit, flowers, and leaves. There is no reason why the same principal cannot be applied in Peel Region. The specific species of trees planted in each locality should be considered in terms of their ability to provide optimal shade for inhabitants, habitat for fauna, and insulation for neighbouring buildings. As well, with an increasing movement toward harvesting fruit and vegetables grown in urban environments developing in the GTA, there is no reason why food production should not be a consideration in species selection.

Finally, the importance of conducting long-term studies should be integrated into Peel’s management plan. Jim and Chen (2009) reflect that maintenance of regular, long term study of urban environments is crucial. The same holds true for Peel. Consistent monitoring and surveying of the success of its urban forest and the
distribution of its trees should be carried out. This thesis has now provided a baseline species assessment against which future studies can be compared.

The next step that should be taken to build on the results of this study, is to take a much more comprehensive survey of the species and abundances of Peel Region’s trees. The UFORE data used in this study was exceptionally useful, it did however lack exhaustive sampling in all land use types. Because of the limited number of species-specific studies that exist at the moment, performing this baseline analysis without waiting for more data collection was justified. However, in order to obtain a more veracious representation of Peel’s urban forest, a much more thorough sampling effort is needed.

Addressing the needs and restrictions of each individual land use type should be an essential component of Peel’s urban forest management. However, as advised by Jim and Liu (2001), balance must be struck between addressing needs on the small scale while also applying a comprehensive, city-wide management plan, which they deemed imperative.

It is obvious from this data that the land use type with the greatest contribution to the diversity of trees in Peel is residential land and that this is largely the result of decision making taking place at small scales. That being true, increasing public awareness of the benefits of planting native, and land use-appropriate tree species is going to be an essential part of strengthening Peel’s urban forest.

Ultimately, Peel’s goal should be to increase its total abundance of trees within which, both evenness and diversity among species should be optimized. Thereby, the region’s urban forest will provide the most benefit to its citizens in terms aesthetics, biodiversity, sustainability and ecological function.
Chapter Five
The Influence of Neighbourhood Socioeconomics and Urban Form on Tree Diversity

5.1 Introduction

Previous studies have identified that the population density and socioeconomics of a region have significant impacts on its species richness and abundance (Hope et al. 2000; Iverson and Cook 2000; Sudha and Ravindranath 2000; Pickett et al. 2001; Conway and Hackworth 2007; Troy et al. 2007; Luck et al. 2009). While these impacts vary between locations with respect to the strength and direction of their influence, some common patterns have emerged within the literature. For example, it has generally been found that income and tree abundance share a positive correlation (Landry and Chakraborty 2009), while presence of renters and minority dominated neighbourhoods tend to have few trees and species.

This chapter will consider the ways in which neighbourhood socioeconomics and urban form are related to urban tree species richness and abundance. The socioeconomic factors and most of the urban form variables that are included in this analysis are inherently tied to housing and will therefore be restricted to UFORE data gathered within residential plots. Since residential areas are often the most diverse components of the urban landscape, it is possible to identify the cause of much of the variety that is observed in the urban forest by focusing on residential areas. In total, this study examines the relationship between nine socioeconomic and urban form factors and the distribution of tree species in Peel. The effect that roadways have on urban tree richness will also be examined over the entirety of Peel Region however, as roads are ubiquitous throughout the urban landscape.

Given Peel’s location within the Greater Toronto Area, one of North America’s fastest-growing urban areas (Conway and Hackworth 2007), it is important that the socioeconomic controls over Peel’s urban forest are understood
now to help develop policy and management plans on private property that will impact current and future urban locations.

5.2 Literature and Background

Several studies have looked at the implication that socioeconomics have on urban vegetation. Many of these studies identify significant disparity within the distribution of trees across economic gradients by considering total canopy cover (Perkins et al. 2004; Landry and Chakraborty 2009) or tree count (Pedlowski et al. 2002). The findings of such studies suggest that tree abundance --- and therefore the host of environmental, economic, social and spiritual benefits imparted by trees --- are heavily biased toward neighbourhoods with high levels of income, property value and housing ownership. These studies also find significant evidence that these economic factors as well as the level of neighbourhood tree abundance are deficient in neighbourhoods comprised predominantly of racial minorities (Landry and Chakraborty 2009).

Other work has produced findings that indicate that elements of urban form, such as neighbourhood age, are determining factors in the composition and structure of tree species (Sudha and Ravindranath 2000). The analyses of urban forests around the world have echoed these results, yet other studies have found exceptions to some of the expected trends. These examples will be considered in this section.

The majority of studies that investigate these relationships consider residential areas at a scale somewhere in between the level of the land use type and the individual housing unit. For example, studies have been scaled to the level of building parcel (Troy et al. 2007), neighbourhood (Pedlowski et al. 2002), and right-of-way segment as defined by census block boundaries (Landry and Chakraborty 2009). In order to attribute socioeconomic data to these units, national census data is often used, for example from the US Census Bureau (Landry and Chakraborty
2009), the Australian Bureau of Statistics (Luck et al. 2009), or Statistics Canada (Conway and Urbani 2007).

By conducting analyses that incorporate aspects of citizen’s economic and social realities, urban forest studies are able to address issues raised by political ecology of urban social equity. Such studies therefore carry direct implications for improving the quality of life in areas where they identify ecological deficiencies. As well, understanding the social drivers that shape the species richness and tree abundance of a residential area aid in the effectiveness of reforestation efforts (Perkins et al. 2004).

5.2.1 Average Household Income

Much of the research that considers the influence of household income on urban vegetation points toward the existence of a “luxury effect” (Hope 2000). Troy et al. (2007) use a similar term, “ecological prestige,” to describe the phenomenon. Both of these phrases allude to a strong relationship between wealth and vegetation abundance. In theory, residents with the financial freedom to choose where they would like to live tend to gravitate toward more verdant neighbourhoods (Grove et al. 2006). Within these neighbourhoods, landscaping becomes a marker of social status, further driving people’s preferences for certain types of vegetation. This driving force creates interesting patterns in the composition of residential flora.

