

Exploring the Importance of Saproxyllic Beetles (Coleoptera) as Indicators of
Forest Biodiversity and Available Resources in Kitchener, Ontario

by

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Author's Declaration

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

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Abstract

Across North America insects have generally taken a backseat to more conspicuous animals (e.g. birds, mammals) and are not regularly monitored by ecosystem managers. They commonly enter the spotlight when an insect is an invasive pest species causing significant damage, whereas less attention is given to studying the population dynamics of native species. This type of monitoring can be difficult for municipalities or conservation authorities due to economical limitations, time needed for sampling, and required taxonomic knowledge. However, this type of research needs to be incorporated into management plans in order to effectively facilitate sustainable ecosystems. Trees and forests provide unique ecosystem services and an important component of their health lies with saproxylic beetles. Relentless urban sprawl and other anthropogenic influences continue to pressure these ecosystems into new stable states, altering their function and composition. Invasive species like the emerald ash borer (*Agrilus planipennis* Fairmaire) and the resulting management practices put into place by managers may have effects on resident insect species that remain unknown if insect monitoring initiatives are not put into place. My study catalogues saproxylic beetles within three parks in Kitchener, Ontario for the first time to create a baseline inventory for future research and identify potential indicators of biodiversity and resources. Based on correlation analysis, cerambycidae and curculionidae (scolytinae) were identified as possible indicators of biodiversity and deadwood. Only one site was found to contain a significantly different assemblage which may be attributed to management and tree composition. Additionally, it is suggested that the high abundance of scolytines in two sites may be related to dead ash trees and woody debris resulting from EAB infestation and management, but this study did not delve further into this issue and more research is necessary. Creating a method of sharing insect sampling information between the public, managers, and researchers needs to become a reality if successful ecosystem management is expected to be achieved not only in Kitchener, but across the province.

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“The planet has survived everything, in its time. It will certainly survive us.”

-Michael Crichton, Jurassic Park

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1.0 Introduction

Thesis Overview

Consistent monitoring of insect population dynamics has been generally excluded by ecosystem managers when creating and implementing conservation strategies. Unless there is a specific species creating cause for concern, such as the emerald ash borer (*Agrilus planipennis* Fairmaire) or Asian long horned beetle (*Anoplophora glabripennis* Motschulsky), few resources are allocated towards insect conservation. Insect surveys typically take place over years in order to sufficiently grasp population trends, can be expensive, and require a high degree of taxonomic expertise. All of these factors lead managers to generally avoid this taxa in their sampling strategies. Insects have also typically been seen as pests and garner a general negative attitude from the public, furthering the unlikelihood of regular monitoring regimes. Baseline inventories of insects are hard to come by in most areas, and species that may act as indicators of biodiversity and resources within Ontario are relatively unexplored. My study aims to contribute knowledge to both of these aspects of insect research, and demonstrate the importance of insect monitoring and conservation. Follow-up monitoring recommendations for the City of Kitchener will be provided based on my findings, and hopefully add to their Urban Forest Management Plan (City of Kitchener, 2017b).

Forests are facing increasing anthropogenic pressures, including deforestation, pollution, inefficient management practices, and the introduction of invasive species (Noss, 1999; Aber *et al.*, 2003; Pimental *et al.*, 2005; Zeran *et al.*, 2006; McKenney *et al.*, 2007; Holmes *et al.*, 2009; Jennings *et al.*, 2017). Old-growth forests, which provide insight into the deadwood dynamics necessary to maintain a forest ecosystem, have become greatly depleted. More research into the relationship between deadwood and the organisms reliant on this resource is necessary in determining the best methods of managing second growth forests which comprise the majority of forests within Ontario.

It has been suggested that fallen trees and decaying wood support at least one fifth of the fauna within a forest, but this is most likely an underestimate (Grove, 2002a; Elton, 1966). A vast expanse of vertebrate, invertebrate, and fungal species utilize these resources and help break down wood and return nutrients to the soil which supports forest growth (Siitonen, 2001; Grove, 2002a; Grove, 2002b; Ulyshen *et al.*, 2004). It is therefore imperative to support and maintain the acquisition and persistence of a diversity of dying, dead, and decaying wood in all ecosystems, including cities.

The term 'saproxylic' refers to all organisms that feed and depend upon dying and deadwood to survive; in the Order Coleoptera, this specifically includes all beetles in the families of buprestidae, cerambycidae, scolytinae, curculionidae, and other minor wood-feeding groups (Nilsson and Baranowski, 1997; Speight, 1989; Siitonen, 2001; Grove, 2002a; Jacobs *et al.*, 2007). Limited efforts have gone into exploring the impact of forest management techniques on saproxylic beetle species. For example, ash tree removal is a common practice for dealing with EAB infestations in Kitchener but this may be limiting important resources for other organisms.

This study explores the saproxylic beetle populations within Kitchener, Ontario and uses this data to propose possible indicators of forest biodiversity and resources. Additionally, sites are compared in

terms of beetle family abundance and diversity to identify any significant differences and possible causes will be explored. Finally, suggestions for monitoring techniques, information sharing, and future research are provided. The goals of this study are: 1) to determine if there are accessible and easily monitored saproxylic beetle families that can be used to infer forest biodiversity and resources; 2) to create a baseline inventory for any future research in these areas, and; 3) compare beetle assemblages pre- and post-ash tree removal to identify any differences.

Research Questions and Predictions - The following questions are addressed:

- Are there saproxylic beetle families or larval feeding guilds (i.e. functional groups) that are associated with overall saproxylic beetle trends, and may therefore be used as indicators of biodiversity?
- How diverse is the saproxylic beetle assemblage within Kitchener, Ontario? Does it differ between sites? Do functional groups differ between sites? What could be causing these differences?

Lucanidae, cerambycidae, curculionidae, and other beetle families have been identified as possible indicator groups in previous studies (Villa-Castillo and Wagner, 2002; Ohsawa, 2010; Rondeux and Sanchez, 2010; Lachat *et al.*, 2012), and it is predicted that similar results will be found here. Eucnemidae has also been suggested to have potential as an indicator in Ontario forests (Evans, 2014), and this is of interest in the present study. Due to the current infestation of EAB in all study sites and therefore dead ash trees and increased woody debris, the families that utilize freshly dead wood are expected to be most abundant. This includes cerambycidae and curculionidae, which should reflect the abundance of these resources at each site. Tilt's Bush is predicted to have a more diverse assemblage than Breithaupt Park and Laurentian Wetland due to its larger size and diversity of both deciduous and coniferous trees.

Understanding and documenting the current saproxylic beetle assemblages can aid in future monitoring techniques. A current baseline of these insects does not exist and there are few results from similar studies with which to compare. The majority of literature on saproxylic beetles originates from Europe, and studies in Canada have mainly been focused in the west (Lowman and Wittman, 1996; Werner and Raffa, 2000; Vance *et al.*, 2003; Zeran *et al.*, 2006). Research into the saproxylic beetles of Ontario is in need of increased efforts. If indicator families are identified, the monitoring of these families may be beneficial to assessing ecosystem characteristics in the future and allow managers to adjust strategies accordingly. This research will offer a glimpse into the current condition of these forests and the state of saproxylic beetle populations, and will provide insight into the efficiency of current management techniques used by the City of Kitchener.

Saproxyllic Organisms at a Glance

Saproxyllic organisms are those dependent on dying trees or deadwood in some way at some point in their life cycle (Nilsson and Baranowski, 1997; Speight, 1989; Økland *et al.*, 1996; Hammond *et al.*, 2001; Siitonen, 2001; Grove, 2002a; Jacobs *et al.*, 2007). This includes those species that consume the wood itself (xylophagous), consume wood-inhabiting fungi (fungivorous), feed off of detritus (detritivorous), or prey upon or parasitize those that do the former (Jonsell *et al.*, 1998; Siitonen, 2001; Grove, 2002b). Thousands of species of fungi, lichens, bryophytes, vertebrates, and invertebrates are classified into this group (Siitonen, 2001). Saproxyllic insects comprise a large majority of total insects within a forest ecosystem and are important components to maintaining forest function (Grove, 2002a; Grove, 2002b) by contributing to nutrient cycling and facilitating the breakdown of deadwood (Ulyshen *et al.*, 2004).

No matter what size or position on the trophic scale, all saproxyllic organisms play pivotal roles within their habitats, largely due to their effect on nutrient cycling (Reichle, 1977; Jacobs *et al.*, 2007; Stephens and Wagner, 2006). By infiltrating and increasing the rate of decay of distressed and dead trees, they help return nitrogen, carbon, and phosphorous to the soil (among other nutrients) (Ausmus, 1977; Swift, 1977; Hendrickson, 1988; Hendrickson *et al.*, 1989; Hammond *et al.*, 2001), which contributes to the creation of canopy gaps in the forest overstory fostering the growth of new trees and plants. Saproxyllic species also act as valuable resources to other important forest organisms, such as woodpeckers (Martikainen *et al.*, 1998; Hammond *et al.*, 2001). Because species within the saproxyllic beetle assemblage occupy many ecological niches and rely on a wide variety of habitat requirements, they are considered incredibly important for monitoring and inferring overall forest biodiversity and productivity (Ohsawa, 2010; Stephens and Wagner, 2006).

Saproxyllic beetles are especially numerous and taxonomically diverse and thought to outnumber the total global number of mammal, bird, and herptile species (Grove, 2002b). In Finnish boreal forests, saproxyllic insects were estimated to comprise 20-25% of all forest-dwelling species, and beetles comprise 60 families and over 800 species (Hanski and Hammond, 1995; Siitonen, 2001). In Sweden, the estimated number of saproxyllic beetles is at least 1000, which is about one quarter of all beetle species within the country (Jonsell *et al.*, 1998). In Germany, approximately 56% of forest beetle species are saproxyllic (Grove, 2002b). In small patches of forest in England, nearly 1000 species of saproxyllic beetles have been found (Hanski and Hammond, 1995). Studies conducted in Canada have documented thousands of saproxyllic insect species, but many are in need of taxonomic revision and only approximately 55% of terrestrial arthropods in Canadian forests have been described (Langor *et al.*, 2006). With Canada harbouring approximately 10% of the world's forests, the number of saproxyllic beetles is expected to be extremely high (Langor *et al.*, 2006), and further research should be dedicated to organizing and describing these resident species.

Saproxyllic beetles in particular are sensitive to changes in their environment (Stephens and Wagner, 2006; Jacobs *et al.*, 2007; Nadeau *et al.*, 2015). Many have limited movement within ecosystems and so their presence or absence can speak more directly to local disturbances compared to more mobile organisms with wider home ranges (Stephens and Wagner, 2006). Members of the carabidae, lepidoptera, odonata, and formicidae have all been successfully used as bioindicators due to their

sensitivity to habitat disturbances (Stephens and Wagner, 2006). Coleopteran communities have been shown to be less diverse and fewer in number in managed forests when compared to those that are unmanaged (Nadeau *et al.*, 2015), demonstrating their sensitivity to changes within these habitats. Many species are host-specific, therefore a wide range of trees, snags, and woody debris are required to maintain saproxylic beetle diversity (Jacobs *et al.*, 2007; Stephens and Wagner, 2006; Lachat *et al.*, 2012), including a diversity of tree species, age, density, diameter, state of decay, sun exposure, and the presence of fungi (Kaila *et al.*, 1994; Økland *et al.*, 1996; Lachat *et al.*, 2012; Jacobs *et al.*, 2007). Depending on such characteristics, each tree itself can provide a trove of vital substrates. Peeling bark, cavities filled with water and/or detritus, cracked wood, and aquatic woody debris all act as important habitats that dictate saproxylic beetle diversity (Harmon *et al.*, 1986).

The extinction trend of saproxylic beetles is widely documented throughout Europe and is most likely reflected around the globe with human settlement and disturbance. In the mid-1980s, it was estimated that approximately 20% of Europe's terrestrial and freshwater invertebrate species faced the threat of extinction (Speight, 1989). During the 18th and early 19th centuries, saproxylic insects were among the first reported invertebrate extinctions within Europe; nearly 20 of these were saproxylic beetles (Speight, 1989). Speight (1989) explains that during the 1980s, 40% of saproxylic invertebrates in Europe were threatened and most others were in decline. It was estimated only about 10% of saproxylic invertebrate species were not in need of any protection. As of 2007, it was hypothesized that more than 60% of saproxylic beetles in central Europe were threatened with extinction due to forestry practices and the scarcity of old growth trees (Buse *et al.*, 2007). This trend can be mainly attributed to fragmentation of forest ecosystems and rigid management practices that often remove the resources necessary to support beetle diversity (Grove, 2002b; Wermelinger *et al.*, 2007). The presence of these insects relies on the broad ecological features of an area, the environmental characteristics of a forest, previous disturbances, management techniques, and species interactions (Stokland *et al.*, 2012; Bouget *et al.*, 2013; Nadeau *et al.*, 2015). Insect families containing the most abundant saproxylic species include cerambycidae, curculionidae (scolytinae), nitidulidae, and buprestidae, and their presence or absence indicates resources and productivity (Siitonen, 2001). Due to the numerous undescribed species within Canada, little information regarding extinction rates in this country exists, again demonstrating the need for further documentation and classification of native saproxylic species (Langor *et al.*, 2006).

Certain species will arrive and/or thrive within an ecosystem depending on the stages of tree decay present; therefore, identifying those groups helps to communicate the diversity of available resources (e.g. distressed trees, coarse woody debris) within a forest (Esseen *et al.*, 1997; Jacobs *et al.*, 2007). For example, bark beetles (curculionidae: scolytinae) and their associated predators/parasitoids typically increase immediately following tree death along with other phloem-feeding insects (Esseen *et al.*, 1997; Siitonen, 2001; Jacobs *et al.*, 2007). After 1-2 years, secondary phloem feeders and their associated species begin to establish themselves and feed on the remaining phloem. Once all the phloem has been consumed and most of the bark fallen off, mycetophagous insects comprise the majority of those species present, in this case, determined by associated wood-decomposing fungi (Esseen *et al.*, 1997; Jonsell *et al.*, 1998). Once the heartwood of the tree starts to decay, the nutrient levels within the wood

decrease greatly and at this point other invertebrate groups such as molluscs, carabids, centipedes, and ants begin to colonize the debris (Esseen *et al.*, 1997; Siitonen, 2001).

At each stage of decay, the insect species present are associated with specific species of fungi, yeast, mites, and nematodes, the latter having been introduced as the beetles colonize the tree (Jonsell *et al.*, 1998; Siitonen, 2001). Each species therefore directly influences those that will be able to further infiltrate the decaying wood in the various stages of saproxylic succession (Esseen *et al.*, 1997; Jacobs *et al.*, 2007). For example, the bark beetle *Ips typographus* (Linnaeus) has been linked to 140 other species that also contribute to the breakdown of wood and release of nutrients back into the ecosystem (Weslien, 1992; Esseen *et al.*, 1997; Siitonen, 2001).

Ecological Requirements: The Importance of Deadwood

Deadwood is an extremely important aspect of forest ecosystems (Bouget *et al.*, 2013), and is referred to as mature timber habitat by researchers, a term that encompasses all habitat features supporting saproxylic insects (Grove, 2002a). It is the main driver in not only saproxylic beetle abundance and diversity, but also the biodiversity of the entire forest ecosystem. Wood decomposes at a slower rate than other plant matter, thereby providing a long-term resource for many organisms (Larsen *et al.*, 1978; Alban and Pasot, 1993; Kauffman *et al.*, 1993; Hammond *et al.*, 2001). In managed urban forests, distressed and/or dead trees are often seen as a hazard, unwanted, and consequently removed. In stands managed for timber production, little thought is given to retaining a diversity of mature timber habitat following harvest. However, the ecological significance of this resource is now being realized in many countries and factored into forest management strategies and conservation efforts (Grove, 2002a; Wermelinger *et al.*, 2007; Stokland *et al.*, 2012; Bouget *et al.*, 2013).

Each stage of the death and decay of a tree provides a unique array of microhabitats (Berg *et al.*, 1994; Jonsell *et al.*, 1998; Siitonen, 2001). Narrow host specificity is often observed in saproxylic beetles that colonize live and freshly dead trees, and those that depend on fungi (Grove, 2002a; Kaila *et al.*, 1994; Siitonen, 2001). As decay progresses, those species colonizing the substrate change in accordance with their habitat needs (Siitonen, 2001). Fungal species present on a dead tree are dependent on how the tree died, moisture content, sun exposure, and the species already established in the wood (Siitonen, 2001).

Very mature broadleaf trees are possibly the most important feature in mature timber habitats. A single tree of this type is defined as heterogeneous and can support a vast diversity of saproxylic species because of its diversity of microhabitats and stable microclimates (Jonsell *et al.*, 1998; Grove, 2002a; Maeto *et al.*, 2002; Speight, 1989). They encompass a wide variety of physical features including shaggy bark, cracks or hollows, varying branch diameters, and dead branches, all of which provide for heterogeneity (Esseen *et al.*, 1997). They are also capable of exhibiting varying stages of decay at one time, and support a wide array of fungi and a correspondingly high diversity of insects.

For some saproxylic beetles, it is the size of the tree that counts; i.e. certain species have specific preferences for certain tree diameters (Jonsell *et al.*, 1998; Siitonen, 2001). Many saproxylic beetles

that prefer large diameter, old growth trees have become either rare or threatened due to the loss of this habitat type in managed and urban forests, as well as to the global destruction of old growth forests generally (Wermelinger *et al.*, 2007). Trees, stumps, and logs with large diameters have been found to support more microhabitats than those with smaller diameters, and therefore more saproxylic insects (Esseen *et al.*, 1997; Grove, 2002a; Larrieu and Cabanettes, 2012; Winter and Möller, 2008; Bouget *et al.*, 2013).

Dead trees often support richer assemblages of saproxylic beetles than living trees. Previous researchers collected 92 saproxylic beetle species from five spruce trees killed by the spruce bark beetle (*Ips typographus* Linnaeus) with an average of 10 000 individuals caught per tree. In contrast, only 300 individuals were found on living trees (Siitonen, 2001). In a comparison of beetle species between two different forest management types, those left standing as snags during timber harvest harboured 18 rare species while those that felled a selection of trees had none (Siitonen and Martikainen, 1994; Hanski and Hammond, 1995). Rare species are often indicative of biodiversity and the productivity of a forest and are an important consideration in forest management practices.

Snags, in particular, have been found to support a wide array of microhabitats; Vuidot *et al.* (2011) found that snags contained twice as many species as living trees. Snags tend to decay more slowly than fallen trees and support specialized species of cerambycids, ptinids, and scolytines (Siitonen, 2001). A slower rate of decay means that the resource will continue to be provided and support saproxylic communities for long periods of time. Trees become increasingly heterogeneous as they decay, thus allowing more organisms to make use of them due to the development of increased microhabitats (Siitonen, 2001). Wood becomes softer with decay, allowing cavity nesters to burrow into the rotting heartwood, cracks and separation of the bark creates cavities, and more fungal species begin to grow (Vuidot *et al.*, 2007). Many invertebrates, mosses, and fungi are able to inhabit snags as they fall and decay into woody debris (Jacobs *et al.*, 2007).

Coarse woody debris (CWD) is an essential resource (Harmon *et al.*, 1986; Goodburn and Lorimer, 1998; Siitonen, 2001) that provides food, shelter, and reproductive sites for a multitude of organisms, including invertebrates, mammals, birds, and herptiles (Goodburn and Lorimer, 1998; Siitonen, 2001). It is an important source of nutrients that will slowly be associated into the humus layer of the soil and used by vegetation (Goodburn and Lorimer, 1998; Siitonen, 2001; Ulyshen *et al.*, 2004). Some researchers have suggested that coarse woody debris is one of the most important resources within a forest, directly influencing all aspects of biodiversity (Ulyshen *et al.*, 2004).

Saproxylic insects are sensitive to the deadwood dynamics of their ecosystem. In a natural forest, there is a rate of deadwood loss and gain, as well as a frequency of disturbance that results in deadwood being added to the ecosystem (Harmon *et al.*, 1986; Siitonen, 2001; Grove, 2002a). In North America, CWD has been shown to be highest in density following a large disturbance and in old growth forests, but low later during mid-successional stages (Siitonen, 2001). Saproxylic beetle populations will reflect CWD and follow a boom and bust type trend as such resources become abundant and then dissipate.