Overwhelmingly, the results of studies that looked to confirm the presence of a luxury effect occurring in residential neighbourhoods were able to do so; increases in both tree abundance and species richness have been observed with an increase in average household income (Iverson and Cook 2000; Luck et al. 2009). The trend is not ubiquitous across all urban landscapes however. In their study of Baltimore, Maryland, Grove et al. (2006) found social stratification based on income to be the worst of seven models for predicting tree abundance. The best results were obtained from a model that combined lifestyle behaviour factors with median housing age. From this, they concluded that income levels alone are not sufficient
for expressing the relationship between urban vegetation and socioeconomics and that factors that contribute to the luxury effect cannot be ignored.

5.2.2 Population Density

Intuitively, it might be expected that human population density and tree population density strongly oppose each other, however, this is seldom supported by the literature. Based on previous studies, population density does not exert a strong or consistent influence over tree cover. Using remote sensing imagery, Iverson and Cook (2000) found a positive correlation between tree abundance and population density in Chicago. When Luck et al. (2009) investigated the relationship between population density and species richness in nine towns in southeastern Australia they also found a strong positive relationship. Meanwhile, in Central Indiana, Heynen and Lindsay (2003) found no relationship for vegetation abundance, nor did Conway and Hackworth (2007) in Toronto, Canada. However, in Baltimore Maryland, Troy et al. (2007) observed a negative relationship between population density and tree abundance.

Conway and Hackworth suggested that the results from their study indicate generally uniform population density levels within residential land use, and sharp contrasts between these levels and those found in nearby rural areas. Therefore, it is possible that urban tree density is actually more strongly related to land use type, independent of the population level of that land. If this is the case then it should be expected that the data from Peel will not necessarily indicate a positive relationship between population density, species richness and tree abundance. Additionally, population density and tree abundance may both independently be a function of land use type rather than tree abundance a function of population density.

5.2.3 Owner Occupied Houses

Percentage of homes that are occupied by owners, rather than by renters, has been examined occasionally in the literature. Often, this variable is considered to be just another indicator of wealth, and therefore is not considered separately from
other measures such as income. However, some studies have examined the difference in vegetation between houses inhabited by owners versus renters. Conway and Hackworth (2007) found a strong positive relationship between the percentage of owner occupied houses and vegetation abundance in one of their four urban-rural transects running through the GTA. Not surprisingly, this transect was also the one in which the “luxury effect” was most obvious. Luck et al. (2009) observed a positive correlation between owner-occupation and tree cover in Australia. While Troy et al. (2007) found no relationship between percent of owner occupied houses and tree abundance. They speculate that the lack of relationship exists for trees as a result of what they refer to as a “legacy effect,” the consequence of the lag time between the planting of trees and the time it takes for them to reach maturity and therefore, maximum canopy size.

5.2.4 Housing Structure Type

Another variable that has been examined is the percentage of houses that are single-family residences. This measure can also be intuitively interpreted as a proxy for wealth, however, it carries with it some indication of population density as well. In Baltimore, Troy et al. (2007) found the percentage of houses within a residential census block group that are classified as single-family residences is positively correlated with tree cover. They posit that this finding can be explained by “lifestyle theory,” which indicates that the freedom to choose to live in a detached, single family house comes as a result of wealth. It is similar in this way to the demonstration of affluence through landscaping preferences that is summarized by the “luxury effect.”

5.2.5 Housing Age

The study by Troy et al. (2007) is also one of the few to directly consider the effects of housing age on the surrounding tree distribution. Their findings indicated that housing age is quadratically related to the amount of realized stewardship a property offers. From this finding they inferred that older houses (the upper limit of
which was 89 years) are likely to be situated in what are now, but what was not then, city centres. As a result of high urbanization pressures and high infrastructure densities it is to be expected that these neighbourhoods are tree-poor. At the other end of the spectrum, newly built houses often exist at the edge of suburban sprawl and therefore encroach onto farmland which was devoid of prominent tree cover as a function of its earlier land use type. They also propose that these new properties have not yet had the time to establish successful tree communities. The optimum housing age that maximized tree cover they found occurs at 46 years.

Hope et al. (2003) also considered the effects of housing age. Interestingly, their study produced opposite results for two different residential settings, urban and desert. In the urban area of Phoenix, Arizona they observed that younger houses have higher plant diversity than do older houses. In the surrounding desert landscape, they observed that younger houses have lower plant diversity than do older houses. They used these findings to emphasize the varying mechanisms that affect different landscapes, thereby illustrating the value in performing location-specific analyses. Hope et al. attributed the trend in urban areas to developments in landscaping design, technology and cultural values over time. They presumed that the trends in desert areas were a consequence of environmental factors and that the effects of extreme environmental stress were limiting the ability of plants to persist. Though this study was not restricted to tree species, it is likely that the general conclusions it produced would apply to trees also.

5.2.6 Roads

When investigating the relationship between road networks and urban tree species richness and density, an important factor to consider is the extent of influence that the road imparts on the surrounding ecosystem. The concept of a “road-effects zone” (Forman and Alexander 1998) has been investigated in previous studies (Forman and Alexander 1998, Forman 2000, Forman and Deblinger 2000). The basic principle that is encompassed by the “road-effects zone” is that trees surrounding roads will be impacted to various degrees based on their proximity to a
road. The direct effects that roadways have on trees include severe exposure to the elements, poor and potentially compacted soil, physical damage (Jim and Chen 2008), increased levels of air pollution such as ozone, sulfur dioxide, nitrogen dioxide and carbon monoxide (Nowak et al. 2008), as well as exposure to road salt and heavy metals (Forman And Deblinger 2000).

Some of the chemical and atmospheric pollutants that come from roads are carried substantial distances from their source point by wind and ground water; Forman and Deblinger (2000) observed that the effects of road salt leaching into the water table may extend up to 1500m from roadways. While the effects of these substrates are diluted by distance, they may still be a detriment to tree growth even in small quantities. The extent to which the road network of Peel exerts an influence on the region’s urban trees is important to identify as increasing urbanization will undoubtedly lead to an increase in road density.

In the literature, these social, economic and housing form variables are used to discern the cause of vegetative patterns in residential areas, primarily focusing on total canopy cover. In this study they will be used similarly, but to explore species richness and tree abundance in Peel’s residential areas.

### 5.3 Methods

Moran’s I was applied to the UFORE data to test for spatial autocorrelation. This test was required to identify if levels of species richness or tree abundance are distributed within Peel in some spatial pattern that should be accounted for in the following analyses. Results produced by calculating Moran’s I range from 1.0 (indicating total clustering of data points) to -1.0 (indicating total dispersion of data points). A Moran’s I value of 0 indicates a random spatial pattern. Z-scores associated with the results that are above 1.96 or below -1.96 indicate that spatial autocorrelation is significant at the 0.05 level.

Nine socioeconomic and urban form variables were analyzed to test for meaningful relationships between their values and the values of species richness
and tree count. The variables (Table 14) were chosen based on previous studies examining correlates of vegetation abundance and diversity. Data pertaining to income levels, population distributions, and housing trends were obtained from Statistics Canada’s 2006 census at the level of the dissemination area (DA). DA values were used because they provide the highest resolution census data available. Each DA contains between 400-700 people. When the total population of an apartment building surpasses 300 people, a new DA is established within the original DA. The minimum area comprised by any DA is one block.

The UFORE data was overlaid against a map of the DA boundaries using ArcMap and all residential UFORE plots were assigned to a single DA. It was possible for more than one plot to fall within a one DA, however each plot could only be associated with a single DA. The values of each of the eight socioeconomic variables were then attributed to each plot. For variables that are expressed in percentages in this study, such as percentage of owner occupied houses, the Statistics Canada data provided both the total number of houses for each DA as well as the number of houses within each DA that were owner occupied, the percentages were calculated from these data to make plots that fell within DAs possessing different total numbers of houses comparable.

The road data used to calculate the minimum distance from each plot to the nearest road was obtained from DMTI Spatial’s CanMap® Route Logistics Ontario v.4.0 product suite (2009). The UFORE data was again overlaid against the road network map in ArcMap (Appendix 2) and the minimum distance was calculated for each of the 279 plots across Peel using a proximity function.

Species richness and tree count were based on the 146 UFORE plots for Peel that are residential.
Table 14: Socioeconomic and Urban Form Variables

<table>
<thead>
<tr>
<th>Predictor Variable</th>
<th>Data Source</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Percent of residences that are owner occupied</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Percent of residences that are apartments greater than five stories</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Percent of residences that are single family</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Average household income</td>
<td>Statistics Canada 2006 Census</td>
<td>Dollars (CAN)</td>
</tr>
<tr>
<td>Population density</td>
<td>Statistics Canada 2006 Census</td>
<td>People per kilometer squared</td>
</tr>
<tr>
<td>Percent of residences that were constructed pre-1946</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1946-1980</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1980-2006</td>
<td>Statistics Canada 2006 Census</td>
<td>Percent</td>
</tr>
<tr>
<td>Distance to nearest road</td>
<td>DMTISpatial 2009</td>
<td>Meters</td>
</tr>
</tbody>
</table>

The next analysis was a simple correlation using Spearman’s Rho. This correlation coefficient was chosen because the data is not normally distributed. The analysis was conducted in SPSS. The output from a Spearman’s test ranges from 1 to -1, where a value of one indicates a perfect positive correlation between two variables, a value of negative one indicates a perfect negative correlation, and a value of zero indicates no that discernable relationship exists. Correlation values were determined between all species richness and tree abundance values, between each socioeconomic, urban form and road distance measure and species richness and tree abundance, as well as between road distance and species richness and tree abundance when road distance was broken down by land use type.

Because several socioeconomic variables in Peel likely act simultaneously to determine the species richness and tree count for every dissemination area, the relationship of these factors together was also examined. Thus, a generalized linear
model (GLM) was used to investigate the relative contribution of each variable with respect to species richness and tree count. This method of analysis was chosen because of the non-normal distribution of the data. Specifically, a Poisson distribution was used within the GLM because it is most appropriate for count data that are constrained by a minimum value of zero (Hutcheson and Sofroniou 1999). The analysis was conducted in SPSS using the GLM analysis tool and a log link function.

5.4 Results

5.4.1 Spatial autocorrelation

The results for Moran’s I clearly indicated that for plots within residential land use, neither species richness nor tree count are spatially autocorrelated when separated by municipality. Tree count in Bolton produced the value furthest from 0 (0.059), as well as the only value to be deemed significant, but it is still a strong indication of random dispersion. When all the residential plots were considered together irrespective of municipal boundaries, the autocorrelation for species richness was slightly negative (Moran’s I = -0.008), while for tree count it tended slightly positive (Moran’s I = 0.015). This again suggests that no significant spatial autocorrelation exists for the UFORE plots on residential land uses in Peel. When Moran’s I was applied to all the plots in Peel, regardless of land use type, the data also indicated that both species richness (Moran’s I = 0.065) and tree count (0.073) are randomly distributed; these results were significant at the 0.05 level. As a result, the regression analysis did not need to account for spatial pattern.

Moran’s I was also used to test for autocorrelation within the distribution of land use types amongst plots. The results were significant at the 0.05 level and indicated that the dispersion is almost entirely random (Moran’s I = 0.013). Similarly, no autocorrelation was found amongst the distance of each plot to the nearest road (Moran’s I = 0.068).
5.4.2 Testing for Correlation Between Predictor Variables and Species Richness and Tree Abundance

Before socioeconomic variables were considered, Spearman’s Rho was used to investigate the relationship between the species richness and tree abundance of all the plots in Peel. The results indicate that a very strong positive relationship exists (Spearman’s Rho = 0.968, significant at the 0.01 level).

However, when the socioeconomic and urban form variables were considered, the results of the Spearman’s correlation indicate that none of the predictor variables alone is strongly correlated with species richness or tree count (Table 15). The distance to nearest road measure has the strongest correlation to both factors with mid-to-low negative values of 0.278 and 0.235. Percent of residences that are single family has the second highest correlation to species richness with a slightly positive value of 0.202. Also with a slightly positive value (0.219), average household income has the second strongest correlation to tree count. These variables are the only ones deemed to be significant at the 0.05 level.