CWD is found commonly in natural and old growth forests where it is created through natural processes with little to no intervention from humans. Siitonen (2001) outlines the three factors affecting the

volume of CWD within a forest: productivity, decomposition, and disturbances. Maeto *et al.* (2002) found that cerambycid populations were higher in old growth forests than in secondary growth forests and tree plantations, reflecting the amount of available CWD in each ecosystem. Research has demonstrated the temporary population boom of wood-boring insects following logging activities when CWD is briefly abundant, again demonstrating the importance of this resource (Ulyshen *et al.*, 2004).

Comparing Diversity between Old Growth Forests and Tree Plantations

Centuries of logging as a continued practice have resulted in the decline of old growth forests (Esseen *et al.*, 1997). These forests are important because they have a great abundance and variety of woody debris, tree age diversity, and multiple stages of decay that allow many saproxylic organisms to thrive (Esseen *et al.*, 1997; Goodburn and Lorimer, 1998; Siitonen, 2001; Maeto *et al.*, 2002). It is pivotal to support and maintain current old growth forests as researchers have found higher diversity in natural forests than reforested areas (Grove, 2002b; Fernandes *et al.*, 2010).

Some species of saproxylic insects are restricted to old growth forests and researchers in the United Kingdom have used their presence to distinguish between old growth and secondary growth forests when assessing conservation potential (Grove, 2002a). Because there are few old growth forests left, especially in heavily populated areas, proper forest maintenance practices should be followed. Even if a forest is small (<10 ha) and relatively isolated, it can still provide essential habitat to rare saproxylic species if the ecosystem is appropriately heterogeneous (Nilsson and Baranowski, 1997).

Secondary growth forests have become increasingly abundant as a result of past timber harvesting, agriculture, and urban sprawl. They are often the result of tree plantations or succession following clear cutting (Grove, 2002a). Secondary growth forests can have from 2-60% less woody debris than old growth forests (Grove, 2002a; Goodburn and Lorimer, 1998; Siitonen, 2001), and species richness is on average 18-75% greater in old growth forests than in managed stands (Siitonen, 2001). However, secondary and urban forests are still important ecosystems that deserve greater attention for their potential as productive ecosystems (Horák, 2011).

Managing for Sustainable Ecosystems via Saproxylic Communities

Forests are managed for a variety of reasons, including recreation, to conserve wildlife, and for timber production (Shimatani, 2001). Often this management includes the reduction of woody debris and fallen trees to improve aesthetics and usability (Fridman and Walheim, 2000; Wermelinger *et al.*, 2007; Bouget *et al.*, 2013). In managed urban forests, trees are thinned due to sanitary, aesthetic, and/or safety reasons. Trees that are old, diseased, dead, and/or pest-infested are often removed despite their ecological significance (Jonsell *et al.*, 1998; Siitonen, 2001). These are pivotal sources of CWD and their removal eliminates essential saproxylic habitat and long-term resources from the environment. Thinning also contributes to homogenization of the habitat by reducing the diversity of tree age, damage, and decay which are naturally found in forests (Jonsell *et al.*, 1998). Extinction of saproxylic

insects is mainly the result of strict unchanging forest management practices and habitat fragmentation and loss (Hanski and Hammond, 1995; Siitonen, 2001; Grove, 2002a; Grove, 2002b). Saproxylic beetles are sensitive to forest management and it is important to conduct studies to determine how they are affected by regional management techniques (Speight, 1989; Hammond *et al.*, 2001; Martikainen, 2001; Maeto *et al.*, 2002).

Metapopulation and Resource Patch Modelling - Understanding saproxylic beetle metapopulations is important for maintaining species diversity and abundance. Metapopulation modelling helps researchers understand the required frequency and dispersal of substrate patches that best supports saproxylic species (Siitonen, 2001; Lachat *et al.*, 2012). Typically metapopulations of animals are measured over large areas (i.e. km or ha), but saproxylic beetles occur in small patches based on substrate availability, often over a few square metres (Grove, 2002a; Hanski, 1999; Thomas, 2000). The frequency and distance between patches are important considerations in forest management, as are the rate and distance of dispersal within a species (Ranius and Hedin, 2001; Siitonen, 2001; Grove, 2002a).

Dispersal in saproxylic insects is thought to have a negative relationship with the frequency of natural disturbances. When disturbances are recurrent there will be a regular supply of deadwood and insects will have lower dispersal abilities (Siitonen, 2001; Grove, 2002a; Nilsson and Baranowski, 1997; Ranius and Hedin, 2001). Some saproxylic beetles can survive on a single substrate for generations and only relocate when that resource is depleted (Ranius and Hedin, 2001; Siitonen, 2001; Grove, 2002a). These species have uniquely limited dispersal capabilities, so connectivity of patches is essential (Grove, 2002a; Ranius, 2002). They are often used as indicator species of saproxylic biodiversity (Ohsawa, 2010). In managed forests disturbance events are less likely to occur if they are not included in the management strategy. Therefore saproxylic beetles surviving in managed habitats may be those with greater dispersal abilities enabling them to move between fragmented forests in search of resources.

The stage of tree death and decay at which a beetle species colonizes wood is directly related to its dispersal ability. Slowly decomposing snags or woody debris found in tree hollows can sustain insect populations for years, and possibly centuries, allowing species associated with these microhabitats to be successful even with low dispersal capabilities (Nilsson and Baranowski, 1997; Ranius and Hedin, 2001). Those that initially arrive on dying or freshly dead wood often have high dispersal capabilities due to the short lifespan of this resource (Ranius and Hedin, 2001). For example, those in the genus scolytinae are generally thought to have high dispersal capabilities due to their scarce and/or ephemeral habitats (Grove, 2002a).

Each species' ability to move between patches needs to be taken into consideration when determining the most appropriate management techniques. The natural distribution of substrates within old growth or natural forests in a similar geographical area should be studied and recreated to the most feasible extent. Substrates need to be situated within a proximity that is appropriate to support saproxylic species (Ranius and Hedin, 2001). When it is not possible to retain an appropriate amount of a single substrate within an ecosystem then it may be more beneficial to maintain a greater number of a different type of substrate (Jonsell *et al.*, 1998). Those dependent on old trees, for example, may need a

higher number than can be maintained. In this case focusing on species dependent on another substrate that can more readily be managed for will be more effective (Jonsell *et al.*, 1998).

Mimicking Natural Disturbances - The natural history of disturbances in a forest must be considered, whether it be wildfires, pest outbreaks, or windfall (Harmon *et al.*, 1986; Attiwill, 1994). Natural disturbances should be encouraged when possible and emulated when not (Grove, 2002a). Ignoring the historical wood dynamics of an ecosystem can lead to greater fragmentation of metapopulations and local extinction of rare or threatened species (Grove, 2002a). It is possible, through proper forest management, to reintroduce extinct or extirpated species as long as habitat requirements are met (Grove, 2002a).

Wind felling, the use of explosives, girdling, and notching are all methods of creating disturbances within managed forests. Wind felling is a common practice to mimic naturally occurring windfall (Attiwill, 1994). However, this results in all deadwood on the forest floor which increases decay rate and will not support all species (Grove, 2002a; Harmon *et al.*, 1986). Decay is influenced by temperature, precipitation, and tree species and fungal growth is stimulated on the damp forest floor (Yin, 1999; Siitonen, 2001). The use of explosives is also an attempt at simulating windfall with mixed results and is not suitable for urban forests. Notching has been shown to be less effective than girdling at creating slow tree death, which is important for supporting the largest possible saproxylic diversity (Martikainen, 2001). Many urban forests are managed to decrease the possibility of disturbance events but the ecological consequences of this practice may be significant.

Tree Management Practices - Tree species is an important aspect of saproxylic beetle assemblages. Jonsell *et al.* (1998) estimate that nearly all tree genera support monophagous saproxylic insect species. Certain species of trees die and decompose in characteristic ways that affect their colonization (Esseen *et al.*, 1997; Siitonen, 2001). For example, spruce trees are prone to uprooting and stem breakage, pine trees die standing and remain as snags for long periods of time, and birch and aspen tend to snap once dead to form broken snags (Siitonen, 2001). Aspen also tend to rot from the middle creating tree hollows (Siitonen, 2001). The position and species of a dead tree will determine its moisture content, microclimate, and rate of decay and therefore longevity of available habitat (Siitonen, 2001). Each tree will cater to a different cluster of species depending on these factors.

Excessive forest hygiene, salvage logging, and fuelwood or biomass harvesting are all detrimental practices that greatly reduce available deadwood (Fridman and Walheim, 2000; Grove, 2002a; Bouget *et al.*, 2013). Removing large trees in secondary forests can have detrimental effects on the saproxylic beetle assemblage (Økland *et al.*, 1996; Martikainen, 2001; Grove, 2002a; Grove, 2002(b); Ranius, 2002). Old growth trees often support hollows that harbour unique, low-dispersal species (Nilsson and Baranowski, 1997; Ranius, 2002). Ranius (2002) discovered that these specialized species were found to be absent in Swedish forests containing less than 10 hollow oaks. Maintaining large diameter trees, which have been found to support the majority of rare and specialized species, will aid in saproxylic

beetle conservation (Siitonen, 2001; Grove, 2002b). Eliminating trees that have fallen removes important resources from the environment and reduces the amount available nutrients (Grove, 2002a). The decrease in available CWD within managed forests is one of the leading causes of saproxylic beetle reduction (Siitonen, 2001).

Uneven-aged management, which has commonly been used in mid-western North American forests, involves removing a number of trees in different size classes and all of those with a diameter of over 60 cm at breast height (DBH) every 12 to 15 years (Goodburn and Lorimer, 1998). Selectively removing larger trees greatly diminishes the chance of natural tree death and the creation of woody debris and microhabitats. Old growth forests and unmanaged second growth forests have been shown to have increased woody debris and snags (Goodburn and Lorimer, 1998). It is important to maximize the amount of available woody debris, snags, large diameter trees, and tree diversity within secondary growth forests through sustainable forest management (Attiwill, 1994; Maeto *et al.*, 2002).

Clear-cutting greatly disrupts the CWD cycle. All or most of the timber cut is taken away which is in stark contrast to natural processes causing mass tree death such as forest fire or a pest outbreak. The removal of trees and preparation of the soil before planting anew destroys any rotting logs within the soil (Jonsell *et al.*, 1998; Siitonen, 2001). Clearcutting practices can be improved by retaining some live trees, dead trees (snags and logs), and woody debris at time of removal. These trees can help support local populations of saproxylic beetles, provide important old growth substrate once forest cover has re-established, and provide an ecological corridor between intact forests (Martikainen, 2001). The amount and type of substrates left behind should be based on the forest characteristics prior to cutting. Simulating previous conditions will enable some resident species to survive (Jonsell *et al.*, 1998).

CWD Acquisition and Canopy Gaps - The management practice that incorporates dead wood into the ecosystem is often referred to as morticulture (Harmon, 2001; Grove, 2002a). In this practice, a sufficient number of old trees are left to age, die, and decay without being removed. In old growth forests in Finland, dead trees with a diameter greater than 30 cm accounted for 42-54% of all CWD and those with a diameter of less than 10 cm accounted for 1.7-2.7% (Siitonen, 2001). The colonizing fungal species that dominate deadwood and CWD will depend on the characteristics of the substrate, therefore a large number and diversity should be incorporated into the environment (Jonsell *et al.*, 1998; Grove, 2002b).

Felling trees and creating canopy gaps can increase the amount of available CWD and encourage saproxylic beetle diversity (Goodburn and Lorimer, 1998; Ulyshen *et al.*, 2004). Some beetles are thermophilic and have been found in high densities within such clearings and along forest edges, signifying the importance of this type of feature (Martikainen, 2001). Diversity is especially high in forest edges with a gradual transition from low to high vegetation that provide ample heterogeneous resources for saproxylic beetles (Wermelinger *et al.*, 2007).

Ulyshen *et al.* (2004) found that cerambycids, buprestids, brentids, clerids, scolytines, and platypodines all experienced a population influx up to 50 m surrounding newly-created canopy gaps with abundant

coarse woody debris in contrast to old gaps with little debris. They reported that these beetles had approximately double the species richness and six times the abundance in newly created gaps (Ulyshen *et al.*, 2004). Conversely, some species requiring specialized stable microclimates survive within the centre of a forest where conditions are more consistent so careful consideration of species present is necessary (Siitonen, 2001). In Europe, extensive forest management that does not mimic the natural CWD dynamics has led to the extirpation of many beetle species and many more are threatened (Grove, 2002a; Ulyshen *et al.*, 2004).

Population booms associated with newly formed gaps within forests have been observed as being temporary and reach their pinnacle approximately one year following disturbance. However, the value of creating these gaps is still high (Esseen *et al.*, 1997; Ulyshen *et al.*, 2004). Little is known about how saproxylic beetles disperse into the surrounding forest following resource use. These disturbances may be frequent enough in natural forests that saproxylic beetles depend on the woody debris created by these events to maintain diversity and abundance.

Invasive Species and Resource Pulses

North American forests have become particularly susceptible to invasive species since European and other old world human colonization, which contributed to the spread of old world species and the destruction of new world ecosystems (Koenig *et al.*, 2011). Invasive plant and animal species that infiltrate a forest can cause major changes to species composition and ecosystem function. Depending on the severity of tree death, saproxylic pest outbreaks can cause drastic changes to the understory and canopy, microclimatic conditions, enable more invasive species, alter the nutrient cycle, and increase coarse woody debris (Koenig *et al.*, 2013; Herms and McCullough, 2014).

Currently EAB is devastating ash trees (*Fraxinus* spp.) throughout eastern North America. EAB was initially discovered in Canada near Windsor, Ontario in 2002 but had likely been present for at least a decade before detection (Cappaert *et al.*, 2005; Burr and McCullough, 2014; Herms and McCullough, 2014; Jennings *et al.*, 2017). Dendrochronological studies revealed EAB had been present in southeast Michigan by at least the early 1990s (Herms and McCullough, 2014). Since that time, EAB has managed to kill hundreds of millions of ash trees in northeastern North America and continues to threaten the billions that remain along with all of the ecological services provided by these trees (Rebek *et al.*, 2008; Herms and McCullough, 2014; Jennings *et al.*, 2017). The most widespread ash species in North America are susceptible to colonization, including green ash (*Fraxinus pennsylvanica* Marsh.), white ash (*F. americana* L.), and black ash (*F. nigra* Marsh.); however, the white fringetree has been identified as a non-ash host of EAB in Dayton, Ohio, and other trees within the family Oleaceae may be at risk of infestation (Herms and McCullough, 2014; Cipollini, 2015).

Eradication efforts of EAB have proved to be economically and technologically unfeasible because of its ability to infest new areas without notice due to difficulty of detection (Aukema *et al.*, 2011; Jennings *et al.*, 2017). Native to Asia, the most likely method of introduction was infested pallets or other wooden shipping material from China with multiple introduction events (Herms and McCullough, 2014). Studies

to date have focused on the life history, trapping, biological control, and eradication of EAB, but few have evaluated its impact on local saproxylic beetle populations or on insects directly related to ash (Jennings *et al.*, 2017). More than 282 arthropod species in North America depend on ash as one of their main resources, at least 43 of which are monophagous (Herms and McCullough, 2014).

Humans contribute to the spread of saproxylic invasive species through the movement of infested material, such as firewood and nursery stock (Herms and McCullough, 2014). This allows insects to spread faster than they would through natural means (i.e. adult dispersal). Additionally, the homogenization of ecosystems makes them vulnerable to pests. Urban ecosystems, including gardens and boulevard trees, are typically comprised of limited species diversity. Ash trees were popular ornamental trees and often used to line the streets of residential areas. In the continental United States, ash species make up 20% or more of urban and suburban trees in many municipalities (Herms and McCullough, 2014). Their frequent use in urban areas has created corridors through which EAB is able to move across the fragmented landscape. The invasion of EAB has cost hundreds of millions of dollars per year and has managed to spread across 27 states and 2 provinces (Jennings *et al.*, 2017).

Although universally seen as a negative influence, invasive species can provide essential ecological functions to an ecosystem. Researchers have already identified the benefit of high EAB density to woodpeckers, which appear to experience either a population boom or influx in the years following infestation (Edworthy *et al.*, 2011; Flower *et al.*, 2014). The arrival of saproxylic insects following a pest outbreak has been well documented and subsequently influences the populations of other saproxylic species higher on the trophic scale (Drever *et al.*, 2009; Edworthy *et al.*, 2011). Parasitoid wasps and other predatory insects also benefit from high densities of EAB larvae (Jennings *et al.*, 2017).

A resource pulse is one important temporal variation of ecosystem components occurring across all habitat types; notably this is when a particular resource drastically increases in abundance. The basis of this concept lies within the species-energy theory that predicts that local species richness increases with available energy (Holt, 2008; Drever *et al.*, 2009), one that often benefits other organisms dependent on the resource by increasing their birth rates, decreasing death rates, and/or causing them to immigrate to the area (Ergon *et al.*, 2001; Yang, 2004; Holt, 2008; Drever *et al.*, 2009; Norris and Martin, 2010). The responses of organisms to a resource pulse can have dramatic effects on an ecosystem, either in the short or long term (Yang *et al.*, 2008; Holt, 2008), and may result in a shift between stable ecosystem states, thereby creating long-lasting alterations to ecosystem structure (Holt, 2008).

Within forest ecosystems, resource pulses often occur following an insect outbreak (Yang, 2004; Drever *et al.*, 2009). The rapid increase in EAB populations within forests is an example of a dual resource pulse. The beetles themselves are a prey resource that is reaching very high densities. Ash trees are abundant in North America in both forests and urban areas allowing EAB's reproduction rates to increase with little hindrance (Herms and McCullough, 2014). In response, their predators experience similar effects as they take advantage of EAB populations. Although in some cases it is unclear if woodpeckers are experiencing increased fecundity or migrating to high density areas, they are experiencing increasing populations where EAB larvae are abundant and ash tree crown death is apparent (Lindell *et al.*, 2008; Edworthy *et al.*, 2011; Jennings *et al.*, 2013; Koenig *et al.*, 2013; Flower *et*

al., 2014). EAB is also increasing the amount of dying and dead ash trees that act as resources for other saproxylic insects.

The resource pulses resulting from saproxylic insects have been documented by many researchers, mainly their effects on insectivorous species such as woodpeckers in North America (Martin *et al.*, 2006; Bonnot *et al.*, 2008; Lindell *et al.*, 2008; Drever *et al.*, 2009; Drever and Martin, 2010; Norris and Martin, 2010; Edworthy *et al.*, 2011; Koenig *et al.*, 2011; Koenig *et al.*, 2013; Flower *et al.*, 2014;). These studies have documented increased forest bird richness and abundance following an outbreak with particular focus on the mountain pine beetle (*Dendroctonus ponderosae* Hopkins) and EAB. Few studies, however, have tracked the population changes of other saproxylic beetles following these resource pulses.

Despite the copious documentation of resource pulses and their effects, the resulting food web and ecosystem shifts may be difficult to interpret. Koenig *et al.* (2011) observed an increase in insectivorous birds many years following a pest outbreak and suggest although population booms may not be immediately evident, there may be very long term effects. Norris and Martin (2010) documented the potential detrimental effects of resource pulses, demonstrated by the collapse of red-breasted nuthatch populations years following an MPB outbreak. Long term studies of saproxylic beetles in areas where pest outbreaks are occurring are therefore necessary to determine the temporal impacts on forest communities. My study provides a baseline inventory of saproxylic beetles that can be used for comparison to future studies aimed at documenting the impact of EAB.