Table 15: The Correlation Between Predictor Variables and Species Richness and Tree Abundance

<table>
<thead>
<tr>
<th>Predictor Variable</th>
<th>Species Richness</th>
<th>Tree Count</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Spearman’s Rho</td>
<td>Sig.</td>
</tr>
<tr>
<td><strong>Average Household Income</strong></td>
<td><strong>0.199</strong></td>
<td><strong>0.016</strong></td>
</tr>
<tr>
<td>Population Density</td>
<td>0.001</td>
<td>0.993</td>
</tr>
<tr>
<td>% Apartments Greater than Five Stories</td>
<td>-0.071</td>
<td>0.397</td>
</tr>
<tr>
<td><strong>% Single Family Residences</strong></td>
<td><strong>0.202</strong></td>
<td><strong>0.014</strong></td>
</tr>
<tr>
<td>% Residences Owner Occupied</td>
<td>0.135</td>
<td>0.104</td>
</tr>
<tr>
<td>% Built pre-1946</td>
<td>-0.018</td>
<td>0.826</td>
</tr>
<tr>
<td>% Built 1946-1980</td>
<td>0.102</td>
<td>0.222</td>
</tr>
<tr>
<td>% Built 1981-2006</td>
<td>-0.053</td>
<td>0.528</td>
</tr>
<tr>
<td><strong>Distance to Nearest Road</strong></td>
<td><strong>-0.278</strong></td>
<td><strong>0.000</strong></td>
</tr>
</tbody>
</table>

Values in bold were found to be statistically significant at the 0.05 level.
5.4.3 Modeling Each Predictor Variable’s Contribution to Species Richness and Tree Abundance

The original results generated by the GLM produced scaled deviance values that indicated that the Poisson distribution was overdispersed. One of the commonly noted limiting factors of Poisson regression is that it requires equitability between the variance and the mean (Hutcheson and Sofroniou 1997). When this equitability does not exist, as is often the case, overdispersion occurs. This is a very common symptom of geographical data and results from clustering within the data (Haberman 1978); it causes standard errors to be underestimated and makes the data vulnerable to type I error (assigning significance when it is not appropriate). To test for overdispersion the scaled deviance value is divided by the degrees of freedom. In theory, if the product is greater than one it indicates that overdispersion may be present. In this case, the product for species richness and tree count were 2.30 and 4.95, respectively, indicating that significant overdispersion existed.

There are two ways to correct for overdispersion. First, it is possible to use a negative binomial distribution rather than a Poisson distribution as it may better capture the dependant variable. However, here the results of the negative binomial distribution produced overdispersed results like the initial model. The Lagrange Multiplier test further supported the lack of improvement in moving from one distribution to the other. Therefore, the second correction method was applied. This involved changing the scale parameter to Pearson Chi-Square. The results of the model, run with the adjusted scale parameter, corrected the overdispersion.

The final output from the GLM for both species richness and tree count can be seen in Tables 16, 17 and 18.
<table>
<thead>
<tr>
<th></th>
<th>Species Richness</th>
<th>Tree Count</th>
</tr>
</thead>
<tbody>
<tr>
<td>Scaled Deviance</td>
<td>128.995</td>
<td>121.223</td>
</tr>
<tr>
<td>Degrees of Freedom</td>
<td>133</td>
<td>133</td>
</tr>
<tr>
<td>Omnibus Test (Likelihood Ratio Chi-Squared)</td>
<td>26.443</td>
<td>70.178</td>
</tr>
<tr>
<td>Significance</td>
<td>.009</td>
<td>.001</td>
</tr>
<tr>
<td>Predictor Variable</td>
<td>Test of Model Effects (Likelihood Ratio Chi-Squared)</td>
<td>Sig.</td>
</tr>
<tr>
<td>--------------------------------------------------------</td>
<td>-----------------------------------------------------</td>
<td>------</td>
</tr>
<tr>
<td>Percent of residences that are owner occupied</td>
<td>1.015</td>
<td>.314</td>
</tr>
<tr>
<td>Percent of residences that are apartments greater than five stories</td>
<td>1.364</td>
<td>.243</td>
</tr>
<tr>
<td>Percent of residences that are single family</td>
<td>1.348</td>
<td>.246</td>
</tr>
<tr>
<td>Average household income</td>
<td>.008</td>
<td>.928</td>
</tr>
<tr>
<td>Population density</td>
<td>.618</td>
<td>.432</td>
</tr>
<tr>
<td>Percent of residences that were constructed pre-1946</td>
<td>.456</td>
<td>.499</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1946-1980</td>
<td>.010</td>
<td>.920</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1980-2006</td>
<td>.000</td>
<td>.992</td>
</tr>
<tr>
<td><strong>Distance to nearest road</strong></td>
<td><strong>13.655</strong></td>
<td><strong>.000</strong></td>
</tr>
<tr>
<td>Residences within Bolton</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Residences within Brampton</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Residences within Caledon</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Residences within Mississauga</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Values in bold were found to be statistically significant at the 0.01 level
<table>
<thead>
<tr>
<th></th>
<th>Test of Model Effects (Likelihood Ratio Chi-Squared)</th>
<th>Sig.</th>
<th>Model Parameters (Wald Chi-Squared)</th>
<th>Sig.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Percent of residences that are owner occupied</td>
<td>7.562</td>
<td>.006</td>
<td>7.674</td>
<td>.006</td>
</tr>
<tr>
<td>Percent of residences that are apartments greater than five stories</td>
<td>6.667</td>
<td>.010</td>
<td>7.529</td>
<td>.006</td>
</tr>
<tr>
<td>Percent of residences that are single family</td>
<td>0.364</td>
<td>.546</td>
<td>0.365</td>
<td>.546</td>
</tr>
<tr>
<td>Average household income</td>
<td>0.465</td>
<td>.495</td>
<td>0.476</td>
<td>.490</td>
</tr>
<tr>
<td>Population density</td>
<td>1.642</td>
<td>.200</td>
<td>1.655</td>
<td>.198</td>
</tr>
<tr>
<td>Percent of residences that were constructed pre-1946</td>
<td>2.874</td>
<td>.090</td>
<td>2.219</td>
<td>.136</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1946-1980</td>
<td>0.035</td>
<td>.853</td>
<td>0.032</td>
<td>.857</td>
</tr>
<tr>
<td>Percent of residences that were constructed between 1980-2006</td>
<td>0.015</td>
<td>.902</td>
<td>0.014</td>
<td>.904</td>
</tr>
<tr>
<td>Distance to nearest road</td>
<td>33.654</td>
<td><strong>0.000</strong></td>
<td>34.627</td>
<td><strong>0.000</strong></td>
</tr>
<tr>
<td>Residences within Bolton</td>
<td></td>
<td>4.025</td>
<td><strong>0.045</strong></td>
<td></td>
</tr>
<tr>
<td>Residences with Brampton</td>
<td></td>
<td>0.086</td>
<td>0.769</td>
<td></td>
</tr>
<tr>
<td>Residences with Caledon</td>
<td></td>
<td>3.636</td>
<td>0.057</td>
<td></td>
</tr>
<tr>
<td>Residences with Mississauga</td>
<td></td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
</tbody>
</table>

Values in bold were found to be statistically significant at the 0.05 level
First, the omnibus test, which uses a likelihood ratio to test goodness of fit, was calculated for each model as a whole (Table 16). Both the species richness and tree count models were found to return significant results at the 0.01 level. Therefore, the GLM is reflective of real world conditions based on the nine socioeconomic and urban form variables that were used.