Using Saproxylic Beetles as Indicators of Ecological Integrity

Ecological integrity can be defined as the presence of all functions, processes, and elements at appropriate rates that reflect the natural state of a system (Grove, 2002b). Using saproxylic beetles as indicators of ecological integrity can be a more economically and ecologically feasible method than monitoring complete biodiversity (Grove, 2002b). Evaluating ecological integrity speaks more to the sustainability of its current state. If an ecosystem is managed for integrity, it can be assumed that both biodiversity and functions within the ecosystem are simultaneously maintained (Angermeier and Karr, 1994; Grove, 2002b). When the goal of management is to maintain high biodiversity and functionality, historical conditions of the ecosystem may not be met (Goldstein, 1999, Grove, 2002b). Indicators are species, families, or functional groups that communicate the state of certain ecological components, often biodiversity and ecosystem productivity. Saproxylic beetles are particularly useful as indicators because of their sensitivity to habitat changes, robust taxonomic knowledge, and ease of collection (Siitonen, 2001; Grove, 2002b; Bouget *et al.*, 2013). Typically rare or regionally confined beetles are found to be the most useful as indicators. Saproxylic beetle species can help gauge the abundance and presence of these resources that reflect overall ecosystem productivity (Grove, 2002b). Those that are sensitive to changes in their environment and resources are best suited as indicators not only to determine the availability of deadwood, but also as an indicator of overall forest diversity (Hammond *et al.*, 2001).

Creating complete species lists in any single ecosystem is an extremely in-depth and difficult task requiring extensive economical means. In lieu of these lists, indicator species are often used to determine the productivity of an ecosystem (Lachat *et al.*, 2012). Using species richness alone as an indicator of the heterogeneity of an ecosystem can be problematic. Depending on the sample size within a study, the size of the study area, study site productivity, and the number of rare species present, species richness may produce skewed results (Lachat *et al.*, 2012). Assessing species richness does not account for the specific habitat requirements of each species, therefore little is revealed about the heterogeneity of a forest ecosystem (Toïgo *et al.*, 2013).

The practice of using saproxylic beetles as indicators of sustainable forest management and to identify ecologically significant habitats in need of conservation is a common practice in Europe (Grove, 2002a; Nilsson *et al.*, 1995). It is considered to be both logistically and ecologically sensible because saproxylic beetles reflect essential natural processes within a forest ecosystem (Siitonen, 2001; Bouget *et al.*, 2013). They have also been useful as indicators of sustainable forest management practices in tropical forests (Grove, 2002b). In Japan, Maeto *et al.* (2002) suggested the regular monitoring of cerambycid species associated with old growth forests to evaluate conservation and/or restoration status through insect trapping. The diversity and abundance of saproxylic beetle functional groups or guilds is indicative of ecological integrity and the availability of diverse resources (Grove, 2002b).

Challenges of Using Biological Indicators for Ecosystem Assessment - Choosing an appropriate indicator can be cost extensive and time consuming. Beetles should be collected over several years to establish common community trends and determine the most suitable candidates. Many traps over a broad area may be required to obtain a true representative sample that includes rare species because some saproxylic beetle species can be regionally specific and have low dispersal capabilities (Nilsson and Baranowski, 1997).

Many researchers have pointed out the lack of reliability and ubiquity of saproxylic beetle indicators. Whether it be a particular species or functional group, once identified they can often only be used in habitats that are similar to those in which they were identified. Tree species, size of the area, climate, diversity of woody debris, and latitude can all influence the species able to survive in a forest (Siitonen, 2001). For example, indicators identified in southern Ontario may not be viable options to assess habitat quality in the Northwest Territories. They may not even be useful in distant Ontario forests unless they exhibit similar characteristics. This is one major shortcoming of using this method, and the process of identifying indicators is still being researched, refined, and critiqued. Relatively few saproxylic beetle indicators have been identified and much more research is required.

Measuring habitat features is a much easier task than measuring saproxylic beetles (Hodge and Peterken, 1998; Grove, 2002a). Vegetation surveys are relatively straightforward, needing minimal equipment and minimal skill. Capturing beetles requires costly traps and lures, more time, and typically at least one entomologist who is skilled at taxonomic identification. Measuring CWD is considered an option for inferring saproxylic beetles present, but can be more difficult to evaluate in practice. CWD tends to follow a patchy distribution and may not be properly represented through the typical plot

surveys used. It is also difficult to distinguish the effects of CWD characteristics on saproxylic beetle diversity therefore complicating the possibility of inferring species present (Siitonen, 2001).

The European Environmental Agency has deemed the measurement of deadwood volume as an important aspect of forest management and is used as an indicator for the conservation value of forests (Noss, 1999; Bouget *et al.*, 2013). Many researchers have found the diversity of deadwood to be an important ecological predictor of saproxylic beetle diversity (Bouget *et al.*, 2013). However, until the complex relationships between saproxylic beetles and deadwood are completely understood it is suggested that surveys of insects should act as complementary studies of deadwood within a forest.

The Importance of Monitoring Insect Populations

Arthropods are the most diverse and numerous organisms on our planet, are drivers of many ecological processes, and provide multiple ecological services within an ecosystem (Kim, 1993). It is estimated that arthropods make up 91% of all living animals on the planet and 79% of global diversity, with an estimated 8 million species of insects (Kim, 1993; Groomsbridge, 1992). In spite of their known importance this group has large gaps in taxonomic knowledge. With specific habitat requirements, restricted mobility, and their presence in nearly every ecosystem, anthropogenic changes to the Earth are putting immense pressure on insects and putting many at risk (Groomsbridge, 1992; Kim, 1993). Focus on insects as important components of ecosystem management has only recently become integrated into some conservation initiatives and a greater emphasis on integrating insects into management plans is necessary (Kim, 1993). Insects occupy many ecological roles and many different levels within an ecosystem and therefore offer a cost-effective and easy way to monitor ecosystem health and biodiversity of the environment (Kim, 1993; Miller, 1993; Samways, 1993).

Insects are very specialized within the niches they occupy; therefore different species offer a range of ecological services (Kim, 1993). They are, however, typically overlooked by managers and policy makers whose primary focus instead tends to be on large mammals and birds (Groomsbridge, 1992; Hafernik Jr., 1992; Kim, 1993). This is in part due to limited knowledge regarding the role insects play within their habitat and a general negative perception within the public regarding insects (Howarth and Ramsay, 1991; Kim, 1993; Medeiros *et al.*, 2013). A conservation strategy based on insects is a difficult task due to their wide species diversity, gaps in taxonomic knowledge, difficulty in recognizing extirpation or extinction of species, and their variable habitat needs and niche environments (Kim, 1993). Instead of a species- or group-based conservation strategy, management should be focused on an ecosystem as a whole and sustaining the processes within (Soulé, 1991; Kim, 1993). This first requires a greater knowledge of all components of an ecosystem, including greater research of the insect fauna present.

Biodiversity surveys form the basis of ecosystem management (Kim, 1993). Conservation efforts cannot commence until biodiversity and related processes are identified and understood. Uncovering species at risk of extinction or extirpation from a habitat can help identify areas in need of the most attentive conservation efforts. Taxonomic surveys are typically completed to assess the insect species present within an ecosystem (Vane-Wright *et al.*, 1991; Kim, 1993) and can identify rare and threatened species.

If their natural histories are known then the ecological processes they both support and depend on can be the focus of management efforts. Because the processes insects are involved in typically form the backbone of an ecosystem and span across all levels of a habitat, supporting these species will ensure the survival of other resident species (Soulé, 1991; Kim, 1993). Additionally, their short life cycles make insects ideal for indicating rapid changes within an ecosystem and identifying species or groups most likely to reflect these changes will increase management effectiveness (Brown Jr., 1997).

Although some insect species are considered pests and/or invasive, they are simply organisms taking advantage of available ecological niches, typically those impacted by anthropogenic pressures (Kim, 1993). We have provided opportunities for invasives to enter ecosystems, spread throughout the environment, and establish themselves within disturbed habitats. Invasive species are often difficult to monitor and control, so how they fit into native ecosystems is important when determining how to address the issue and adapt management techniques. Before a management strategy is applied, the extent of an invasive species' interactions with other species and processes within a habitat should be known.

Taxonomic assessment and ecosystem classification are two preliminary actions necessary for ecosystem management (Kim, 1993). The three sites included in my study were previously classified by the city of Kitchener using an Ecological Land Classification (ELC) system. No information regarding insect fauna can be found for these areas as detailed insect inventories are seldom completed. The research presented here will provide a baseline inventory for saproxylic beetles that can be used in future comparative studies. Additionally, my research may unveil potential indicator families which the City of Kitchener can further explore for use in ecosystem monitoring.

The first goal of my study is to explore the state of current saproxylic beetle communities within the urban forests of Kitchener and encourage further insect monitoring which is vital to ecosystem management. This is intended to build upon the city's Urban Forestry Management Plan (City of Kitchener, 2017b) which cites dead trees as having no value to either the forest community or the community of Kitchener. The second goal of my work is to determine if certain families or functional groups reflect total beetle assemblage suggesting feasibility as indicators. Tilt's Bush is expected to have the most diverse assemblage due to the presence of both deciduous and coniferous trees and the park's larger size when compared with Breithaupt Park and Laurentian Wetland. Both taxonomic families and functional groups will be evaluated as possible indicators of overall saproxylic beetle populations, and by extension ecosystem integrity.

Chapter 2.0

Methodology

Study Sites

All study sites were located in the city of Kitchener, Ontario. Sites were selected based on information gathered via previous surveys and Ecological Land Classification (ELC) performed and provided by the City of Kitchener in 2015. Sites chosen were representative of different areas of the city, but were in urban environments; all were mainly surrounded by residential neighbourhoods with little connectivity to other parks. Three sites in total were selected: Breithaupt Park, Laurentian Wetland, and Tilt's Bush (Fig. 2.1). All of these sites were also infested by EAB, and collecting saproxylic beetle population data on these areas can improve future mitigation and eradication efforts by the City of Kitchener.

Breithaupt Park comprises a total of 19.38 ha, with a dominant forest community of sugar maple-beech deciduous forest covering a total of 46.5% (Fig. 2.2). Two transects spanning 60m spaced 50m apart were placed in the dry-fresh sugar maple-white ash and dry-fresh sugar maple-black cherry deciduous communities (Fig. 2.3).

Laurentian Wetland comprised a total of 19.26 ha, and the dominant forest community is deciduous thicket measuring 15.89% (Fig. 2.4). Two transects spanning 60m spaced 50m apart across the deciduous thicket (Fig. 2.5).

Tilt's Bush comprised a total of 37 ha with a dominant forest community of white cedar-conifer organic coniferous swamp type, measuring 31.81% (Fig. 2.6). Transects spanned across the sugar maple-white ash deciduous and the white cedar-conifer organic coniferous swamp types (Fig. 2.7).



Figure 2.1: A map of the City of Kitchener, Ontario showing the locations of Breithaupt Park, Laurentian Wetland, and Tilt's Bush. Saproxyllic beetle collection took place in these three urban parks from June-September 2015 to compare the abundance and diversity of populations.

Breithaupt Park Ecosystem Types

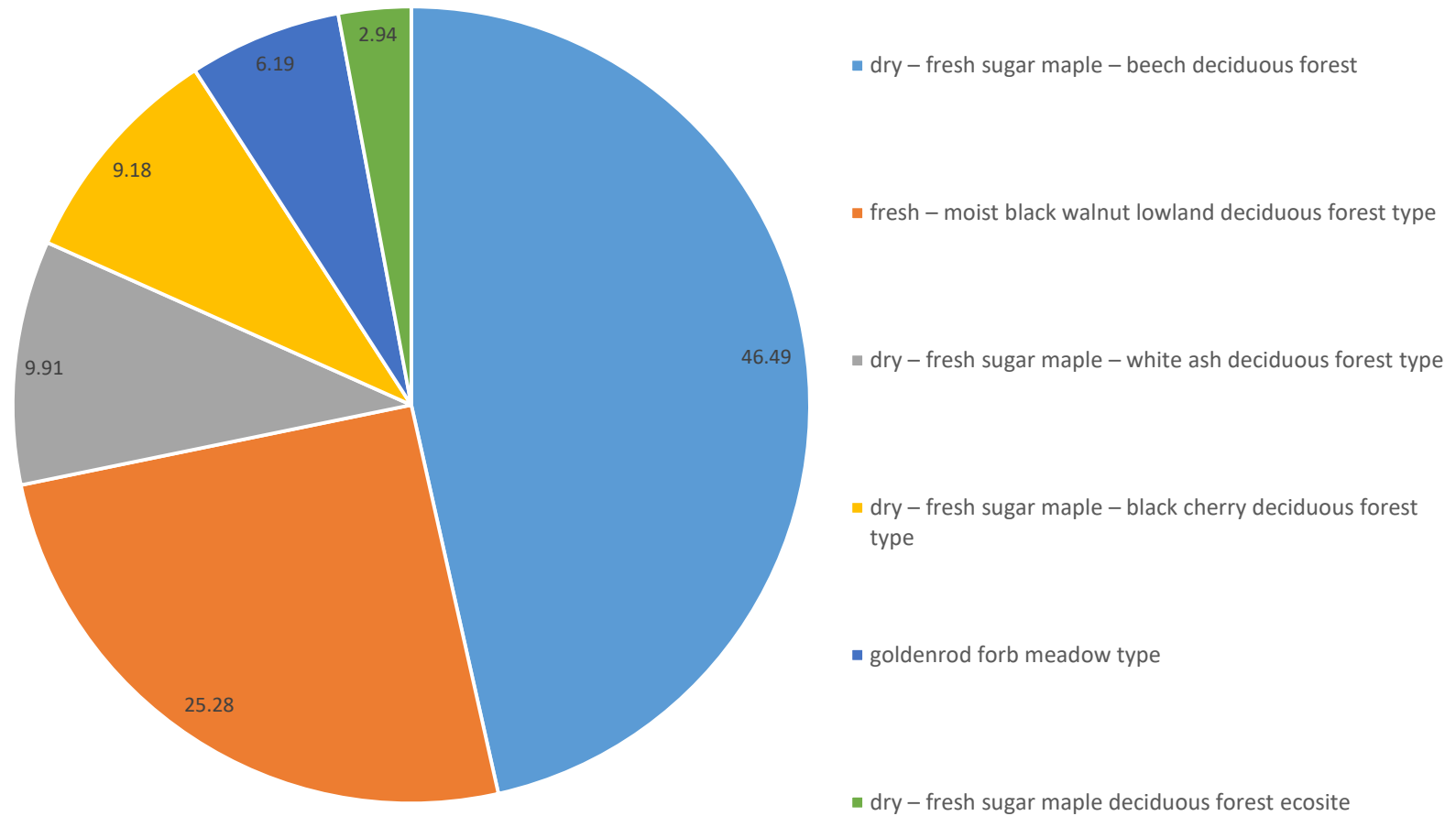


Figure 2.2: Percent abundance of ecosystem types of Breithaupt Park in Kitchener, Ontario. Data is based on surveys provided by the City of Kitchener in 2015.

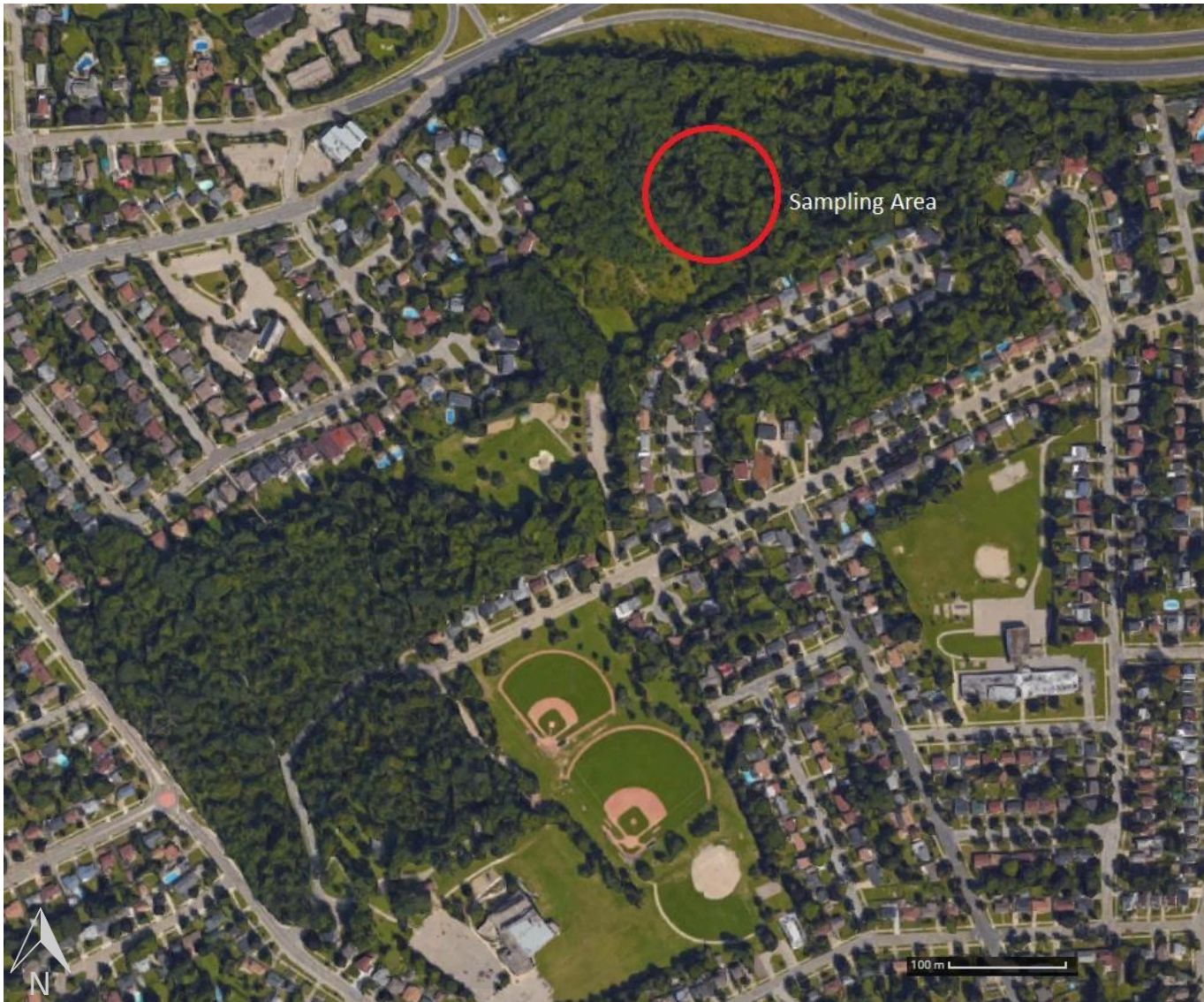


Figure 2.3: A map of Breithaupt Park in Kitchener, Ontario. The sampling area outlines where two 60m long transects of Lindgren funnel traps were placed 50m apart for the collection of saproxylic beetles. In this case, transects ran from SW to NE.

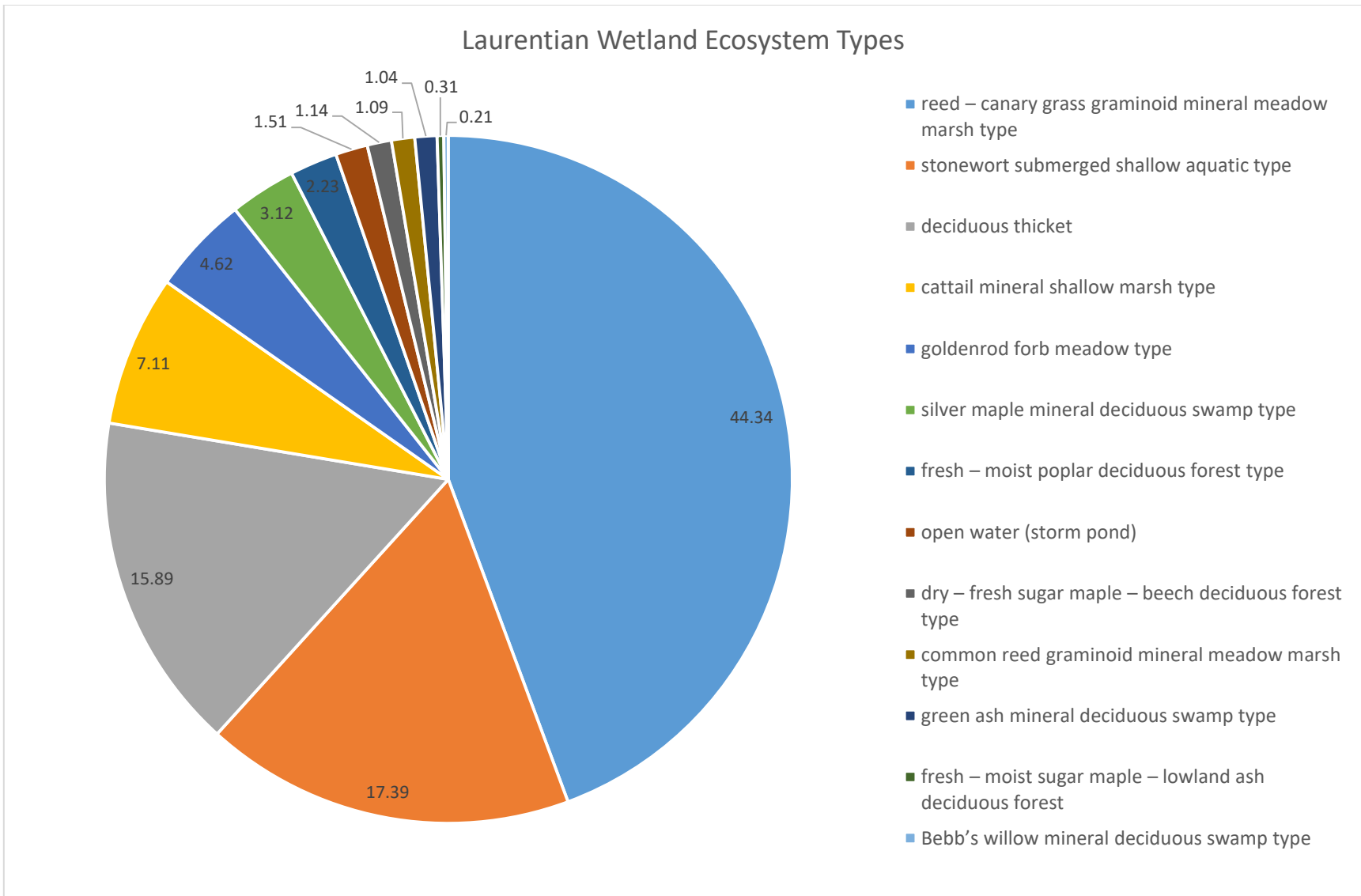


Figure 2.4: Percent abundance of ecosystem types of Laurentian Wetlands in Kitchener, Ontario. Data is based on surveys provided by the City of Kitchener in 2015.



Figure 2.5: A map of Laurentian Wetland in Kitchener, Ontario. The sampling area outlines where two 60m long transects of Lindgren funnel traps were placed 50m apart for the collection of saproxylic beetles. In this case, transects ran from ENE to WSW.

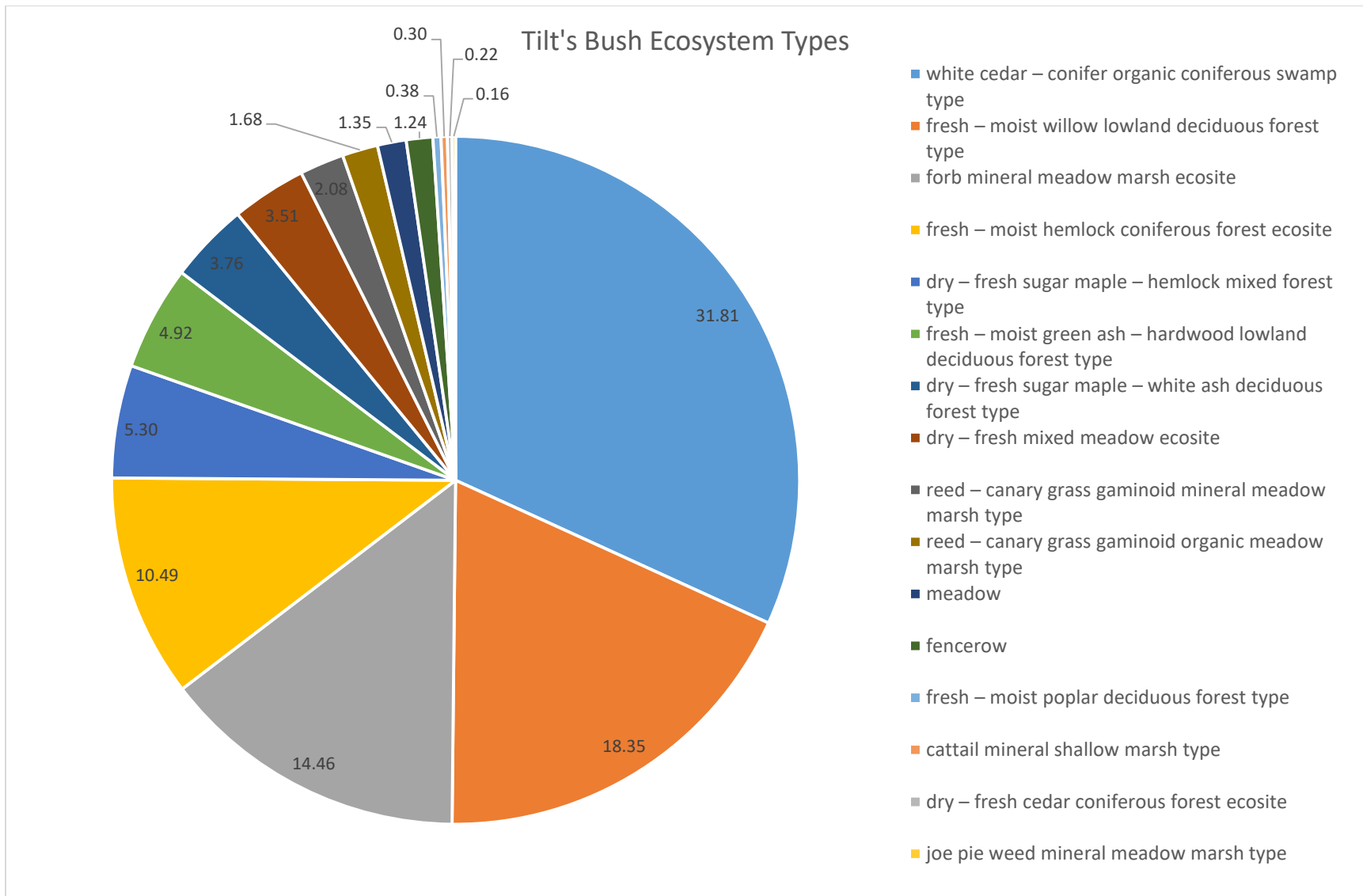


Figure 2.6: Percent abundance of ecosystem types of Tilt's Bush in Kitchener, Ontario. Data is based on surveys provided by the City of Kitchener in 2015.



Figure 2.7. A map of Tilt's Bush in Kitchener, Ontario. The sampling area outlines where two 60m long transects of Lindgren funnel traps were placed 50m apart for the collection of saproxylic beetles. In this case, transects ran from WSW to ENE.

Insect Sampling

Sampling took place from 1 June 2015 to 19 September 2015 to encompass the flight times of a variety of saproxylic beetle species (Morewood *et al.*, 2002; Francese *et al.*, 2008; Allison *et al.*, 2014; DiGirolomo and Dodds, 2014; Nadeau *et al.*, 2015). This was an attempt to maximize the number of beetles caught and gain a sample representative of total beetle populations.

Black Lindgren funnel traps (8-funnel) were used based on their success at attracting and capturing saproxylic beetles (Bashford, 2008; Hanula *et al.*, 2011; Brar *et al.*, 2012; Miller *et al.*, 2013; Allison *et al.*, 2014). The more funnels a trap has, the greater efficacy at catching beetles, and the dark silhouette created by the black material mimics that of a tree to the eyes of an insect (Morewood *et al.*, 2002; Miller and Crowe, 2011). Traps were hung approximately 15 m apart and 1.5 – 2.0 m above the ground, measured from the bottom of the collecting cup to appear as a standing tree to beetles (Morewood *et al.*, 2002; Hanula *et al.*, 2011; Allison *et al.*, 2014; Nadeau *et al.*, 2015; Rassati *et al.*, 2014) (Fig. 2.7). Traps were also placed 1.0 – 2.0 m from any tree trunk in an attempt to maximize visibility (Allison *et al.*, 2014).

Each collecting cup was filled with approximately 250 mL of 70% ethanol and several drops of dish soap (surfactant) to increase beetle capture rate. Wet collecting cups have been shown to be more effective at trapping insects than dry cups (McIntosh *et al.*, 2001; Bashford, 2008; Miller and Crowe, 2011; Graham *et al.*, 2012; Nadeau *et al.*, 2015). Every two weeks (Miller and Crowe, 2011; DiGirolomo and Dodds, 2014; Nadeau *et al.*, 2015) collecting cups at all sites were emptied into 1-L glass mason jars and refilled with ethanol.



Figure 2.8: A Lindgren-funnel trap hung in Tilt's Bush, a park located in Kitchener, Ontario. Traps were placed in two transects of 5 spanning 60m with transects placed 50m apart from June-September 2015. Each trap was approximately 1.5-2.0m from the ground, 1.0-2.0m from any tree trunk, and 15m from any other trap within the transect.

Two perpendicular transects of five traps were erected per sampling site, for a total of 30 traps. Transects were placed roughly 50-m apart from one another (DiGirolomo and Dodds, 2014; Rassati *et al.*, 2014). It is possible for small sampling areas (100m²) to convey similar information regarding saproxylic beetle assemblages when compared to a larger sampling area (Økland *et al.*, 1996). Given the economic restrictions of the present study, the selected spatial scale was deemed appropriate. Transects began within roughly 5.0 m of the edge of the forest and extended inwards due to increased capture rate at the edges and centres of forests (Francese *et al.*, 2008; Graham *et al.*, 2012; Nadeau *et al.*, 2015). Some species preferentially seek sunny areas on tree trunks and foliage, whereas others prefer the more consistent microclimates found in the interior of the forest (Vodka and Cizek, 2013; Herms and McCullough, 2014). In particular, buprestids and cerambycids have been identified as preferring edge habitats rather than forest interior (Wermelinger *et al.*, 2002; Wermelinger *et al.*, 2007)

A single low release ethanol packet was attached to each trap on the day of mounting. One ethanol packet lasts approximately 9 months and did not need to be replaced throughout the duration of the study. Ethanol is used as a general attractant for saproxylic beetles by mimicking the chemicals released by dying and recently dead trees (Millar *et al.*, 1986; Ytsma, 1989; Schroeder, 1992; Hammond *et al.*, 2001; Rassati *et al.*, 2014). Beetles that colonize distressed or freshly dead trees, such as scolytinae, cerambycidae, and buprestidae, are more attracted to ethanol than others (Hammond, 2001). A fuscumole acetate bubble was also attached to each trap at the time of mounting, with a new bubble

added every four weeks, providing a total of four per trap by the end of sampling period. Old bubbles were left on the traps to maximize pheromone release. This semiochemical increases the attraction of saproxylic beetles, especially cerambycids. Male beetles within the family cerambycidae release semiochemicals, including fuscumol acetate, as aggregation pheromones on resources suitable for reproduction (Mitchell *et al.*, 2011).

Samples were strained in the laboratory using flour sackcloth and mesh sieves, and refilled with fresh ethanol. Insects in each sample were identified and sorted by family using the taxonomic keys found in *Insects: Their Natural History and Diversity* (Marshall, 2006). Family totals for each individual trap were pooled to obtain count data per site on each date sampled.

Statistical Analyses

The measurement of biodiversity, which is comprised of species diversity and abundance, is common practice among ecologists (DeJong, 1974; Morris *et al.*, 2014). These factors are capable of providing a glimpse into the productivity of an ecological system (Peet, 1974) and examining the patterns of abundance can reveal important community responses to ecological pressures (Kempton, 1979). Many different indices and equations have been created to help evaluate diversity and abundance and it is important to choose the correct methods (DeJong, 1974; Morris *et al.*, 2014). For the current study, a variety of analyses were performed on beetle families, including: accumulation curves, abundance curves, Kruskal-Wallis analyses, Jaccard's Similarity Coefficient, Simpson and Shannon-Wiener Indices, and Pearson's Correlation Coefficient. All of these measures were completed at family level for each community. Species within each family can vary greatly, but each family has a general life history and larval guild that were relied upon for this study.

All statistics were performed using Microsoft Excel because analyses were simple and manageable using this program. Where applicable, an alpha level of 0.05 was used. Count data was compared between sites and for total beetle counts. Data was transformed into percentage abundance for further comparison. Accumulation curves were calculated for each site using pooled count data for a total of 8 collection dates. Accumulation curves act as a visual aid when attempting to determine if beetle populations have been adequately sampled. If curves reach an asymptote it can be assumed the population is adequately represented (Buse *et al.*, 2010).

Species richness is one of the most widely used and straightforward methods of evaluating diversity within an ecosystem (Peet, 1974, Morris *et al.*, 2014). It can be loosely defined as the "relative wealth of species in a community" (Peet, 1974). In the present study, family richness was represented by the number of families within each site. Species counts are one of the simplest ways of illustrating richness (Peet, 1974), and in this instance, family counts are used for the initial assessment of biodiversity.

A Kruskal-Wallis test adjusted for ties was performed on the count data without transformation. This test is used to convey general differences in family richness based on compared samples but will not identify where this significance exists. Here, it was used to determine whether significant differences existed between insect communities in each site sampled.

Jaccard's Similarity Coefficient - The Jaccard index is used to assess the compositional similarity between assemblages using presence-absence information (Chao *et al.*, 2005). It is most commonly used in species association analyses when the species overlap between two communities is desired (Real, 1991; Chao *et al.*, 2005). Samples were analyzed in pairwise comparisons using the following formula:

$$J = \frac{C}{A + B - C}$$

where C is the number of families shared, A is the number of families present in sample a, and B is the number of families present in sample b.

The coefficients take on a value between 0 and 1, where 1 indicates the same families present in each community (complete family overlap), and 0 representing no shared families (no family overlap). This analysis will help understand the similarity of families between each pair of sites and therefore whether beetle assemblages differ between the three.

This calculation does not encompass the abundance of species within each assemblage. Presence/absence binary information is used to calculate the similarity coefficient between each pair of assemblages (Chao *et al.*, 2005). Therefore this index is most useful when the abundance of species/families is not the driver of community differences (Jonsell *et al.*, 1998). A more diverse beetle community is expected in Tilt's Bush, due to its larger size and tree composition, which the Jaccard Similarity Coefficient will show if true; pairwise comparisons with Tilt's Bush will have a value closer to 0 indicating fewer shared families. This would suggest Tilt's Bush may have more families present than Breithaupt Park and Laurentian Wetland. Additionally, this index can prove unreliable if a large number of rare species or families (1-2 individuals) are present within the data. If this is true of the data, then further analyses to compare similarity are necessary (Chao *et al.*, 2005).

Diversity Indices - Percentage abundance was calculated for each site and plotted in family abundance curves. These figures provide a visual aide by displaying the evenness across families in each community. The population within Breithaupt Park, Laurentian Wetland, and Tilt's Bush can be visualized and compared via abundance curves. These relationships are further explored through the use of the Simpson's Index. Both the Simpson and Shannon-Wiener Indices can be considered measures of assemblage heterogeneity (Peet, 1974). These indices are commonly used together when evaluating biodiversity and provide a more in-depth understanding of population dynamics (Morris *et al.*, 2014) and in this study aid in the comparison of beetle composition within each site.

The Simpson index reveals whether some families show dominant abundance over others within an assemblage (Villa-Castillo and Wagner, 2002; Stephens and Wagner, 2006, Morris *et al.*, 2014). The number of individuals within each group and the concentration or strength of abundances, i.e. evenness, are taken into account (Peet, 1974).

The distribution of abundance within a community can be a more important indicator of ecosystem disturbances than the overall biodiversity. An established, stable community is typically comprised of only a handful of families with high abundances and the majority of the community comprised of moderately common and rare families. This indicates ecosystem processes and species populations are in a stable state with little change. Contrary to this expected abundance pattern, a post-disturbance community shows uneven distribution of abundance even when biodiversity has reached pre-disturbance levels. Previous studies have demonstrated such communities exhibit a large number of highly abundant species and rare species, with few in between (Kempton, 1979).

The Simpson index formula calculates the probability that two randomly selected individuals will belong to different families and is particularly sensitive to abundant families (DeJong, 1974; Shimatani, 2001; Morris *et al.*, 2014). It takes into account the number of species or families present (i.e. species richness), and the number of individuals within each species/family (i.e. evenness). As the evenness of an assemblage increases, meaning the distribution of abundance is not limited to a small number of families, the diversity similarly increases (DeJong, 1974). The calculated value D falls between the values of 0 and 1, with 0 representing high diversity and 1 representing zero diversity. D is calculated using the following formula:

$$D = \sum \frac{n_i(n_i - 1)}{N(N - 1)}$$

where n_i is the number of individuals in family i , and N is the total number of individuals in the sample. It is considered more intuitive when this value is subtracted from 1, known as Simpson's Index of Diversity. With this adjustment, richness and evenness are increasing as $1-D$ approaches 1 (Peet, 1974). The resulting values will indicate if any of the sites have an asymmetrical distribution of rare and common families. All three sites are under constant anthropogenic pressures and further urban development is ongoing throughout the City of Kitchener. Due to this, it is possible sites will have several rare and highly abundant families suggesting unstable communities.

The Shannon-Wiener Diversity Index was used to determine the diversity per treatment and has been referred to as an index of equitability (DeJong, 1974; Peet, 1974; Villa-Castillo and Wagner, 2002; Stephens and Wagner, 2006). Similar to the Simpson Index of Diversity this index assumes that the greater the number of species and their evenness the greater the diversity within a community (Peet, 1974). The Shannon-Wiener index is sensitive to both rare and abundant species and will put more weight on their evenness (Peet, 1974; Morris *et al.*, 2014). This method is best suited for data sets with large sample sizes and a small proportion of rare families (Shannon and Weaver, 1949; Jennings *et al.*, 2017). It is calculated as follows:

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

where p_i is the percent abundance in family i .

Using percentage abundance, the calculated H' will give the probability a selected individual will belong to a particular family. The greater the number of families and evenness within the sample, the greater the uncertainty the individual will belong to a particular family, therefore the greater the diversity (DeJong, 1974). As the H' value increases from 0, diversity and evenness similarly increase. There is no upper limit to Shannon's H' , but it is typically found to be 5. To compare H' between sites, pairwise Hutcheson t-tests were used. Tilt's Bush was expected to have the greatest H' value compared to Breithaupt Park and Laurentian Wetland based on the tree composition and park size.

Correlation Analyses - The use of indicator species is a widely used method of evaluating and monitoring ecosystems. It is, however, criticized because the process of identifying indicators can be costly, time consuming, and the resulting indicators may not be widely applicable. Identifying indicators within an assemblage can be done in several ways. For these data Pearson's correlation coefficient was used to identify potential indicators by exploring the relationships between beetle families (Ohsawa, 2010).

The trend in abundance over the entire sampling period was used for each family at all sites. The count data for all families on every collection date was compared to the total abundance of all beetles combined (minus the family being compared), between each individual family, and between each family and its functional group (Ohsawa, 2010). This information not only contributed to a more thorough understanding of saproxylic beetle dynamics, but also acted as an indicator of the types of resources available to these insects.

Comparing a single family to the total abundance of beetles combined (excluding the family being compared) relays information regarding how a family's population trend reflects that of saproxylic beetles within the site. If a family is found to be positively correlated, then it is considered to be potentially useful as an indicator of overall saproxylic beetle abundance. If the same indicator is identified at all sites, targeting this group/family would allow managers in Kitchener to monitor trends in saproxylic beetles at all parks. Similarly, exploring the correlation between individual families can reveal relationships that can aid in ecosystem monitoring, and thus lead to a more comprehensive understanding of saproxylic beetle dynamics within urban managed forests.

Finally, determining the relationships between an individual family and the total beetles within their functional group can reveal indicators for resources which can help keep track of ecosystem changes. Functional groups were defined by the larval habitat guilds for each family (Grove, 2002b; Gandhi and Herms, 2010; Nadeau *et al.*, 2015). These categories included those that feed on wood (xylophagous), those that feed on fungus (mycetophagous), and those that are predators. Categorization was based on available information about families within the literature (Grover, 2002a; Grove, 2002b; Hammond *et al.*, 2004; Jacobs *et al.*, 2007; Ohsawa, 2010; Lachat *et al.*, 2012; Toigo *et al.*, 2013; Evans, 2014). If a large number and diversity of xylophagous beetles are present then there is most likely an abundance and diversity of dead trees and woody debris. If the diversity and abundance of mycetophagous beetles are present, it can be inferred that there is a wide variety of decaying wood. The presence of predatory families indicates a suitable prey population and their associated habitats. By determining the families

that correlate significantly to each functional group, they can be used as indicators to monitor groups and therefore specific resources.