Each predictor variable was then considered individually (Tables 17 and 18). The output from the GLM produces results for the likelihood ratio-chi squared test and the Wald chi-squared test that reflect different characteristics of the model. The likelihood ratio acts as a goodness of fit measure between each predictor variable and the model by quantifying the effect that each variable has on the deviance of the model. As the likelihood ratio approaches zero, the model represents a better fit. The Wald chi-squared measure tests the null hypothesis that the predictor variable has no effect on the response variable. Therefore, statistics that are deemed to be significant by the Wald test indicate than those predictor variables do have an effect.

The only variable in the GLM (Table 17) for species richness to produce a significant relationship for the likelihood ratio and Wald statistic was “distance from nearest road.” In comparison with other likelihood ratios in the literature that were obtained from empirical data (Hutchenson and Sofroniou 1999), the value of this variable is relatively low and it can be assumed that its deviance is not particularly substantial. The result of the Wald chi-squared test indicates that the proximity of plots to the nearest road does influence their species richness. However, it can be concluded that none of the other drivers of species richness patterns within the urban residential trees of Peel Region can be adequately described by the variables used in this model.

The individual variable results of the GLM for tree abundance (Table 18) indicate that the variables for which the model and the data are a good fit are “percent of owner occupied houses,” “percent of apartments greater than five stories,” and “distance to nearest road.” These variables also have the highest likelihood ratio values, although “percent of apartments greater than five stories”
experiences less deviance than does “percent of owner occupied houses.” In comparison with the reference likelihood ratio values mentioned above, the values of these two variables are also relatively low and their deviance is slight. The deviance of the “distance to nearest road” variable is more comparable to average values. The Wald statistic also indicates that these three predictor variables are the only ones that are significant and therefore are the only ones that reject the null hypothesis. The Wald test also finds that the municipality of Bolton has a significant effect on the model of tree abundance. It can be concluded from these results that of the variables included in the model, the percent of owner occupied houses, percent of apartments greater than five stories, distance to nearest road, and being inside or outside Bolton are the only ones to significantly effect the tree abundance in Peel Region. Of those, only the first two are well described by the model.

5.4.4 The Impact of Road Proximity on Species Richness and Tree Abundance

The GLM output indicates that the distance between a tree stand and the nearest road significantly influences both the species richness and tree count of that stand. Therefore, further investigation is required to determine the nature of that influence. The correlation between the distance from the centre of each of the 279 plots in Peel to the nearest road and the species richness and tree abundance of those plots was investigated using Spearman’s Rho. The results indicated that a weak negative correlation exists between distance and species richness (Spearman's Rho = -0.278, significant at the 0.01 level) and the effect is slightly weaker between distance and tree abundance (Spearman’s Rho = -0.235). The median distance of all of Peel’s plots to the nearest road is 82 meters.

Interestingly, when road distance is correlated with the same factors but broken down by land use type (Table 19), the relationship only remains significant for three land use types, and for two of those, the sign changes.
### 5.5 Discussion

The results of this study reinforce two prevailing themes. The first is that tree abundance is positively tied to measures of wealth. The second is that road networks negatively influence species richness and tree count.

#### 5.5.1 No Indication of Spatial Autocorrelation

The results of the spatial autocorrelation analysis indicate that there is no spatial clustering of species richness or tree abundance values in the sample. This is a positive finding analytically because it suggests that there is no overarching factor (such as lake effects) that are imparting a region-wide gradient that could bias the data and detract from the ability to associate various levels of species richness and tree abundance to specific land use characteristics. Similarly, the distribution of land use types are randomly distributed across the landscape and do not indicate any location bias.
5.5.2 Little Correlation Between Species Richness, Tree Count and Socioeconomic Variables

Only three of the nine socioeconomic and urban form variables were found to be significantly correlated to tree species richness and abundance. These variables are: average household income, percentage of houses that are single family, and distance to nearest road. The urban form component (distance to nearest road) is discussed in the following section.

The results indicate that wealth, as represented by average household income, is tied to measures of species diversity and abundance. However, although the correlation analysis suggests that as wealth increases, so too do species richness and total tree count, this measure cannot provide deeper insight into the driving factors behind the connection. Therefore, the results of the GLM are used to shed light on the nature of this economic influence.

The ability to live in a single family home often comes as a result of financial success, therefore it might be assumed that the measure operates as a proxy for income. In this case study, however, there is not a strong relationship between percent of single family homes and average household income. This suggests that the significance of this measure is more likely an artifact of its urban form characteristics rather than its economic influence. As well, there may be cultural influences acting within this measure that would require more detailed analysis to tease out (Grove et al. 2006).

5.5.3 Distance to Roads is Correlated with Species Richness and Tree Abundance

The correlation analysis between distance to the nearest road and both species richness and tree abundance suggests a negative relationship for these variables across Peel Region. Therefore, tree diversity and abundance are highest close to roads. It is likely that this indicates that road networks in Peel are actually important dispersal and recruitment corridors for its urban tree species. Humans, animals, and vehicles may be inadvertently aiding the dispersal of tree species
across both short and long distances. As well, the large swathes of open land that roads create make natural dispersion and propagation more likely to occur. They also allow for high levels of sunlight to reach trees and for successful drainage networks, both of which benefit tree growth (Forman and Alexander 1998). Alternatively, people may be planting along road to create a visual buffer or as part of municipal urban forestry programs focused on street trees.

When each land use type is considered individually, distance to roads does not seem to influence species richness or tree count in plots for five of the land use types in Peel, however meaningful results were achieved for residential, commercial and golf land use types.