Each insect group requires a different stage of tree death/decay and can provide information regarding different types of resources available in an ecosystem. Because the use of larval resources is not uniform between species in the families elateridae, staphylinidae and zopheridae in Ontario, these groups were not included in any of the functional groups. In this way, families whose larvae are more strongly associated with each guild would be expected to provide more significant and useful results. Two-tailed t-tests were used determine which correlations were significantly different and therefore more suitable as indicator families.

Results

A total of 43 saproxylic beetle families and 10,016 individuals were identified (Table 3.1). Overall curculionidae, comprised mainly of the subfamily scolytinae, was the most abundant family (38.5%) collected. Scolytinae is of particular interest due to their habitat requirements compared to the rest of curculionidae. Although labelled curculionidae throughout analyses, the majority of these individuals were scolytines (Table 3.2). Nitidulidae (13.9%) and Itrididae (10.7%) were the next most abundant. Curculionidae was the most abundant family in both Breithaupt Park and Laurentian Wetland (43.2% and 45.5% respectively), and curculionidae was only second to nitidulidae in Tilt's Bush (23.5% and 31.6% respectively).

Laurentian Wetland contained the highest percentage of all beetles found (36.1%) followed by Breithaupt Park (34.2%) and Tilt's Bush (29.5%). Not all families were present at all sites; Breithaupt Park contained 36, while Laurentian Wetland and Tilt's Bush both contained 37 out of the 43 families. Anthribidae, passandridae, scaptiidae, and stenotrachelidae were only present in Breithaupt Park, and ptilodactylidae was only present at Tilt's Bush. No families were unique to Laurentian Wetland.

Regionally rare species (containing 1-2 individuals overall) included anthribidae, lucanidae, scaptiidae, and ptilodactylidae. Two individuals in total were identified for each of the former three families (0.0189%), and only one individual belonging to ptilodactylidae (0.00943%) was identified. A few additional families were found to be rare at specific sites (Table 3.3) but this was not explored further. This follows the expected abundance pattern typically found in a stable community, whereby only a handful of families are in high abundance and the majority of the community is comprised of moderately common and rare families (Kempton, 1979).

The total accumulation curve reached an asymptote by the final sampling date, suggesting an adequate sample of beetle populations overall (Fig. 3.1). Accumulation curves for each site revealed that Tilt's bush and Laurentian Wetland were more adequately sampled than Breithaupt Park. The Tilt's bush curve reaches more of an asymptote than Laurentian Wetland and Breithaupt Park. Breithaupt Park appears to decrease in beetle abundance by the seventh week of sampling, but abundance increases in the eighth week along with the slope of the curve suggesting that these samples were less representative of beetle populations than the other two sites. The Kruskal-Wallis test showed no significant difference between sites ($p=0.492$, $H=2.41$). This suggests there were no differences in family assemblages between sites and/or treatments.

Table 2.1: Count data and percentage abundance for saproxylic beetles caught in three parks (Breithaupt Park, Laurentian Wetland, and Tilt’s Bush) in Kitchener, Ontario combined over the entire sampling period (June-September 2015). The total number of saproxylic beetle families is 43 with a total of 10,016 individuals. Families are ranked from highest number of individuals to lowest.

Ranked Family	# of Individuals	% Abundance	Ranked Family	# of Individuals	% Abundance
Curculionidae	3764	37.58	Cleridae	16	0.16
Nitidulidae	1460	14.58	Trogossitidae	14	0.14
Latridiidae	1081	10.79	Carabidae	13	0.13
Corylophidae	916	9.15	Lampyridae	12	0.12
Eucnemidae	452	4.51	Melandryidae	12	0.12
Staphylinidae	452	4.51	Mycetophagidae	12	0.12
Laemophloeidae	393	3.92	Ptiliidae	11	0.11
Zopheridae	215	2.15	Endomychidae	10	0.10
Elateridae	175	1.75	Cryptophagidae	9	0.09
Cerambycidae	167	1.67	Oedemeridae	8	0.08
Mordellidae	140	1.40	Salpingidae	7	0.07
Tenebrionidae	120	1.20	Buprestidae	6	0.06
Throscidae	101	1.01	Bostrichidae	5	0.05
Leiodidae	99	0.99	Stenotrachelidae	5	0.05
Anobiidae/Ptinidae	86	0.86	Cucujidae	4	0.04
Histeridae	55	0.55	Clambidae	3	0.03
Ciidae	41	0.41	Passandridae	3	0.03
Cerylonidae	32	0.32	Anthribidae	2	0.02
Erotylidae	32	0.32	Lucanidae	2	0.02
Scarabaeidae	32	0.32	Ptilodactylidae	1	0.01
Rhipiphoridae	29	0.29	Scraptiidae	1	0.01
Tetratomidae	18	0.18			
			Total	10016	100

Table 2.2: Percentage abundance of Scolytinae within Curculionidae caught in three urban parks (Breithaupt Park, Laurentian Wetland, and Tilt's Bush) from June-September 2015 in Kitchener, ON. Shown for each sampling site and for total counts (all samples combined) demonstrating the high composition of Scolytinae at each site. The total number of Curculionids was 61, with 3703 Scolytines.

	Curculionidae	Scolytinae	Total	% Scolytinae
Breithaupt Park	14	1401	1415	99.0
Laurentian Wetland	39	1616	1655	97.6
Tilt's Bush	8	686	694	98.8
Total	61	3703	3764	98.4

Table 2.3: Count data of rare families found in each sampling site and totalled over three sites within Kitchener, Ontario: Breithaupt Park, Laurentian Wetland, and Tilt's Bush. Sampling took place from June-September 2015. A family was considered rare if only 1 or 2 individuals were identified. Families with asterisks were considered rare at a regional scale because 2 or less individuals were found overall.

Family	Breithaupt Park	Laurentian Wetland	Tilt's Bush	Total
Anthribidae	2			2*
Bostrichidae	3	1	1	5
Buprestidae	2	2	2	6
Carabidae		2	11	13
Clambidae		1	2	3
Cryptophagidae	4	3	2	9
Cucujidae	6	1		7
Erotylidae	23	7	2	32
Lucanidae	1	1		2*
Melandryidae	13		1	14
Mycetophagidae	4	2	7	13
Oedemeridae		1	7	8
Ptiliidae		1	10	11
Ptilodactylidae			1	1*
Salpingidae	3	1	4	8
Scarabaeidae	24	2	12	38
Scraptiidae	2			2*
Total Rare Families	4	11	7	4

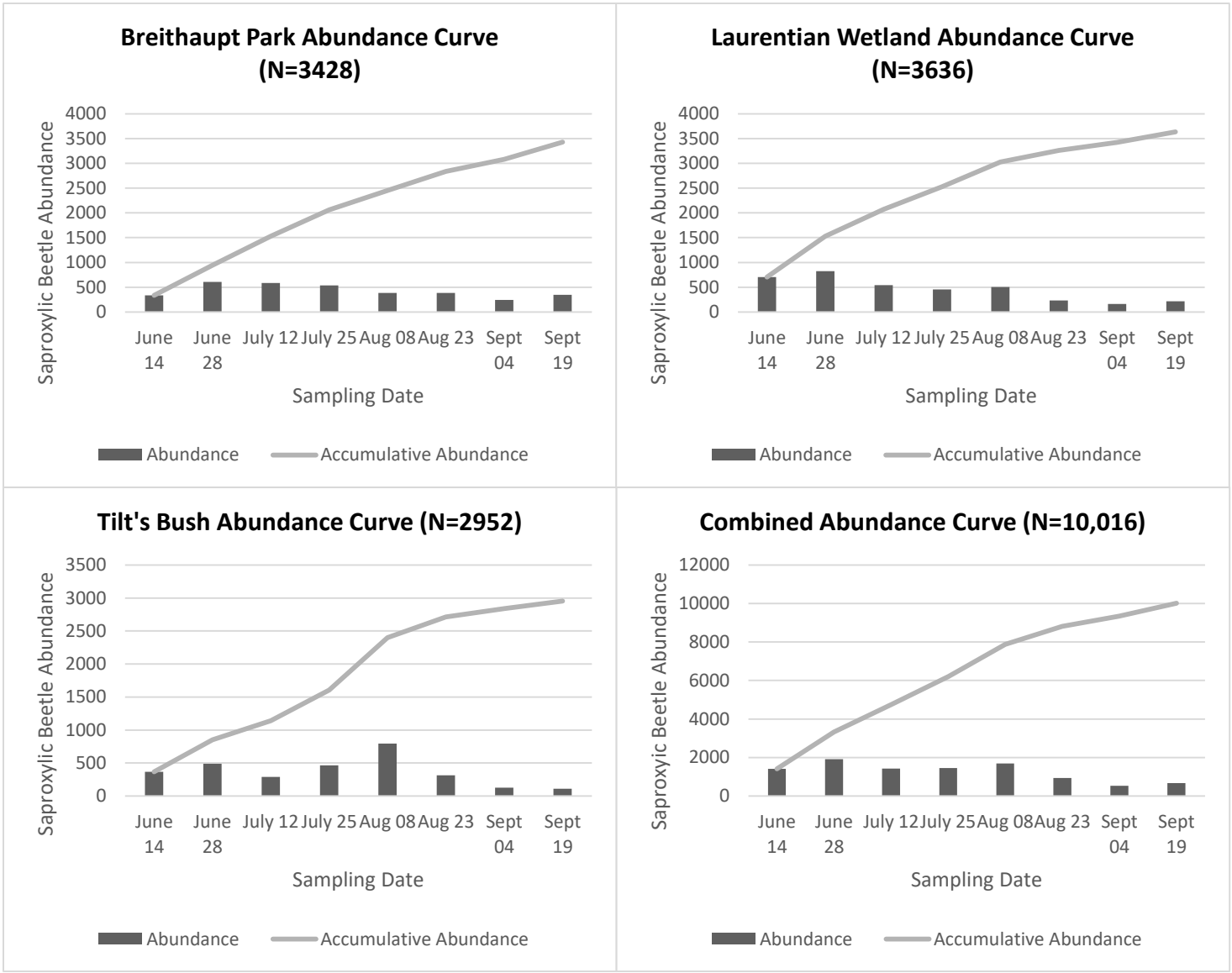


Figure 2.9: Family accumulation curves for three urban parks in Kitchener, Ontario and for combined counts based on saproxylic beetle sampling data obtained from June-September 2015. Parks include Breithaupt Park, Laurentian Wetland, and Tilt's Bush. These curves illustrate the ability of each sample to accurately represent the beetle populations. The more complete the sample, the more each curve will reach an asymptote at the end of the sampling period.

A diversity of xylophagous, fungivorous, and predatory families were collected from all sites indicating the presence of available resources for each group, including dead wood, tree-rotting fungus, and saproxylic prey (Fig. 3.2). In total, fungivorous beetles were the most abundant (51.2%). Both Breithaupt Park and Laurentian Wetland sites were dominated by xylophagous beetles (52.5%). Tilt's Bush was comprised mainly of fungivorous beetles (64.6%), followed by xylophagous (33.1%) and predatory (2.3%) beetles. Elateridae, staphylinidae, and zopheridae were excluded from these calculations due to the diversity of their larval habitat guilds. Based on this information, Tilt's Bush has the greatest composition of fungivores and predators compared to the other sites, and Laurentian Wetland and Breithaupt Park have an equally high abundance of xylophagous beetles.

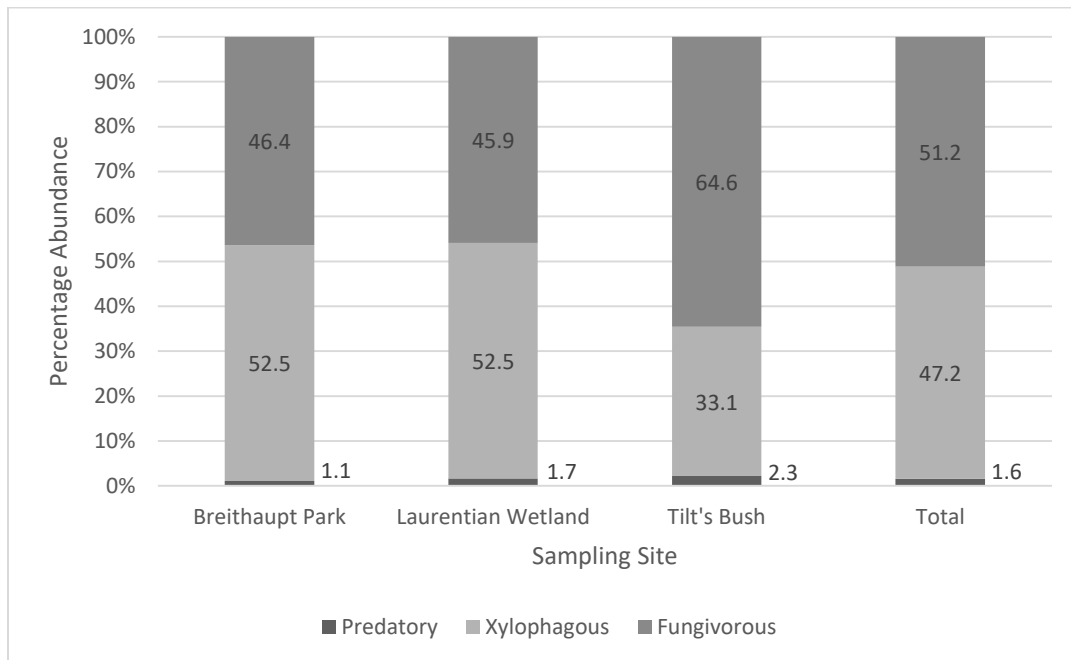


Figure 2.10: Percentage abundance of larval feeding guilds (i.e. functional groups) shown within three parks in Kitchener, Ontario (Breithaupt Park, Laurentian Wetland, and Tilt's Bush) and for the total individuals collected over the study period (June-September 2015). Adults were collected during this timeframe and functional groups estimated based on beetle family.

Functional groups were compared between all sites using Kruskal-Wallis tests. No significant results were found (Table 3.5) signifying no differences in functional group assemblages. This suggests available resources do not significantly differ between the three sites; therefore, the differences in size and tree composition between sites may not be impacting the local saproxylic beetles.

Table 2.4: Kruskal-Wallis tests for xylophagous, fungivorous, and predatory larval guilds (i.e. functional groups) collected in three parks in Kitchener, Ontario. Sites included Breithaupt Park (BP), Laurentian Wetland (LW), and Tilt’s Bush (TB), and the numbers of beetles were compared between the three for each group. Adults were captured from June-September 2015. No significance was found at $\alpha=0.05$.

Xylophagous		
Site	H	p
BP*LW*TB	1.30	0.730

Fungivorous		
Site	H	p
BP*LW*TB	3.90	0.272

Predatory		
Site	H	p
BP*LW*TB	2.03	0.566

Jaccard’s Similarity Coefficient - All three pairwise comparisons resulted in similarity coefficients over 0.600 (Table 3.6). Laurentian Wetland and Tilt’s Bush had the most similar assemblages ($J=0.875$), Breithaupt Park and Tilt’s Bush were the least similar ($J=0.682$), and Breithaupt Park and Laurentian Wetland fell between the two ($J=0.721$). In assemblages with numerous rare species, possible unrepresented species, and vastly different assemblage sizes, the Jaccard index underestimates true similarity (Chao *et al.*, 2005; Buse *et al.*, 2010). All three sites had a similar number of families collected and the presence of few rare families. These correlation coefficients provide a general estimate of assemblage similarity between pairs of sites.

Table 2.5: Jaccard Similarity Coefficient values for pairwise comparisons between three sites in Kitchener, Ontario: Breithaupt Park (BP), Laurentian Wetland (LW), and Tilt’s Bush (TB). Adult saproxylic beetles were sampled from June-September 2015. Families were compared between each site to determine the amount of differentiation between beetle assemblages.

	Shared	Total	J
BP/LW	31	43	0.721
BP/TB	30	44	0.682
LW/TB	35	40	0.875

Diversity Indices - Family abundance curves demonstrated a somewhat heterogeneous dispersal of individuals within families and moderate evenness in all sites (Fig. 3.3). Upon visual assessment, each site appears to contain very few families with high abundance and a large number of moderately abundant to rare families, which is a commonly observed abundance pattern for a stable community.

Both the Simpson Index of Diversity and the Shannon-Wiener Diversity Index suggested moderate diversity in all sites (Table 3.7). Tilt's Bush was the most diverse ($D-1=0.825$, $H'=2.25$) followed by Breithaupt Park ($D-1=0.771$, $H'=2.08$). Laurentian Wetland was found to be the least diverse by both indices ($D-1=0.757$, $H'=2.04$). The pairwise t-tests revealed no significant difference in the Simpson values between any of the sites. These results are not unexpected, given the differences in abundance observed when visually assessing abundance curves. The pairwise Hutcheson t-tests between Shannon's diversity values showed significant difference between Breithaupt Park and Tilt's Bush ($p<0.001$) and between Laurentian Wetland and Tilt's Bush ($p<0.001$). No significant difference was found between Breithaupt Park and Laurentian Wetland.

Correlation Analyses - Relationships were considered significant if $r > +0.707$ or $r < -0.707$ at all three sites. Families that met these requirements were deemed possible indicators. When considered at the site level, no family was consistently significantly correlated with the overall population of saproxylic beetles. However, Cerambycidae was the most highly correlated of all the families ($r_{BP}=0.926$, $r_{LW}=0.854$, $r_{TB}=0.677$). When calculated at the regional scale (i.e. comparing total cerambycids to total saproxylic abundance), seven different families were considered significant (Table 3.8). Corylophidae showed the strongest regional correlation to total beetle abundance (Fig. 3.4) followed by cerambycidae (Fig. 3.5).

When comparing abundance of individual families, two significant correlations were found spanning across all sites. Curculionidae showed a positive significant correlation with cerambycidae at each site ($r_{BP} = 0.932$, $r_{LW} = 0.726$, $r_{TB} = 0.751$) and at the regional scale ($r=0.857$) (Fig. 3.6). Cerambycidae was negatively correlated with corylophidae at each site ($r_{BP} = -0.719$, $r_{LW} = -0.763$, $r_{TB} = -0.786$) and regionally ($r=0.735$) (Fig. 3.6).

Within each site, cerambycidae was positively correlated with the xylophagous functional group ($r_{BP} = 0.921$, $r_{LW} = 0.723$, $r_{TB} = 0.829$), as well as at a regional scale ($r=0.858$) (Fig. 3.7). As the number of cerambycids increased within each ecosystem, a similar increase was observed for the other xylophagous beetle families suggesting an overall increase in woody debris resources. No families were significantly correlated across all sites amongst the mycetophagous or predatory groups. Within the predator group however, Histeridae was the most strongly correlated, significantly in both Laurentian Wetland and Tilt's Bush ($r_{BP}=0.696$, $r_{LW}=0.874$, $r_{TB}=0.758$), as well as regionally ($r=0.790$) (Fig. 3.7).

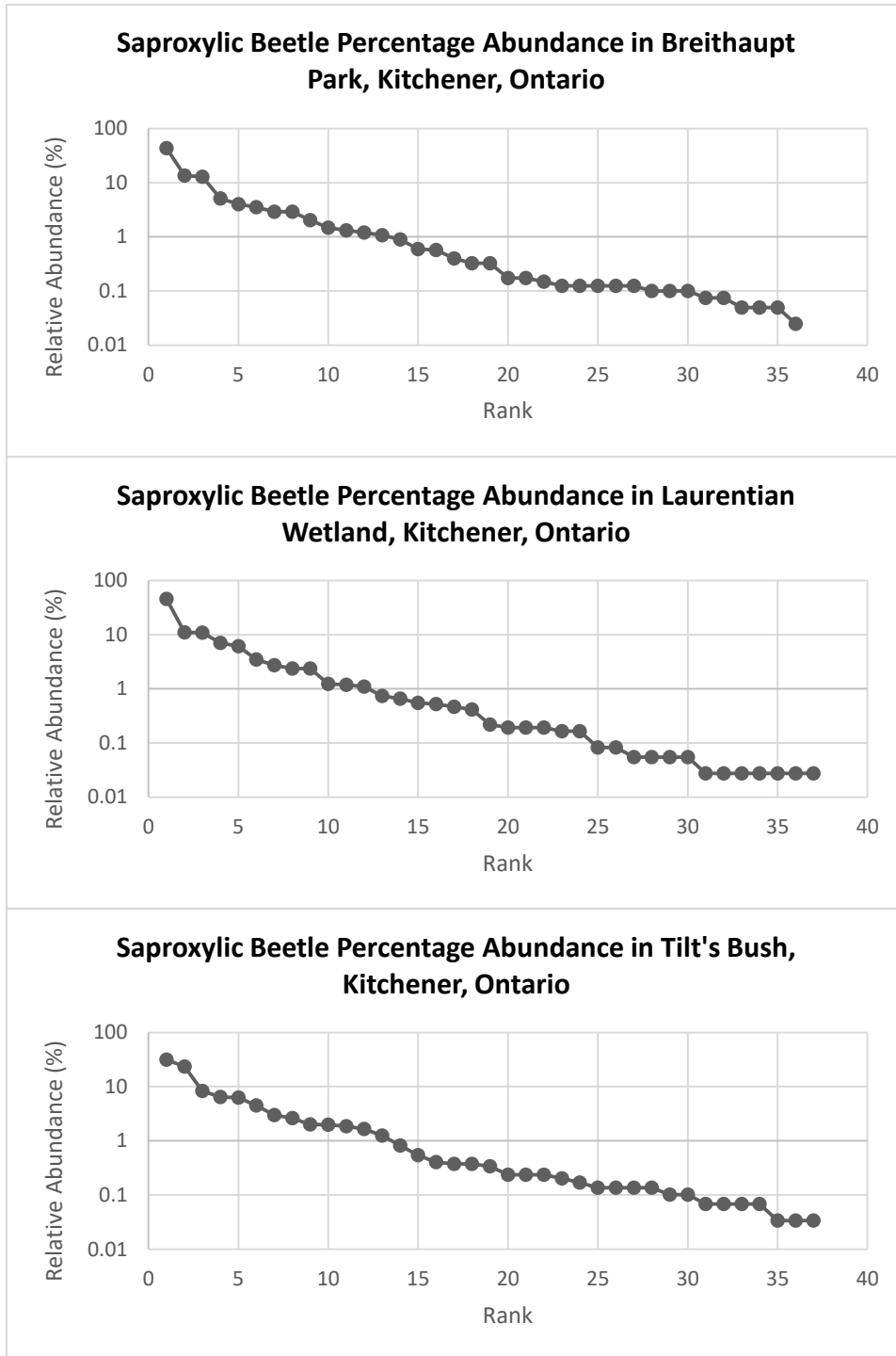


Figure 2.11: Percentage family abundance of saproxylic beetles sampled in June-September 2015 from three parks in Kitchener, Ontario: Breithaupt Park, Laurentian Wetland, and Tilt's Bush. Trends illustrate the heterogeneity of percent abundances for each sampling site. Beetle families are ranked from high abundance to low abundance. A low number of families with high abundance is found in typical communities, which these three figures exhibit.

Table 2.6: Shannon-Wiener and Simpson Diversity Indices for pairwise comparisons of saproxylic beetle assemblages in three parks in Kitchener, Ontario: Breithaupt Park (BP), Laurentian Wetland (LW), and Tilt’s Bush (TB). Beetles were sampled from June-September 2015. Significance at $\alpha=0.05$ indicated by *. As Shannon’s H' increases from 0 (and approaches 5), diversity and evenness increase. As the Simpson’s value increases from 0 to 1, diversity and evenness increase.

Site	Shannon H'	Pairing	t-test	p	Simpson 1-D	Pairing
BP	2.08	BP*LW	1.45	0.146	0.771	BP*LW
LW	2.04	BP*TB	5.09	3.75E-07*	0.757	BP*TB
TB	2.25	LW*TB	6.37	2.00E-10*	0.825	LW*TB

Table 2.7: Saproxylic beetle families exhibiting significant correlation with total beetle abundance based on adults collected from June-September 2015 in three parks in Kitchener, Ontario: Breithaupt Park, Laurentian Wetland, and Tilt’s Bush. Data from each park was pooled together for each sampling date so correlation could be determined at the regional scale. Pearson’s Correlation Coefficient was used to compare beetle count trends over the dates collected between each family and the sum of the remaining families.

Family	Abundance of Family	Total Abundance (all beetles collected minus Family)	Pearson’s Correlation Coefficient
Corylophidae	916	9100	-0.868
Cerambycidae	167	9849	0.835
Laemophloeidae	393	9623	-0.821
Staphylinidae	452	9564	0.800
Tenebrionidae	120	9896	0.760
Histeridae	55	9961	0.713
Elateridae	175	9841	0.708

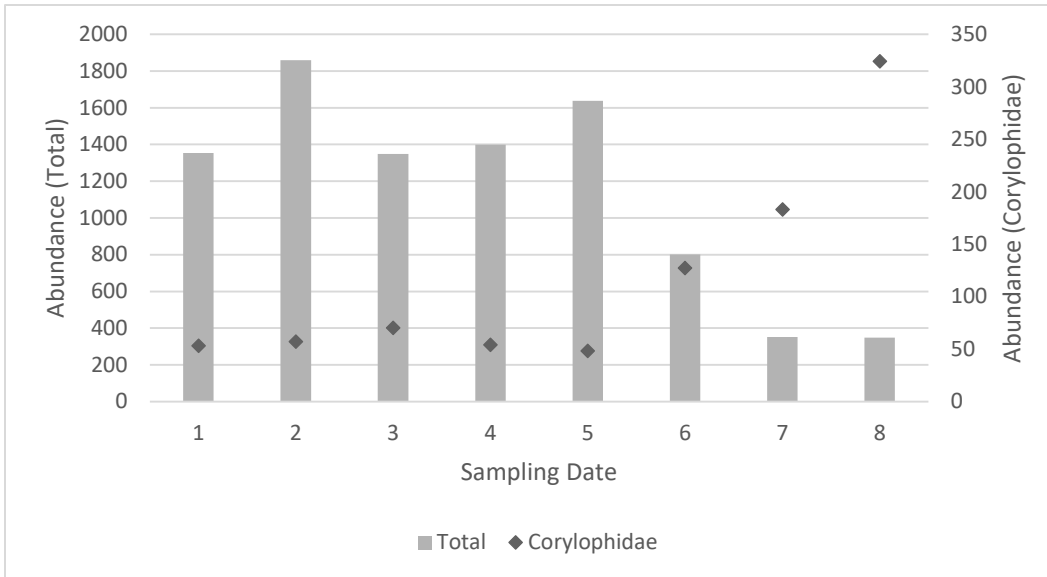


Figure 2.12: Correlation between the abundance of corylophidae and the total beetle abundance (excluding corylophidae) collected in Kitchener, Ontario from June-September 2015. The abundance trend seen in corylophidae negatively reflects that of the total beetle abundance.

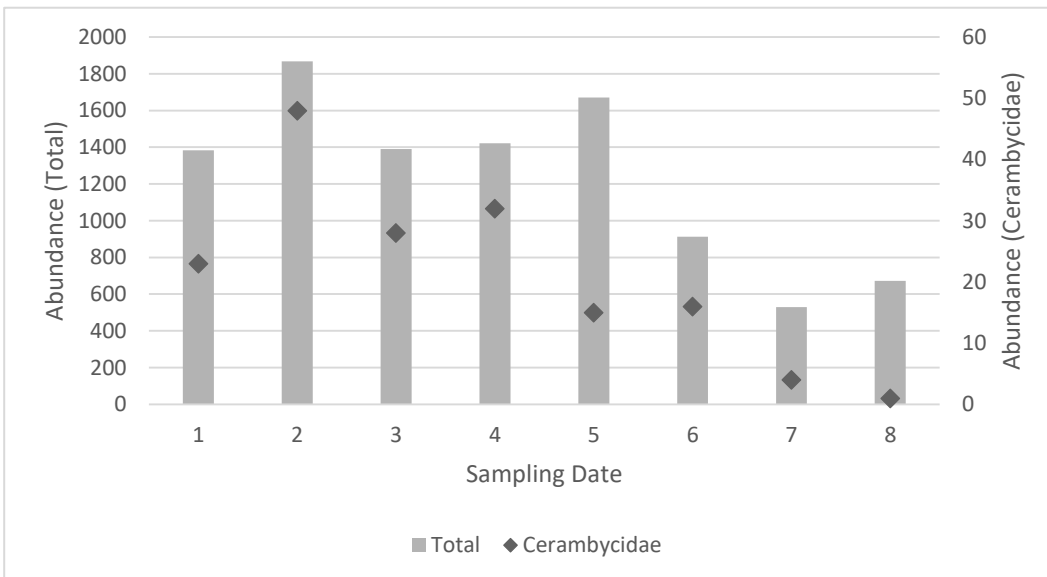


Figure 2.13: Correlation between the abundance of cerambycidae and total beetle abundance (excluding cerambycidae) collected in Kitchener, Ontario from June-September 2015. The abundance trend seen in cerambycidae positively reflects that of the total beetle abundance.

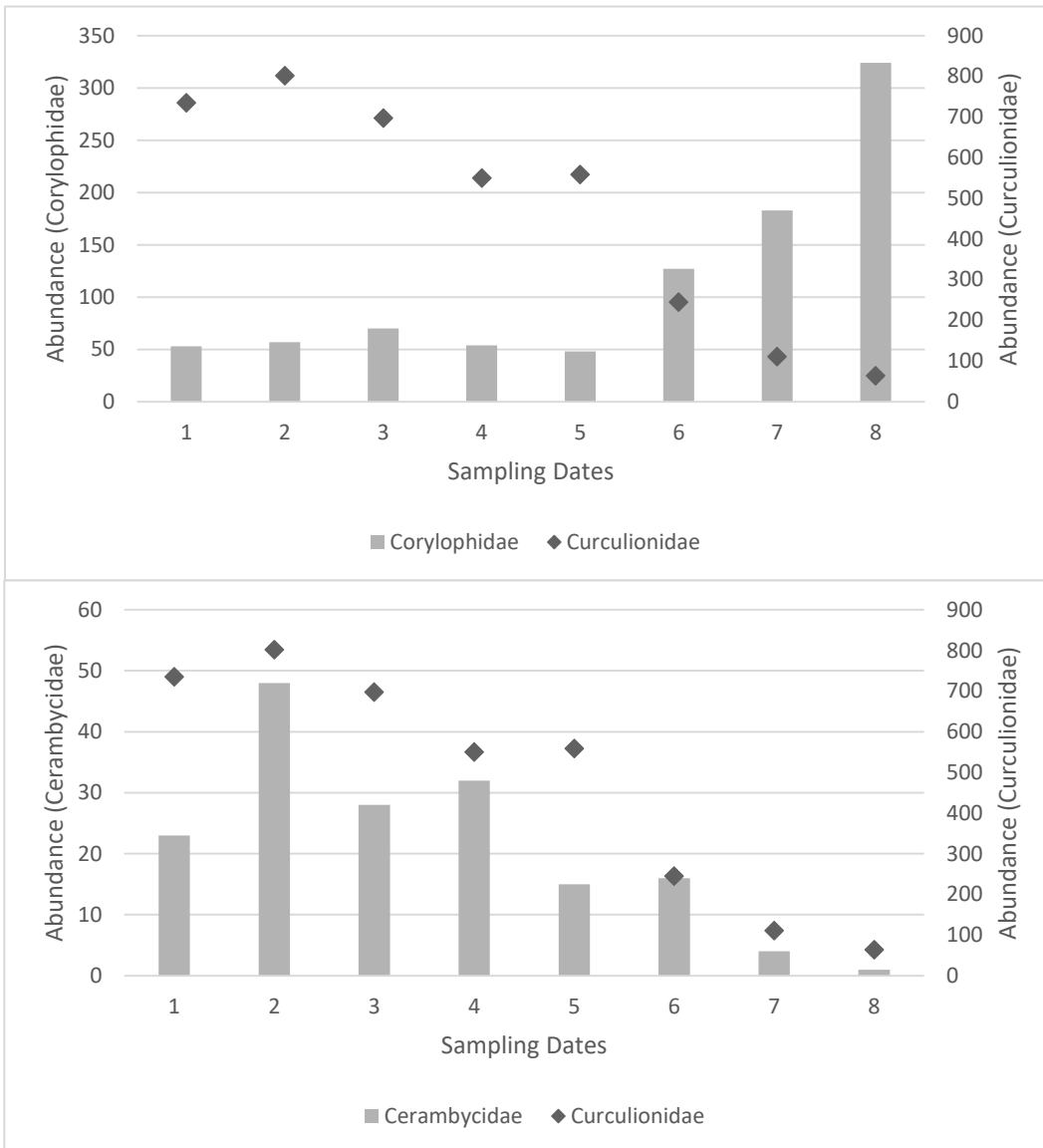


Figure 2.14: Correlation between the abundances of curculionidae and corylophidae, and curculionidae and cerambycidae collected in Kitchener, Ontario from June-September 2015. Curculionidae and corylophidae exhibit a negative relationship, whereas curculionidae and cerambycidae exhibit a positive relationship.

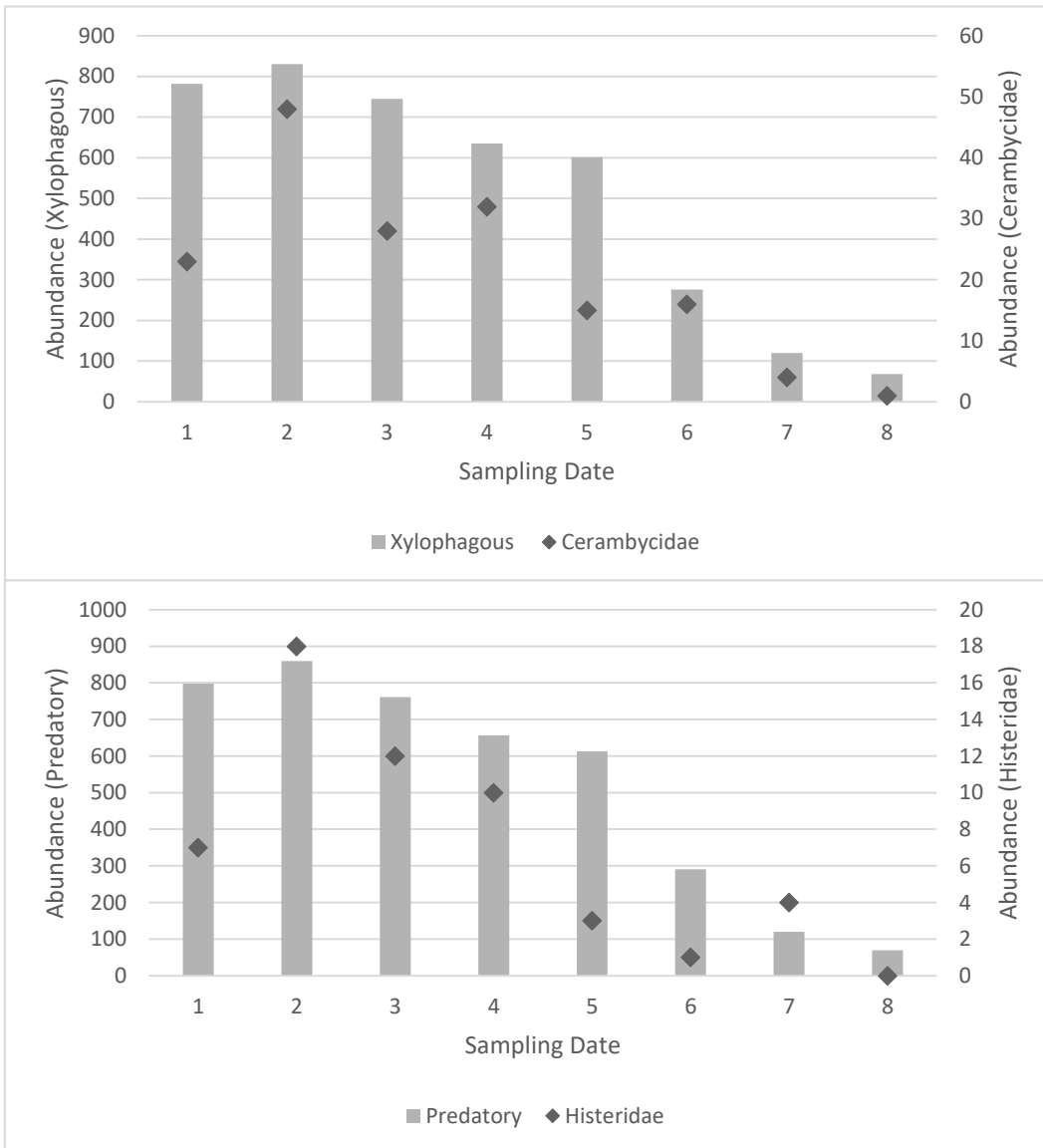


Figure 2.15: Correlation between the abundances of cerambycidae and xylophagous beetles, and histeridae and predatory saproxylic beetles collected in Kitchener, Ontario from June-September 2015. Both cerambycidae and histeridae exhibit positive correlations with their associated groups and may be useful as indicators of available resources within an ecosystem.

Chapter 3.0

Insight into Saproxylic Beetle Communities in Kitchener, Ontario

The findings in this study provide insight into the saproxylic beetle populations of Kitchener, Ontario and suggest all three sites have similar saproxylic beetle assemblages. My three sampling sites were of similar composition in terms of the abundance of rare, moderate, and common families. In each site, only one or two families had very high abundance (i.e. curculionidae and nitidulidae) when compared to all others, with the majority of families being moderately abundant and several rare families (less than or equal to two individuals identified). This pattern is typical of well-established and stable communities, suggesting the saproxylic beetle communities within the urban forest of Kitchener are not heavily impacted by the current anthropogenic pressures affecting them. These include invasive species, pollution, management, deforestation, and fragmentation. A different pattern (high abundance of common and rare families) would have been expected if the community were recovering from a disturbance and/or going through succession (Kempton, 1979). Because all three sites exhibited the same population pattern, there is no evidence for a disturbance response based on abundance curves. Thus, it is likely that current management efforts are not negatively influencing saproxylic beetle populations in Kitchener.

Comparing Biodiversity between Study Sites

Overall saproxylic beetle abundance and diversity was comparable to previous surveys in North America and Europe (Hammond *et al.*, 2001; Wermelinger *et al.*, 2007; Buse *et al.*, 2010; Vinstad *et al.*, 2014; Weiss *et al.*, 2016). A total of 43 saproxylic beetle families containing 10,016 individuals were identified within the city of Kitchener. The percentage abundance of individuals was similar between all sites; Laurentian Wetland exhibited the greatest number of beetles (36.1%), followed closely by Breithaupt Park (34.3%) and Tilt's Bush (29.5%). The presence of the three functional groups (xylophagous, mycetophagous, predatory) in all sites speaks to the availability and diversity of resources available. There is enough deadwood and woody debris in various stages of decay to support a diversity of beetle families in all three of these guilds.

Saproxylic beetle abundance was not found to be significantly different between any of the sites suggesting that current management techniques, differences in park size, and differences in tree species do not impact local saproxylic beetle assemblages. Bouget *et al.* (2013) found that deadwood diversity had a greater impact on saproxylic beetle assemblages than the abundance of deadwood types, and other studies have come to similar conclusions (Økland *et al.*, 1996; Brin *et al.*, 2009). Additionally, the greater the number of tree species present in a stand, the greater the species richness of beetles (Bouget *et al.*, 2013). All three sites contained a variety of tree species, and despite differences in tree diversity, all are able to support similar assemblages. The size differences between parks also seemingly did not affect the abundance of beetles based on my findings.