Within residential areas, a moderate positive relationship exists for both species richness and tree count. This is interesting because it opposes the negative trend that applies to the Peel plots all together. Several reasons for this positive relationship may exist. One may be that this reflects an absence of tree–lined or vegetated boulevards along residential streets. Another reason that species richness and tree count decrease closer to roads in residential land might be tied to the position of houses on their lots. Houses that are not set back far from roads may not allow for tree density and diversity due to lack of space or suitable planting conditions near the street. As well, houses that do have front yards very often limit that space to only grass. Therefore, the majority of tree species and stems are likely to be situated behind houses, in back yards as well as in ravines and parks that connect these areas.

Southwood and Owens (1993) have noted that house set backs have changed as neighbourhood design principles have changed over time and that the general trend for large set backs was most popular just after the middle of the twentieth century, a period during which 38 percent of the houses considered in this study were built. They also state that larger set backs have the effect of making neighbourhoods less pedestrian friendly, which may make tree abundance near street edges seem less important to residents. Southwood and Owens’ study also
found that in general, more recently built streets do not have street trees. They posit that this is because trees have increasingly become viewed as private property and therefore people are more likely to locate trees closer to the interior of their properties.

As well, many streets in Peel are organized so that backyards face inward toward the centre of the block thereby creating large areas of relatively similar surface cover. While property boundaries are often defined by fences and hedges, these may not be significant barriers for tree species and increased patch area has been shown to promote increased species richness (Honnay et al. 1999).

The correlation between distance to roads and species richness and tree count is negative for commercial land, like it is for Peel. This is not surprising because one of the principal functions of commercial land is to attract customers, therefore commercial developers likely care most about urban trees from an aesthetic perspective. Trees are likely planted close to streets in commercial areas because these are the only locations in which citizens have access to them.

Golf courses experience a very strong positive correlation between distance to roads and species richness and tree count. Again, this is not a surprising finding, given the design of golf courses. Large, open fairways comprise the majority of golf course land and these spaces must be largely contiguous, uninterrupted, and be easily accessible. Trees will naturally be located together, clustered within more remote areas of the course.

5.5.4 Regression Analysis Found Some Socioeconomic and Urban Form Variables to be Important Predictors of Tree Species Diversity and Abundance

Tree abundance in Peel reflects the distribution of wealth within the region, which suggests that Peel suffers from the some of the same social inequalities as do other cities. Specifically, the GLM indicates that neighbourhoods with more residents who own their houses are more likely to reap the benefits that come from living in areas with a high tree density. Therefore, residents who rent, and may not
be able to afford to own their houses, have unequal access to these amenities provided by the urban forest. The fact that lower income residents are also less likely to be able to afford other means of access into the natural environment (by taking vacations or through memberships to exclusive country clubs and recreational facilities, for example) only compounds the disparity between income levels.

The GLM also found that residing in Bolton is a predictor variable for a positive relationship with tree abundance. Interestingly, Bolton has the lowest average household income of all four of the municipalities examined in this study (Caledon = $124,666, Brampton = $100,862, Mississauga = $ 94,390, Bolton = $85,394). This indicates that economic factors are not the only controlling variables and some other factor must be influencing tree abundance in Bolton. This can likely be explained by the fact that Bolton is the most rural of the municipalities. Therefore, less of its land has been cleared for urban expansion and its urban forest still remains largely integrated with the surrounding natural forest.

Another of the measures to suggest a positive relationship with tree abundance, “percentage of apartments greater than five stories in height,” is also an encouraging sign that Peel holds potential for residents of lower incomes to live within highly verdant areas since average household income is negatively correlated with percentage of apartments greater than five stories (Spearman’s Rho = -0.455). As Peel residents who rent houses in low income neighbourhoods are the least likely to experience high tree abundance, high density housing provides an opportunity to mitigate this deficiency. It is however notable that some of the benefits that urban trees provide (such as reduced heating and cooling costs) are not applicable to apartment buildings.

In their study of the GTA, Conway and Hackworth (2007) found that canopy cover was positively correlated to average household income, which supports the findings here. Conway and Hackworth note, however, that high canopy cover may still exist while the land beneath remains highly impervious. Even though the areas
around apartment complexes may have a significant amount of leaf cover, if the complex itself is largely paved with concrete and sidewalks, as they often are, apartment dwellers may not experience maximized ecological benefits (for example, cooling effects or storm water runoff control) that house inhabitants with large amounts of pervious cover may.

Dissemination areas that have high percentages of residential units belonging to tall apartment buildings reflect areas where human populations are high but because urban densification exists through vertical growth, the surrounding land may be left available for urban forestry. This form of development holds significant potential in terms of increasing the exposure of lower income residents to urban vegetation, and in terms of increasing the total tree abundance of urban environments as a whole. Pursuing urban expansion through vertical infrastructure will enable municipal regions such as Peel to optimize both of these factors as population levels rise. The lack of restriction that human population density places on tree abundance and the positive relationship between percentage of tall buildings and tree abundance found in this study support this vertical form of development in Peel Region.

However, because the GLM does not directly reiterate the influence of average household income on species richness or tree count, as the correlation analysis did, it is possible that factors other than wealth are influencing vegetation patterns in Peel. For example, both the “percentage of owner occupied houses” and “apartments greater than five stories” variables may actually be more indicative of the influence that housing type has on the level of control that residents posses over their surrounding tree composition and abundance.

Residents of owner occupied houses are much more likely to have direct control over tree planting choices than are residents of apartment buildings, where planting decisions are likely made by property managers or outsourced to landscape maintenance companies. While house residents may also employ landscapers to carry out the actual labor, they are still likely to be making the
decisions about which tree species should be planted and at what abundance. Therefore, the values of house owners (which are shaped by multiple economic and cultural factors) are more directly reflected through landscaping choices than are the values of apartment dwellers. The urban forest surrounding apartment buildings more likely reflects the values of upper-level boards or managers who may not actually inhabit the building.

Discerning who the decision-makers are in regards to landscaping choices across various housing types is an important topic for future research to address in order to truly identify how neighbourhood socioeconomics influence the urban forest.

The results of the GLM indicate that distance to roads significantly influences both the species richness (it is the only variable to do so) and tree abundance within plots. The ways in which such influences manifest themselves within Peel and between land use types were discussed in the previous section. While work has been done that examines how roads affect tree species (Forman and Deblinger 2000; Eigenbrod et al. 2009; Avon et al. 2010), and the success of trees that grow along roads (Jim and Liu 2001; Jim and Chen 2009), currently, the direct effect that proximity to roads has on tree patch species richness and abundance has not been clearly studied. This is an important future area of research because as urbanization increases, road abundance and density are likely to also. As the results of the regression analysis emphasize, road network expansion will have a substantial impact on surrounding tree populations, it is therefore vital to determine what those specific impacts will be.