Saproxylic beetle assemblages were similar in composition based on the Jaccard Similarity Coefficients calculated for each pair. Laurentian Wetland and Tilt's Bush showed the most similarity and Tilt's Bush and Breithaupt Park were the least similar. The vegetation present at each site may help to explain these relationships. Breithaupt Park is comprised only of deciduous forest and meadow ecotypes, while Tilt's Bush and Laurentian Wetland contain these habitats and additionally share meadow marsh and marsh ecosystems. The greater similarity between the latter two sites is reflected in the same pattern for saproxylic beetle assemblages in each, and is not surprising given that beetle diversity depends on deadwood species and available resources for both larvae and adults.

No major differences were observed between Simpson index values; all sites were similarly diverse based on this measurement. The Shannon-Wiener tests revealed that the assemblage in Tilt's Bush was significantly more diverse than both Breithaupt Park and Laurentian Wetland. Tilt's Bush had the greatest Shannon-Wiener value ($H' = 2.25$) suggesting the abundance of families was more evenly distributed than the other two sites, and therefore more diverse. This may be attributed to the greater diversity of tree species and ecosystem types available at this site compared to the other two, and to the presence of coniferous forest that may have increased its ability to support a wider diversity of saproxylic beetles. It has been previously noted that many researchers consider the diversity of tree species and coarse woody debris as the determining factors for saproxylic assemblages.

While conducting this study, the removal of ash trees by the City of Kitchener was identified as a potential driver of change in saproxylic beetle assemblage. This theory, however, is not supported based on my findings. Breithaupt Park and its surrounding areas still contained standing ash and therefore should have had the most diverse and abundant beetle assemblage. It is possible the high abundance of beetles in Laurentian Wetland may be related to the removal of ash trees in the winter of 2015, which resulted in woody debris. Breithaupt Park had the largest number of unique families, which may be attributed to the ash trees still standing. Similarly, the assemblages in Breithaupt Park and Laurentian Wetland were dominated by xylophagous beetles, which could also be related to dead ash. Without further investigation these claims cannot be substantiated. What is certain, however, is the importance of monitoring saproxylic beetles before and after management techniques, such as ash tree removal. It is unknown how assemblages compare to those present before EAB invasion or ash tree removal. The results of such a monitoring effort can communicate the response of communities and dictate further actions by the City.

Families of Interest Due to Ecological Importance and as Indicators of Forest Integrity

1) Curculionidae (Scolytinae)



Figure 3.1: An example of some of the curculionids/scolytines captured in Kitchener, Ontario from June-September 2015. Sampling with Lindgren funnel traps occurred in three urban parks around the city: Breithaupt Park, Laurentian Wetland, and Tilt's Bush. The top left specimen is the introduced pale green weevil (*Polydrusus impressifrons* Gyllenhal) and the other four belong in the subfamily scolytinae.

Scolytinae is an important subfamily of saproxylic beetles within the curculionidae because the tunnels they create in tree trunks act as access routes for many other successional beetle and fungal species, all associated with continued wood decomposition (Zhong and Schowalter, 1989; Hammond *et al.*, 2001). Scolytines feed on a variety of resources, including leaves, shoots, roots, live trees, deadwood, fruit, and seeds (Ohsawa, 2010). The larvae are generally known to feed on distressed or freshly dead trees and are often the first beetles to arrive when moisture content in the wood is still high (Esseen *et al.*, 1997; Ulyshen *et al.*, 2004; Burr and McCullough, 2014). In my study, curculionidae (Fig. 4.2), which mainly consisted of scolytine species, is the most abundant family overall (38.5%). Similarly, Ohsawa (2010) identified this group as the most abundant in their study of Japanese mountain forests, as did Grove (2002b) in his Australian study. Wermelinger *et al.* (2007) discovered that this group was rather evenly distributed between the interior and edges of forests and between the canopy and ground level, most likely due to their broad substrate selection and high rate of migration. In my study transects began at the edge of forests and extended into the interior and scolytines were caught in all traps.

As Wermelinger *et al.* (2007) mentioned in their study, scolytines are more migratory than other saproxylic beetles and may also be more mobile in urban forests due to restricted habitat sizes and resources. It may be that the resources used by Scolytines were in abundant supply at all of the sites sampled here. Because they are known to depend on live or freshly dead trees, this may mean that they had an advantage over other beetles dependent on dead woody debris typically less abundant in

managed forests (Burr and McCullough, 2014). Some researchers have suggested scolytines actually benefit from forest management (Hammond *et al.*, 2001; Zeran *et al.*, 2006). Bussler *et al.* (2011) suggested that in addition to available trees, scolytine abundance can also be attributed to their small size and ability to exploit small twigs which other larger beetles cannot use. This broadens the resources available to them and allows them to occur in large numbers even in habitats of restricted size because they can take advantage of stressed and dead limbs more common on dead trees.

Ohsawa (2010) emphasized the usefulness of curculionidae as a biodiversity indicator within forest ecosystems. Although not found to be significantly correlated with overall saproxylic beetle diversity, here I found this group to be significantly correlated with cerambycidae at all three sites and regionally ($r=0.857$) and cerambycidae was identified as a candidate for use as a bioindicator. Further research into how curculionids interact and respond to other factors in their environment (e.g. other species, habitat disturbances) may reveal their importance as potential bioindicators as well.

2) Nitidulidae

The nitidulids are known commonly as sap beetles since they are most often associated with tree sap; however, most are fungivorous and can be found on a variety of substrates (Zeran *et al.*, 2000). This group was the second most abundant in the present study overall, and most abundant in Tilt's Bush. Because nitidulids are not considered strictly saproxylic, their presence and abundance hints at a potentially wide array of resources. Adult foraging grounds include flowers, fungi, sap, decaying plant matter and carrion (Zeran *et al.*, 2006; Evans, 2014). Some nitidulid species found in Canada are known to be associated with Scolytine galleries, feeding on fermenting sap or fungi within their tunnels, (Hammond *et al.*, 2001), therefore the abundance of both groups is not surprising. Some of these species also tend to be associated with coniferous trees, and this may explain the higher number of nitidulids in Tilt's Bush because of the greater number of conifers and scolytines present.

3) Other Rare Families

Four families had a total of one or two individuals regionally and were considered rare including the anthribidae, lucanidae, scaptiidae, and ptilodactylidae. Lachat *et al.* (2012) found lucanidae to include the highest percentage of deadwood and temperature indicator species out of all saproxylic beetles found in European beech forests. Similarly, Valente-Neto *et al.* (2016) found ptilodactylidae to be less than 1% of total percentage abundance in their study. Grove (2002b) found anthribidae to be one of the most abundant families in his research of Australian saproxylic beetle fauna, however his study took place in the lowland tropical rainforest which may harbour greater abundance and diversity of the fungi anthribidae species depend on. Rainforests may support greater fungi than temperate forests due to their moisture, dense vegetation, higher amount of woody debris, and warmer climate. In a review of Canadian saproxylic beetle surveys, Langor *et al.* (2008) noted low occurrences of anthribidae, lucanidae, and scaptiidae similar to my study. They did not collect any ptilodactylidae whereas I collected one individual.

Besides lucanidae, the other three families noted as regionally rare were not the focus of any previous studies on saproxylic indicators. Little information was found regarding the importance of these families and further research into their significance in Ontario ecosystems would be beneficial. Rare families are often of interest to researchers because they typically display specific ecological requirements and are not very mobile within their habitat (Nilsson and Baranowski, 1997; Grove, 2002b; Lachat *et al.*, 2012). Identifying the rare species and their niches can help managers determine how best to conserve the resources these species depend on and support saproxylic diversity as a whole, which in turn supports forest biodiversity.

4) Cerambycidae



Figure 3.2: An example of some of the cerambycids captured in Kitchener, Ontario from June-September 2015. Sampling with Lindgren funnel traps occurred in three urban parks around the city: Breithaupt Park, Laurentian Wetland, and Tilt's Bush. The top left specimen is *Lepturges confluens* (Haldeman), top middle is *Urgleptes signatus* (LeConte), top right is *Clytus ruricola* (Olivier), bottom left is *Analeptura lineola* (Say), bottom middle is *Graphisurus fasciatus* (De Geer), and bottom right is *Gaurotes cyanipennis* (Say).

Although only significant within two sites, the cerambycids (Fig. 4.3) were the most strongly correlated with saproxylic beetle abundance. These correlations were most prominent in Breithaupt Park ($r=0.926$) and Laurentian Wetland ($r=0.854$). When total cerambycid abundance was compared with total beetle abundance (all sites combined), the relationship was significant ($r=0.835$), although slightly less than that for the corylophidae ($r=0.868$); this latter family displayed a negative correlation to overall abundance and was therefore deemed unsuitable as an indicator (Ohsawa, 2010). Based on these results, the cerambycidae was considered the best family to be used as an indicator group representing total saproxylic beetle abundance. Further research is necessary to determine the robustness of cerambycids as a universal indicator in Ontario. Correlation may be stronger where deciduous trees

dominate; Tilt's Bush had a greater abundance of conifers and an insignificant correlation between cerambycidae and beetle abundance ($r=0.677$). Vance *et al.* (2003) discovered that although deciduous forests in south-central Ontario boasted greater abundances of cerambycids, coniferous stands included greater diversity of species.

The cerambycidae are a diverse and easily identifiable family of saproxylic beetles, and are important components within forest biomes acting as herbivores, detritivores, pollinators, and prey (Maeto *et al.*, 2002; Graham *et al.*, 2012). Their larvae are commonly found in healthy trees, dead or dying trees, logs, and woody debris and bark at varying stages of decay (Ulyshen *et al.*, 2004; Ohsawa, 2010; Burr and McCullough, 2014; Evans, 2014). Those that colonize stressed or freshly dead trees are often some of the first beetles to arrive, and can influence the subsequent colonizing species (Brewer *et al.*, 1989; Esseen *et al.*, 1997; Hammond *et al.*, 2001). Cerambycids have been previously identified as a possible indicator for monitoring overall biodiversity within an ecosystem (Yanega, 1996; Bond and Phillips, 1999; Vance *et al.*, 2003). Ohsawa (2010) found the cerambycids to be the strongest biodiversity indicator in Japanese forests, however, as also seen here, the lures used in his study may have preferentially attracted cerambycids and created a collection bias.

The diversity of adult cerambycid feeding habits, their ease of identification, and their need for dead or decaying wood all contribute to their suitability as indicators. Adults can be found feeding on foliage, flowers, sap, fruit, and twigs, thus their numbers reflect the diversity and abundance of resources available within their habitat (Vance *et al.*, 2003; Evans, 2014). As with other saproxylic beetles, cerambycid diversity is dependent on tree species, flowering plant abundance, leaf area, light intensity, and topography. Certain species are also strongly associated with either the canopy or understory of a forest (Vance *et al.*, 2003). Some are caught only along the edges of forest where flowering plants and sunlight are abundant, while others are restricted to forest interiors where microclimates are more controlled (Vance *et al.*, 2003). Cerambycids may reflect total saproxylic beetle abundance most effectively due to the wide diversity of resources utilized by this group. Their substrate use encompasses those needed by many other beetle families during adult and larval stages which may explain this correlation.

The highest abundance of cerambycidae was found in Breithaupt Park and may be attributed to the adjacent open field where transects began. The pathways throughout this park also created gaps in the canopy increasing sunlight penetration into the forest. The other sampling sites lacked accessibility and nearby natural fields, possibly limiting the availability of flowering plants and sunlight. Saproxylic beetles that colonize living or freshly dead trees have been known to readily disperse to find resources in new habitats (Hammond *et al.*, 2001). Similar to scolytines, highly mobile species are often caught in high numbers along forest edges. The abundance of edge habitat and the arrangement of transects in Breithaupt Park may have enabled the capture of cerambycids. Allison *et al.* (2001) found that many species within the genus *Monochamus* prefer large diameter logs for oviposition, and is likely the case with other cerambycids due to their larger size. Logs were observed at all sites within the sampling areas.

A large number of scolytines and cerambycids may reflect a sudden increase in resources including stressed and recently dead trees; under natural conditions this is typically caused by weather events,

fire, or other natural disturbances (Burr and McCullough, 2014). Jennings et al. (2017) began their sampling midway through an EAB invasion and noted a high number of cerambycids during the first year of sampling, followed by a marked decline several years after. Ghandi and Herms (2010) predicted an increase and possible outbreak of native bark and wood-boring beetles that preferred stressed trees, such as cerambycids and scolytines. Edworthy et al. (2011) discussed the secondary infestation of wood-boring beetles following a pest outbreak, particularly buprestidae and cerambycidae. The populations of beetles that preferentially reproduce on stressed trees are predicted to decline once ash trees are in advanced stages of decay, as demonstrated by Jennings et al. (2017). Beetles dependent on the seeds, foliage, sap, and galls of ash trees are hypothesized to have an overall steady decline in abundance as ash tree decay advances and in some cases may begin to attack alternative hosts (Ghandi and Herms, 2010).

Based on the timeframe of the Kitchener EAB infestation and the state of ash tree death, the sites studied here are most likely at the peak stage of EAB infestation, based on the fact that most trees were visibly distressed and in decline (Burr and McCullough, 2014) at all sites. Scolytines were the most abundant beetles found regionally and at two of the sites (Breithaupt Park and Laurentian Wetland), and were the second most abundant at the third site, Tilt's Bush. However, cerambycids did not display a high percentage abundance at any of the sites, and thus it is uncertain whether the dead ash are acting as resources for this family, as might be expected when responding with high density to a significant surge in resources.

Functional Groups

Families within all three larval habitat guilds were identified at each of the sampling sites in the present study. Typically, fungivores dominate saproxylic assemblages after the first 1-2 years following a disturbance when xylophagous beetles are in high abundance, especially scolytinae (Hammond *et al.*, 2004; Jacobs *et al.*, 2007; Langor *et al.*, 2008). At the regional scale, I found the percentage abundance of fungivores to outnumber the xylophagous group (51.2% and 47.2%, respectively), although this remained true only in Tilt's Bush; xylophagous beetles were the most abundant in both Breithaupt Park and Laurentian Wetland. Tilt's Bush is a larger park with lesser access and therefore fewer visitors, which may result in less intense management by the city. Its relative isolation and greater interior forest may support a larger abundance of dead wood and fungi that could explain the abundance of fungivores. The larger number of coniferous trees may also provide a different variety of fungi able to support more fungivorous beetles. Some conifers (e.g. spruce) tend to have smaller root systems and are prone to falling once dead (Siitonen, 2001). Logs typically have microclimates more suitable for fungal growth compared to standing snags, and this may contribute to the populations found.

In addition to being a possible indicator of total saproxylic beetle abundance, Cerambycidae was also the most strongly correlated family with the xylophagous functional group. The correlation was not only significant at all three sites, but at a regional scale. The larvae of all families within this group depend on wood for food during development, whether it be living, dead, or colonized by fungi. Curculionidae was more strongly correlated with the xylophagous group at a regional scale, but not significantly correlated

within Tilt's Bush at the local scale. This may not, however, eliminate curculionidae as an indicator for xylophagous beetles, as this difference in correlation between the two families may reflect the broader range of substrates utilized by cerambycids. Their host substrate ranges from living to rotting wood and from standing trees to logs (Ulyshen *et al.*, 2004; Ohsawa, 2010; Burr and McCullough, 2014; Evans, 2014), whereas curculionids (scolytines) prefer living or freshly dead trees (Esseen *et al.*, 1997; Ulyshen *et al.*, 2004; Burr and McCullough, 2014). No indicator family for the fungivorous functional group was identified. Several were found to be significantly correlated at the regional scale but no family's relationship was significant within more than one sampling site. This may have resulted from lumping all fungivorous families into one functional group. Some are associated with specific types of fungi which may not correlate with one another. The type of fungi present within a forest depends on a multitude of factors, including tree species, saproxylic beetle vectors, and microclimate conditions (Paine *et al.*, 1997). Since the three sites in the current study were not identical in composition, the types of fungi present may also differ, and this would affect the fungivorous beetles able to survive. The species composition of scolytines may also have differed between sites, and these beetles are essential to the introduction of fungal species into dead wood (Paine *et al.*, 1997). The identification of beetles to family level may have resulted in a misrepresentation of this and other functional groups. Different species have varying substrate preferences and groupings were based on generalizations of families from compiled information in the literature.

Another potential reason for no indicators being identified for the fungivorous group may be the result of sampling. The time frame in which sampling took place may not have been broad enough to encompass the entirety of beetle populations. Corylophidae and laemophloidae both exhibited accumulation curves that were exponentially increasing up to the final collection date suggesting that individuals were still emerging and an entire snapshot of their numbers was not accurately represented. Had sampling continued, their populations may have more accurately reflected that of the fungivorous beetles.

Histeridae was the only family significantly correlated to the predatory functional group at a regional scale. Although not significantly correlated with the populations at Breithaupt Park, this group was the best candidate for an indicator of this guild. The potential of this family as an indicator of saproxylic arthropod diversity and abundance is largely unexplored within the literature and unclear based on these findings. Predators overall were the least abundant and therefore difficult to extract results from especially at the family level.

Recommendations and Conclusions

Building Upon This Study - Many considerations were made in the planning and execution of this study, but due to time and resource constraints certain aspects could not be incorporated. These include: deadwood surveys; woodpecker surveys; identifying beetles to species, and; ash tree assessment.

Quantifying and identifying deadwood volume and species can help researchers understand the association between the beetles found and the resources they require. Assessing trees for other known

saproxylic beetle habitat, such as cracks and hollows, will also contribute to our understanding of their needs and aid in planning management. Some researchers suggest using coarse woody debris as an indicator for saproxylic diversity that may be an avenue of interest to the city of Kitchener. This was not included in my study due to the previous limitations mentioned, and because my main focus was to identify a baseline of insects that can be compared to future studies and aid in beetle monitoring.

Monitoring woodpeckers and other forest birds can help indicate whether the availability of saproxylic insects is sufficiently supporting their populations. Because these animals have longer generation times and lifespans, a long-term study of their numbers would provide the most meaningful results. My study took place in one year, so I could not include this aspect. Resources such as eBird may prove useful for preliminary studies into their populations in Kitchener, and monitoring can be put into place using citizen science similar to the Audobon Christmas bird count or the MNRF Plover Guardian program.

Identifying specimens to species was not possible in my study due to limited time and my taxonomic knowledge, but would provide more detailed results. Further processing of the beetles collected here and in future surveys could uncover indicator species that can aid with biodiversity monitoring and relationships between beetles that remain unseen at family level. Management should focus a significant effort into discovering these species and providing sufficient resources to saproxylic beetles and other saproxylic organisms.

Finally, ash tree quantification and emergence traps on these trees would identify the decay state of ash present in each park and improve our knowledge of what families/species actually use these distressed trees. Some species are monophagous on ash (Fig. 3.3) and many others use the wood and foliage, so understanding which are utilizing these resources can help managers plan their survival in the years to come as ash disappear. Further study into these species may reveal they are turning to alternative resources as the ash disappear, similar to EAB reproducing on white fringetree (Cipollini, 2015).



Figure 3.3: The sole eastern ash bark beetle (*Hylesinus aculeatus* Say) and one of the few redheaded ash borers (*Neoclytus acuminatus*) collected in Kitchener, Ontario from June-September 2015. Further research into their populations and others dependent on ash may reveal the impacts of EAB infestation and eradication techniques used by the City.

Additional Considerations - Bouget *et al.* (2013) identified two key features influencing saproxylic beetle richness in deciduous forests: the diversity of deadwood and stand openness. Increases in common and rare species were positively correlated with these forest attributes. Sunlight influences the decay rate of wood, the species of colonizing fungi, and the microclimate conditions of important substrates (Vodka *et al.*, 2009; Bouget *et al.*, 2013). Many saproxylic beetles are thermophilic and prefer areas with plenty of sunlight (Bouget *et al.*, 2013). Additionally, microhabitat diversity and availability has been confirmed as being associated with saproxylic beetle assemblages, but this relationship requires further research to fully understand (Winter and Möller, 2008; Bouget *et al.*, 2013). Trees containing a greater amount of hollows, sap flows, crown death, and fungal growth are thought to be especially important in this relationship (Jonsell and Nordlander, 2002; Ranius, 2002; Yoshiomoto *et al.*, 2005; Bouget *et al.*, 2011). Incorporating diversity into management techniques is therefore essential to saproxylic diversity and overall ecosystem productivity.