5.5.5 Finer Resolution Data is Required

The majority of the results indicate that employing a DA-level analysis may be too course of a resolution for most socioeconomic measures to be able to indicate their relationship with species richness. This is not surprising given that the factors determining which residents invest in tree species diversification, for what reasons
and which species they ultimately plant operate at the household level (Iverson and Cook 2000; Hope et al. 2003; Martin et al. 2004; Grove et al. 2006). While similar levels of interest in property management are likely to be comparable throughout a neighbourhood (as a result of the mechanisms discussed above), the influence that each socioeconomic and urban form variable imparts will be only observable through the decisions made by each household individually.

The UFORE data used in this study, is based on plots small enough to potentially fit onto individual housing plots. However, during data collection, there were no restrictions placed on plot locations to keep them from overlapping multiple property boundaries. In addition, UFORE plots are not designed to represent the full tree diversity of a single property. The smallest unit of census data that could be attributed to these plots is at the level of the DA, which was used in this study, but is still much larger than the plots themselves. Future research should rectify these spatial incongruities so that the analysis of species richness could be conducted at a household level to potentially better understand such dynamics. While this type of work has been conducted for herbaceous species (i.e. Marco et al. 2007) it has not been conducted for trees.

5.6 Conclusion

The results of this study confirm that Peel Region is subject to many of the same forms of urban forest inequality as other cities around the world (Hope et al. 2000; Iverson and Cook 2000; Sudha and Ravindranath 2000; Pickett et al. 2001; Pedlowski et al. 2002; Perkins et al. 2004; Troy et al. 2007; Landry and Chakraborty 2009; Luck et al. 2009). Like those cities, this inequality manifests itself through the uneven distribution of ecological resources and can be quantified by observing the urban forest. This study found that the economic standing of residents in Peel influences their exposure to tree abundance. Urban tree species richness was found to vary less predictably with respect to neighbourhood characteristics but is hypothesized to be more sensitive to socioeconomic drivers operating at the household level. Finally, distance of plots from roadways was found to correlate
negatively with tree abundance and species richness. The lack of negative influence exerted by roads and population density is an optimistic conclusion, it indicates that urban tree populations, under the right management and care programs, should be able to succeed as Peel continues to urbanize. These results hold social, political, and reforestation implications for the region of Peel as well as for other urban areas concerned with providing all of their residents equal access to critical ecological services.
Chapter Six
Conclusion

This thesis has addressed two main objectives in an attempt to quantify the diversity and distribution of tree species within the region of Peel and to understand the socioeconomic and urban form variables that shape those patterns.

First, species richness, alpha diversity, evenness and beta diversity values were calculated for each of eight land use types and four municipalities within Peel. The results indicated that species richness and tree abundance differ significantly according to land use and municipality and possible reasons for these differences were speculated. The levels of diversity observed within each discrete land use type and municipality fell within a range commonly observed in the literature (Magurran 1988); only residential land substantially exceeded these levels. The high level of diversity attributed to residential areas is often explained as being a result of private planting decisions made at the level of individual households (Hope et al. 2003; Grove et al. 2006; Walker et al. 2009) and its observation in this study warranted the second main objective to focus primarily on residential land.

Apart from on commercial land, low species evenness was found to occur almost ubiquitously across Peel. This finding carries management implications as increasing evenness, with particular attention paid to planting native species, throughout the region will improve the odds that Peel’s urban forest will remain diverse under strenuous circumstances or after stochastic events. In general, diversity values between land use types were found to be negatively correlated with evenness values. Therefore, to increase the population of rare species within Peel, a focus on decreasing beta diversity, thereby dispersing current land use-specific species across a greater range of land use types, may be a beneficial strategy.

However, species accumulation curves illustrated that the sampling effort conducted by the TRCA was not exhaustive in all land use types. Therefore, an
important area of future research lies in conducting a more thorough survey of Peel’s urban trees to both confirm the results produced from this preliminary study and to provide a means of identifying changes to the urban forest that will occur over time.

To complete the second objective, a correlation coefficient and regression model were used to identify which of nine socioeconomic, urban form and spatial variables are significant factors controlling the distribution of trees within Peel’s residential areas. Ultimately, it was shown that Peel is vulnerable to many of the same environmental inequalities that have been documented across much of North America (Perkins et al. 2004; Landry and Chakraborty 2009). Urban forest managers must address this issue and find ways to balance the distribution of Peel’s valuable urban trees so that residents of all income levels may enjoy equal access. This study found that the occupation of tall apartment buildings by low-income residents is one way to promote such access.

However, a generalized linear model found that in residential neighbourhoods, many of the socioeconomic variables considered lack influence over surrounding tree richness and abundance. This indicates that processes governing these traits are occurring at a smaller scale than can be observed using DA data. Therefore, future work should be conducted to examine how socioeconomic drivers in Peel region operate at the level of the household. In so doing, more direct conclusions may be drawn about how residents’ decisions directly influence the urban forest.

The urban form variable, proximity to roads, was found to be an important factor contributing to both species richness and tree abundance values throughout the entire study area. This finding highlights important urban design considerations. Urban planners must keep in mind that road networks directly impact the surrounding vegetation and therefore design principals that seek to minimize negative impacts should be employed as urban development in Peel continues to occur.
These findings are important for Peel because they identify critical controls and relationships between the urban environment and the people who live within it. As well, this study will stand as a point of reference so that the extent to which future urbanization affects the richness and distribution of Peel’s urban trees may be measured. However, these results may also be used to represent the processes that are occurring across North America as urban landscapes develop and expand. The types of land use and the distribution of habitants within Peel are comparable to those found across the continent, therefore other municipalities may expect to find similar relationships operating within their urban ecosystems.

Collectively, the results of this thesis help to expand the developing field of urban ecology. This project specifically addressed a gap in the literature concerning species-level analyses of urban tree populations. It was found that species richness and diversity vary considerably between different components of a single urban region and that they are shaped, in part, by human practices. Therefore, this level of focus represents an important contribution to the literature. In combination with other research that has focused primarily on total canopy cover (Heynen and Lindsey 2003; Conway and Hackworth 2007) and the distribution of broad vegetation classes across urban environments (Lundholm and Marlin 2006; Troy et al. 2007), species-level analyses will help to achieve a comprehensive understanding of urban ecological systems.

As well, by dividing the urban landscape into discrete land use types, and comparing the vegetation patterns between them, this project has addressed a gap in the literature considering urban environments at the meso-scale. Previous work that has considered the urban environment as one contiguous land use (Walker et al. 2009), or else focused solely on a single component of the urban mosaic (Tanner and Grange 2005; Smith et al. 2006; Allison et al. 2008), has failed to identify how intermediate scales of urban vegetation composition influence total urban forest composition. Distinct vegetation patterns within and across mosaic components were found to be integral in explaining the distribution of tree species and abundance across Peel region as a whole.
Finally, this thesis has contributed toward bridging the deep divide between social and physical themes in urban ecology. The significant relationships that were found between social measures of income and residential structure and physical aspects of urban forest structure and composition in Peel emphasize the inseparable nature between the two themes. This connection has only recently begun to be appreciated within urban ecology; however, it is crucial, for as much as urban trees influence urban residents, the opposite is also true.

As well, this project associated the well-established UFORE Model methodology, often used in evaluating the physical components of urban forests, with socioeconomic correlates for the first time. Increased use of UFORE data in this way may prove to be a powerful tool for expanding the unification of social and physical analyses to municipalities world wide.

Despite the progress that this thesis has made toward achieving a more comprehensive understanding of the factors that shape urban forest diversity, there are still areas within the field of urban ecology where questions remain and research opportunities exist.

One of these areas is the development of specific species profiles. This study found that no two tree species are equally distributed across land use types or municipal boundaries. The unique characteristics of each tree species contribute to its distinct distributional pattern, as does its historical context within the region. More time needs to be spent to inventory the characteristics and histories of each species in order to draw precise conclusions about why the 179 tree species in Peel exist where they do. This level of specificity will aid in the accuracy of projections regarding the future of Peel’s urban forest and will enhance management capabilities.

Another factor that contributes to shaping the composition, distribution, success, and equality of the urban forest is municipal policy. By expanding the quantitative analysis conducted in this study to include details about the management programs that influence diversity values, it will become evident which
municipal policies are most affective. It will therefore be possible to inform regional governments about which policies they should adopt and which to avoid in order to maximize the potential of their urban forest.

Regarding the findings of Chapter Five, two additional areas of research exist that would help to further explain how species diversity and abundance are driven by neighbourhood socioeconomics. The first of these areas requires investigating residents’ level of interest in having trees on their property. It is commonly assumed that trees are universally desired - and while indeed, they often are – individual values and practices attributed to culture, ethnicity or income level may reduce their appeal. It is important to determine how strong the influence of such taste-related factors is in order to design tree planting initiatives that reflect the desires of urban inhabitants and improve the urban forest, particularly in highly multicultural regions like Peel.

The second area involves discerning who is actually making decisions at the household level regarding what tree species are planted and where. Individual residents, neighbourhood committees, gardeners and landscapers, property managers, business owners, or other stakeholders may be responsible for making such choices. The values and tastes of these decision makers again come into question, as does their level of investment in the local landscape. Determining which groups educational programs and incentives should be catered toward and what tree-providing sources (i.e. local nurseries, private companies) are being selected from will increase the efficiency of efforts to diversify and amplify local tree populations.

Urban trees provide irreplaceable spiritual, recreational, economic, environmental and aesthetic services to city residents. Furthering the discipline of urban ecology is imperative so that, as human and climate driven stressors increasingly strain urban environments, managers and planners are able make informed decisions that benefit both the urban vegetation and the humans living amongst it, thereby improving quality of life for both.
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Appendices

Appendix 1: Study Area – Map of Peel

Municipal boundaries outside of Peel data source: Geography Network Canada 2010. Municipal boundaries within Peel data source: Peel Region Data Centre 2010.
Appendix 2: Peel Plots and Road Network

Road Network in Peel Region

Legend
- Mississauga Plots
- Caledon Plots
- Brampton Plots
- Bolton Plots
- Roads

### Appendix 3: Most Abundant Species by Land Use Type

<table>
<thead>
<tr>
<th>Park</th>
<th>Commercial</th>
<th>Agriculture</th>
<th>Residential</th>
<th>Transportation</th>
<th>Vacant</th>
</tr>
</thead>
<tbody>
<tr>
<td>Black Ash Fraxinus nigra</td>
<td>White Ash Fraxinus americana</td>
<td>European Buckthorn Rhamnus cathartica</td>
<td>European Buckthorn Rhamnus cathartica</td>
<td>Northern White Cedar Thuya occidentalis</td>
<td>Northern White Cedar Thuya occidentalis</td>
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<td>(17)</td>
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<td>(17)</td>
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*Top ten most abundant species in each land use type, in order from most abundant to least. Bottom row indicates the total number of sampled species per land use type. Numbers in brackets indicate the abundance of each species in that land use type.*
Appendix 4: Species Accumulation Curves by Land Use Type

- Plot of Agriculture Species Accumulation
- Plot of Commercial Species Accumulation
- Plot of Golf Species Accumulation
- Plot of Institutional Species Accumulation
- Plot of Park Species Accumulation
- Plot of Residential Species Accumulation
- Plot of Transportation Species Accumulation
- Plot of Vacant Species Accumulation
Appendix 5: Rank-Abundance Graphs

- Rank Abundance: Agriculture
- Rank Abundance: Commercial
- Rank Abundance: Golf
- Rank Abundance: Institutional
- Rank Abundance: Park
- Rank Abundance: Residential
- Rank Abundance: Transportation
- Rank Abundance: Vacant
Appendix 6: Most Abundant Species by Municipality

<table>
<thead>
<tr>
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<th>Brampton</th>
<th>Mississauga</th>
<th>Caledon</th>
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The ten most abundant species for each municipality, in order from most abundant to least. Bracketed numbers indicate the number of each species sampled in that municipality. Bottom row indicates the total number of species sampled in each municipality.
Appendix 7: Species Accumulation Curves by Municipality

Plot of Bolton's Species Accumulation

Plot of Brampton's Species Accumulation

Plot of Caledon's Species Accumulation

Plot of Mississauga's Species Accumulation