The resource needs of adult beetles is an important factor to consider when managing for saproxylic biodiversity. Adults in families such as cerambycidae and buprestidae depend on foliage and flowers (Wermelinger *et al.*, 2007), and these resources must compliment those needed for larval development. All forests sampled in the present study were surrounded by neighbourhoods containing both trees and gardens where beetles would be able to feed. Breithaupt Park encompassed a natural field area offering increased flowering plant abundance, while Laurentian Wetland was surrounded by residential areas and had an abundance of Jack-in-the-Pulpits present in the understory. Tilt's Bush had the lowest amount of flowering plants in the understory due to the abundance of conifers, but was adjacent to a soybean field. The planting of native flowering plants by the community should be encouraged by the city of Kitchener, and the use of flowering plants by beetles dependent upon them should be evaluated.

A long-term study of the sampled sites and others in the surrounding area is recommended in order to more accurately understand saproxylic insect communities. The ebb and flow of saproxylic beetles depends on a multitude of factors including the type of deadwood present, and differences may be seen within other urban parks in Kitchener. My study took place over a single reproductive season and mainly provides a basic beetle inventory that can act as a baseline for comparison with future studies. No baseline beetle inventory is available for this area so further monitoring is the only way to track changes in populations that the current study has uncovered. The populations of certain families can vary greatly over time depending on ecosystem characteristics, so monitoring beetles and their resources is an important aspect of forest management. Additional studies may help explain the abundance and diversity found, for example the high abundance of scolytines, and determine how they fluctuate in response to ecosystem changes.

Diversifying trapping methods may increase the number of families caught and provide a more complete view of assemblages. Malaise traps, pitfall traps, and studying emergence holes have all been employed in previous studies and evaluated for their effectiveness (Jennings *et al.*, 2017). Including multiple methods of collection could result in grander implications than those suggested by this study that was meant as a jumping off point for future research. Certain species are restricted to particular microhabitats within a forest, and are not as likely to be caught in traditional flight-intercept or Lindgren-funnel traps (Ranius and Jansson, 2002; Bouget *et al.*, 2013). These species are often rare due

to their restricted habitats and are especially important to conservation and management efforts. Evaluating rare species as potential indicators was beyond the scope of this study, but future research into the regionally rare families and those found to be rare in each sampling site may prove fruitful.

Adaptive management - In the face of climate change and increasing anthropogenic pressures impacting forest ecosystems, adaptive management strategies are necessary. It is difficult to predict how the intricate web of forest organisms will change as the climate warms and more invasive species take root. In the case of EAB, infested forests have already been shown to go through varying stages of population booms and busts that correlate with EAB density. However, this can differ depending on geographical location, tree species, and numerous other factors. Enforcing a singular blanket method of forest management will not be in the best interest of these ecosystems as they will not be suited for each individual case. The connectivity between urban forests and the mobility of resident species needs to be considered.

The city of Kitchener is experiencing urban growth and development every year, and this is threatening forests and increasing fragmentation. However, an urban forests management plan has recently been put into action for the period 2015-2018 in an attempt to foster the conservation and growth of urban trees and natural areas (City of Kitchener, 2017b). The plan involves creating a proactive forest management program that allows for adaptive methods of urban forestry and viewing trees as assets to the community. The city should embrace past and present research focused on varying forestry techniques applicable to this region. In order to adapt this management to encourage saproxylic organism diversity, the literature pertaining to this subject should be reviewed.

Previous research has outlined the requirements of maintaining a high diversity and abundance of saproxylic arthropods and some offer suggestions for management strategies (Attiwill, 1994; Martikainen, 2001; Siitonen, 2001; Grove, 2002a). By encompassing such strategies into their proactive forest management plan, the city of Kitchener will be increasing the ecological productivity of their urban forests. In the past, saproxylic beetles have not been the main focus of forest management, especially in urban settings. By managing for maximum biodiversity and resilience against present and future anthropogenic pressures the city will be ensuring a healthier environment for its residents, be they human, plant, or animal.

Kitchener's proactive urban forestry plan is largely focused on community involvement (City of Kitchener, 2017b). Communication is an important aspect of adaptive strategies and especially necessary when attempting to manage urban forests that are surrounded and frequented by residents. Increasing the understanding of the importance of maintaining urban forests amongst the public can lead to greater appreciation for these areas, which may reduce pollution, invasive species, and harvesting. By including the community in forest management strategies and educating them about the importance of these assets, urban forest management should be a more manageable task.

Collaborative Insect Monitoring and its Importance to Urban Forest Management - In many regions around the world, more conspicuous animals and plants receive the brunt of conservation efforts. This is reflected in funding allocations; little is directed at entomological research by cities and conservation authorities unless there is a destructive pest having a significant ecological or economic impact (Medeiros *et al.*, 2013). In Hawaii, the importance of protecting endemic insect species has become recognized by researchers, as has the repercussions their protection will have on other species (i.e. birds) whose populations are threatened (Groomsbridge, 1992; Hafernik Jr., 1992; Kim, 1993; Medeiros *et al.*, 2013). The studies focused on these larger species should incorporate insect research into their initiatives, as many of these species depend on insects in some way for survival. If relationships are uncovered we can start to develop entomological studies that tie into the studies of birds, mammals, and so on.

North America needs to invest more interest into insect conservation. In the United States, only one arthropod per 16 vertebrate species and plants is listed as an endangered species, despite the hypothesis that many more are either endangered or threatened (Medeiros *et al.*, 2013). In Ontario, 22 insect species are listed as 'At Risk' within the province. Rather than focusing on the survival of any particular insect species, however, the bigger initiative should be to understand why these insects are at risk. If we understand the ecological changes interrupting their populations, we can work towards ecosystems that can sustain their populations and, in turn, sustain all other organisms within the habitat (Hammond *et al.*, 2001).

Developing a working understanding of insect populations can be a time consuming and expensive task. As mentioned previously, it is a lengthy process that requires at least one person with intimate knowledge of insect taxonomy (Medeiros *et al.*, 2013). These factors may be deterring managers from adopting these processes as a part of their conservation initiatives despite the importance of insects to ecosystem function. Insects follow population trends (e.g. boom-and-bust) that cannot always be predictable, including seasonal variation and yearly changes (Ulyshen *et al.*, 2004). These types of changes mean multiple sampling events throughout the year and multi-year long studies, which managers may not be prepared to commit to (Medeiros *et al.*, 2013). As demonstrated by the current study, a single person is capable of collecting a large number of insects in a single season with minimal equipment and funding, and any municipality or conservation authority should be able to incorporate this work into their management plans. There are many resources for insect identification including available taxonomic keys and universities with knowledgeable students and staff.

Additionally, the response to various environmental stressors can differ between different groups and even species of insects making it complicated to interpret changes and identify indicators. The current solution to this is mass sampling and consistent monitoring until enough information is gathered to allow for interpretation (Medeiros *et al.*, 2013). Where broad sampling is not possible, insects should be sampled in areas of interest and should take place both before and after any management efforts within the ecosystem in question and include a control site where no management is implemented (Medeiros *et al.*, 2013). For example, the removal of ash trees within the City of Kitchener should include insect monitoring to capture potential changes it is causing. Ideally, each area where trees are removed would

be sampled before and after tree removal. The city of Kitchener should adopt this practice moving forward as more areas are cleared of ash.

Medeiros *et al.* (2013) suggest that an effective method of collaboration and organization between insect conservation initiatives would be an online database comprising all information. While completing this study, I found a lack of such a resource for Ontario despite similar online databases for other animal groups, such as eBird. Arthropods seem to have been neglected when it comes to collaborative information sharing and the ability for others to find baseline databases for this group are difficult. This limits researchers' and managers' abilities to compare current populations to previous numbers making it impossible to infer trends without completing a long-term study. There needs to be a greater awareness regarding this missing information and a collaborative effort to create a functional sharing application that can benefit managers and researchers alike. For example, publishing the University of Guelph insect collection database online so anyone can access, search, and add to it would be a fantastic resource and could facilitate public awareness of native insects. Creating an application for phones such as the Ontario Reptile and Amphibian Atlas App would make it easy for citizens to upload and share their insect photos and increase public appreciation for insects. Social media and online sharing are powerful tools that bring together many entomological experts and enthusiasts that are happy to identify species for others. Tapping into this enthusiasm would be sure-fire way to create baseline inventories for the province.

Consistent insect sampling will not only reveal species' reactions to disturbance events, but can also help detect pest insects before there is a noticeable infestation. In Ontario, purple and green prism traps are now frequently used to monitor for EAB outbreaks. Some Italian ports use insect traps to detect possible invasive insect species arriving on foreign ships and in foreign cargo (Rassati *et al.*, 2014). Incorporating insect monitoring into management can have the added benefit of alerting managers and researchers to potential pest species. In a time when global warming is increasing the possibility of insect pest outbreaks (Laštůvka, 2009), monitoring is an important aspect to consider for managers and may work to prevent the ecological destruction these pests can cause.

A Feasible Plan for the City of Kitchener - Outlining what the City of Kitchener should be doing with regards to saproxylic beetles is easily said but poses difficulty in practice. With limited funding and a wide range of projects and initiatives the city invests in, there is likely little left for insect monitoring. I suggest that they take advantage of the resources available to them to reduce costs as much as possible. With the Universities of Waterloo and Laurier so nearby, it is possible for the city to collaborate with professors to establish an insect monitoring regime. Undergraduate, graduate, and PhD students may be willing to incorporate such research into their projects and share their results with the city and help them plan for future management. This would mitigate the cost of long-term monitoring with the benefit of learning more about the parks within Kitchener's borders.

If some funding is available for monitoring, the city may want to look into monitoring the edges of forests within urban parks. These areas generally have high capture rates and should garner a diversity of cerambycids, the group identified as the best indicator family based on my results. Restricting

monitoring to forest edges can reduce the number of traps necessary when compared to sampling throughout an entire wooded area, yet still produce meaningful results.

Citizen science is another important resource that can be utilized by the city. As previously mentioned, other such initiatives have resulted in valuable information regarding a variety of species (e.g. eBird, Plover Guardians, etc.). The City of Kitchener could recruit citizens to aid with insect, bird, and deadwood and tree monitoring. By cooperating with the local universities, the city could even develop an app for citizens to report any sightings of organisms of interest which is an easy way to gather data. The only requirement would be someone to confirm sightings through pictures taken on the app and sent to the city, which could also be outsourced to a university. In their urban forestry management plan, the City of Kitchener identified the importance of citizen participation in the survival of the urban forest, and they need to take full advantage of this resource. It not only results in important data, but provides a learning experience for those involved which creates a more informed and concerned community.

Conclusions - The continuation of saproxylic insect monitoring is an important aspect of urban forest management. Horák (2011) investigated the biodiversity of saproxylic beetles in secondary urban forests of the Czech Republic and concluded that these areas, despite being considered less ecologically important by many, house a wide diversity and abundance of saproxylic organisms. When compared to old-growth forests in the same geographical region, Horák (2011) discovered that the urban forests contained a greater number of red-listed indicator species. With saproxylic beetles being as critical as they are to a forest ecosystem, and the demonstrated importance of urban forests as habitat, greater focus needs to be given to this relationship.

My study has provided a snapshot into the saproxylic beetle populations within the City of Kitchener, but further research is necessary to evaluate population trends. With the city's recent adoption of a proactive and adaptive forest management strategy, Kitchener seems to be well equipped to handle the current and future ecological changes resulting from anthropogenic pressures. However, this new strategy must incorporate the study of saproxylic organisms if the productivity and benefits of forests is to be secured. Hopefully, this research has created a sufficient baseline inventory of saproxylic beetles that can be used for comparison with any future studies exploring this issue. Taking advantage of all available resources to reduce necessary funding for research should prove beneficial to the health of the urban forest across Kitchener. Through their support of multiple processes within a forest, saproxylic insects in turn support forest diversity and changes in their populations can predict change in other species. By taking their populations into consideration, the City of Kitchener will be closer to achieving their goal of a healthy urban forest.

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Appendix

Table A1: Raw count data for Breithaupt Park (BP), Laurentian Wetland (LW), and Tilt's Bush (TB) for each sampling date.

	BP June 14	BP June 28	BP July 12	BP July 25	BP Aug 08	BP Aug 23	BP Sept 04	BP Sept 19
Anthribidae				2				
Bostrichidae	3							
Buprestidae	1	1						
Carabidae								
Cerambycidae	4	23	17	11	4	4	1	1
Cerylonidae		2	2					1
Ciidae	1	3	6					
Clambidae								
Cleridae			2	2				1
Corylophidae	24	17	38	35	26	59	100	177
Cryptophagidae		2		2				
Cucujidae			3					
Curculionidae	141	301	318	228	180	122	71	54
Elateridae	11	10	3	11	4	1		
Endomychidae								
Erotylidae	2	4		4	5	1	3	4
Eucnemidae	8	47	19	20	4			
Histeridae	1	1	2	4	2	1		
Laemophloeidae	13	15	17	20	16	43	29	36
Lampyridae								
Latridiidae	46	87	52	105	55	86	15	46
Leiodidae					4			1
Lucanidae		1						
Melandryidae		8	2	1				
Mordellidae	1		6	18	6	7	2	2
Mycetophagidae			1				1	1
Nitidulidae	24	6	10	12	16	33	12	19
Oedemeridae								
Passandridae			1	1	1			
Ptiliidae								
Ptinidae	5	6	6	9		2	2	
Ptilodactylidae								
Ripiphoridae			2	1				
Salpingidae			1				1	
Scarabaeidae	3	3	6	3	3			
Scraptiidae			1					
Staphylinidae	18	9	10	17	37	10	3	3

Stenotrachelidae	3	2						
Tenebrionidae	17	16	5	1	6	3		
Tetratomidae	2	1		2				
Throscidae	8	16	8	5	3	8	3	
Trogossitidae	1			2		2		
Zopheridae		24	48	20	15	3	1	2

	LW June 14	LW June 28	LW July 12	LW July 25	LW Aug 08	LW Aug 23	LW Sept 04	LW Sept 19
Anthribidae								
Bostrichidae	1							
Buprestidae		1	1					
Carabidae		1				1		
Cerambycidae	8	15	6	8	2	4		
Cerylonidae	5	7	10		1		1	
Ciidae	5	4	10	4	4			
Clambidae	1							
Cleridae	5	1	1					
Corylophidae	18	21	17	8	4	31	56	100
Cryptophagidae			1	2				
Cucujidae	1							
Curculionidae	404	357	272	219	302	70	26	5
Elateridae	34	33	10	8		1		
Endomychidae		2	1			2	1	1
Erotylidae	1	2			2	1	1	
Eucnemidae	9	120	25	59	9			
Histeridae	3	10	3	2	1		1	
Laemophloeidae	7	6	4	11	9	35	16	39
Lampyridae	1	7						
Latridiidae	99	166	78	38	8	1	4	5
Leiodidae		1		1				4
Lucanidae		1						
Melandryidae								
Mordellidae			10	13	5	8	2	2
Mycetophagidae		2						
Nitidulidae	42	12	13	44	126	58	47	53
Oedemeridae	1							
Passandridae								
Ptiliidae			1					
Ptinidae	9	2	2	2	4			
Ptilodactylidae								
Ripiphoridae		1	6	8				

Salpingidae	1							
Scarabaeidae	1					1		
Scraptiidae								
Staphylinidae	29	15	19	14	6	7	4	5
Stenotrachelidae								
Tenebrionidae	9	4	2		2			
Tetratomidae	3		2		1			
Throscidae	6	16	11	1		8	3	
Trogossitidae	0	3						
Zopheridae	2	13	39	12	16	3	1	

	TB June 14	TB June 28	TB July 12	TB July 25	TB Aug 08	TB Aug 23	TB Sept 04	TB Sept 19
Anthribidae								
Bostrichidae			1					
Buprestidae	1	1						
Carabidae		6	1					4
Cerambycidae	11	10	5	13	9	8	3	
Cerylonidae	1	2						
Ciidae	2		1		1			
Clambidae	1			1				
Cleridae	1	1	2					
Corylophidae	11	19	15	11	18	37	27	47
Cryptophagidae	1			1				
Cucujidae								
Curculionidae	190	144	108	103	77	53	14	5
Elateridae	11	10	13	8	6	1		
Endomychidae	1	2						
Erotylidae		1					1	
Eucnemidae	4	85	12	16	13	2		
Histeridae	3	7	7	4			3	
Laemophloeidae	3	3	4	3	5	21	17	21
Lampyridae		3						1
Latridiidae	33	76	42	25	5	5		4
Leiodidae					47	22	18	1
Lucanidae								
Melandryidae						1		
Mordellidae		3	5	23	13	12	2	
Mycetophagidae		6						1
Nitidulidae	31	22	10	204	527	100	25	14
Oedemeridae	2			3	2			
Passandridae								

Ptiliidae	2	3	1	1				3
Ptinidae	8	3	4	12	9	1		
Ptilodactylidae			1					
Rhipiphoridae		1	6	4				
Salpingidae	2	2						
Scarabaeidae	5	2	3	2				
Scraptiidae								
Staphylinidae	29	56	26	11	57	42	16	9
Stenotrachelidae								
Tenebrionidae	9	19	15	9	3			
Tetratomidae					1	6		
Throscidae	1			2		1	1	
Trogossitidae	2	1		1		1	0	1
Zopheridae			7	6	3			

Table A2: Ecosystem composition of each site expressed in area (ha) and percent abundance.

Site	Ecosystem Type	ha	% abundance
Breithaupt Park	dry – fresh sugar maple – beech deciduous forest	9.01	46.49
	fresh – moist black walnut lowland deciduous forest type	4.9	25.28
	dry – fresh sugar maple – white ash deciduous forest type	1.92	9.91
	dry – fresh sugar maple – black cherry deciduous forest type	1.78	9.18
	goldenrod forb meadow type	1.2	6.19
	dry – fresh sugar maple deciduous forest ecosite	0.57	2.94
	Total	19.38	
Laurentian Wetland	reed – canary grass graminoid mineral meadow marsh type	8.54	44.34
	stonewort submerged shallow aquatic type	3.35	17.39
	deciduous thicket	3.06	15.89
	cattail mineral shallow marsh type	1.37	7.11
	goldenrod forb meadow type	0.89	4.62
	silver maple mineral deciduous swamp type	0.6	3.12
	fresh – moist poplar deciduous forest type	0.43	2.23
	open water (storm pond)	0.29	1.51
	dry – fresh sugar maple – beech deciduous forest type	0.22	1.14
	common reed graminoid mineral meadow marsh type	0.21	1.09
	green ash mineral deciduous swamp type	0.2	1.04
	fresh – moist sugar maple – lowland ash deciduous forest	0.06	0.31
	Bebb's willow mineral deciduous swamp type	0.04	0.21

		Total	19.26
Tilt's Bush	white cedar – conifer organic coniferous swamp type	11.77	31.81
	fresh – moist willow lowland deciduous forest type	6.79	18.35
	forb mineral meadow marsh ecosite	5.35	14.46
	fresh – moist hemlock coniferous forest ecosite	3.88	10.49
	dry – fresh sugar maple – hemlock mixed forest type	1.96	5.30
	fresh – moist green ash – hardwood lowland deciduous forest type	1.82	4.92
	dry – fresh sugar maple – white ash deciduous forest type	1.39	3.76
	dry – fresh mixed meadow ecosite	1.3	3.51
	reed – canary grass graminoid mineral meadow marsh type	0.77	2.08
	reed – canary grass graminoid organic meadow marsh type	0.62	1.68
	meadow	0.5	1.35
	fencerow	0.46	1.24
	fresh – moist poplar deciduous forest type	0.14	0.38
	cattail mineral shallow marsh type	0.11	0.30
	dry – fresh cedar coniferous forest ecosite	0.08	0.22
	joe pie weed mineral meadow marsh type	0.06	0.16
	Total	37	

Table A3: The number of ash to be injected and removed in comparison to total ash and total trees in each ward. Breithaupt Park is located in Ward 10 and removal/injection is scheduled for 2017. Laurentian Wetland is located in Ward 4 and removal/injection was scheduled for 2015. Tilt's Bush is located in Ward 10 and was scheduled for removal/injection in 2013. All ash tree removals took place in the winter.

	Total Ash	Injected	Removed	Total Trees
Ward 10 (BP, 2017)	343	71	272	6556
Ward 4 (LW, 2015)	719	126	593	5714
Ward 5 (TB, 2013)	196	0	196	3441