The Implications of the Emerald Ash Borer (*Agrilus planipennis*) on Riparian Canopy Cover in Three Halifax Regional Municipality Parks

Environmental Science Honours Thesis

Jordan Haughn

Supervisor: Peter Duinker

April 2020

Table of Contents

Acknowledgements

I would like to thank Dr. Peter Duinker for his guidance and support over the course of this study. I would also like to extend my gratitude to Dr. Susan Gass for her help and structured format and my colleague Brandon Patterson for field work assistance. Finally, I would like to thank Dr. Chris Greene for his advice and resources in GIS.

Abstract

Invasive species are posing an increasingly large threat to Canada's urban forest. The Emerald Ash Borer (EAB) is an invasive pest with the potential to eliminate entire ash stands. The EAB was discovered in Nova Scotia in 2018. In this study I aimed to address the research question "what are the implications of the EAB on short-term riparian zone canopy cover in three Halifax Regional Municipality (HRM) parks?". Both census and cruise-transect-sampling methods were exercised to determine ash proportions and distribution in each park. For the census method, all trees within the park were measured (diameter at breast height (DBH), crown condition, crown position) and digitally geo-positioned to determine distribution. Using an equation derived from existing ash data, ash crown projections were then calculated for each ash tree based on DBH. The cruise method was conducted using plots dispersed along a transect line stretching the length of the stream. It was found in the park undergoing census method (Fish Hatchery Park) that ash accounted for approximately 30% of total canopy cover within the assessed park. Analysis of plot data demonstrated ash presence in 41% of all plots sampled, only five of which were projected to fall below Nova Scotia's riparian-zone regulatory basal area as a result of ash removal. In Moirs Mill Park, ash accounted for 20% of all trees measured, and were present in eight of the 11 plots. In Sir Sandford Fleming Park, ash accounted for 5.5% of all trees measured and were present in eight of the 20 plots. Overall, the arrival of EAB will have a modest impact on riparian canopy cover in the HRM parks sampled. The visual distribution and location-specific data along the riparian zone will help forest managers and planners understand areas of highest risk.

Chapter 1: Introduction

The urban forest delivers myriad environmental, economic, and social benefits to people in cities around the world. Trees provide many ecosystem services to cities such as carbon capture, mitigation of urban heat island effect, increased property value, pollution absorption, and aesthetic value; they are thus powerful agents of sustainability in many urban areas (Duinker et al., 2015).

A predominant threat to the urban forest is invasive insect species, which are capable of decimating entire tree populations (Ward, 2004). Climate change can exacerbate the rate of non-native pest invasions, as warmer temperatures have been shown to facilitate invasive pest movement (Huang et al., 2011). Invasive pests are responsible for killing tens of millions of trees over the past several decades and threaten billions more (MacPherson et al., 2017). Once a pest is established, few treatments are available to keep trees alive (MacPherson et al., 2017). These threats place the urban forest at risk and in need of enhanced management.

The impact of invasive pest species can be particularly severe in ecologically sensitive areas such as riparian zones (Robertson et al., 2018). Riparian areas support a high diversity of flora and fauna due to their rich, moist soils and proximity to diverse biomes (Watersheds Canada, n.d.). These areas also support hydrologic processes and aid in the regulation of water temperature and control of erosion (Nisbet et al., 2015). These unique values hold true in both urban and rural environments (Nisbet, 2014)

The emerald ash borer (EAB) (*Agrilus planipennis*) is a species of wood-boring beetle native to East Asia (Natural Resources Canada, 2019). The EAB feeds on all trees in the ash genus (*Fraxinus* spp.), boring through the bark as larvae, feeding on phloem and cambium in S-shaped patterns along the bole (Herms & McCullough, 2014). These feedings impede the tree's ability to transport nutrients and water and eventually girdle the tree, causing mortality after two to four years of canopy decline (Herms & McCullough, 2014). As much as 99% of infected trees in an area can be eradicated after 8-10 years of EAB establishment (Natural Resources Canada, 2019).

The EAB has become invasive in North America since it was first identified in Detroit, Michigan, and Windsor, Ontario, in 2002 (Robertson et al., 2018). The EAB is believed to have arrived and spread through the transport of firewood and other ash wood materials such as nursery stock, wood packing, and building materials from East Asia (Roland & McCoullough, 2006). Biological factors have also played a part in the spread, as there are few natural predators of the EAB in North America, and the existing natural predators (for example, woodpeckers and parasites) have not been able to slow the spread (Liang & Fei, 2013). Climate change will also accelerate the spread of the EAB. Climate suitability models have predicted a northern expansion of its habitat range in North America, reaching 80% of ash distribution range in as few as seven years (Liang & Fei, 2013).

Currently, the EAB has become established in Manitoba, Quebec, Ontario, New Brunswick, and Nova Scotia (Ryall, 2013; CFIA, 2018). The presence of the EAB in Nova Scotia was confirmed by the Canadian Food Inspection Agency (CFIA) in September 2018, citing the town of Bedford in the Halifax Regional Municipality (HRM) as the location of discovery (CFIA, 2018). This places HRM's urban forest and ash canopy at risk.

Ash is an ecologically and economically significant tree genus in North America. Ash trees are widely distributed and found in both urban and rural areas (Burns & Honkala, 1990). Ash is a non-conifer genus with nine species present in North America found in greater than 25 forest cover types (Morin, 2010; Cappaert et al., 2005). It supports a community of some 282

Gandhi & Herms, 2010). As ash declines, the populations of at least 50% of these species are projected to decline (Gandhi & Herms, 2010). The decline of ash can also put various other tree species at risk as non-specialist arthropods shift to alternate hosts (Gandhi & Herms, 2010). Within Canadian municipalities, the loss of ash trees due to the EAB is projected to cost about CAD $890 million total (McKenney, 2012). Black ash (*Fraxinus nigra*) specifically has cultural significance, as it is valued by many indigenous groups and has been traditionally utilized for basket-making (Diamond & Emery, 2011). Clearly, ash trees are valuable enough to warrant special attention with the advent of EAB.

Since identification in 2002, the EAB has killed millions of ash trees and spread rapidly across parts of Canada and the United States (Herms & McCullough 2014). Considerable research on the effects of the EAB on both urban and rural forest ecosystems in the Northeastern United States has been completed, but relatively little is known about the implications on Canada's urban forests and even less on a provincial scale for Nova Scotia (Nisbet et al., 2014; Poland & McCullough, 2006). There have been various studies quantifying the financial implications of the EAB in urban areas (Haur & Peterson, 2017) but few on the implications on urban canopy cover. The large threat posed by EAB, along with the poor understanding of impacts in Canadian cities, creates a need for targeted research.

Additionally, even fewer studies focus on the implications for riparian ecosystems. One study on the effects of the EAB on riparian regions in Southern Ontario was done as a master's thesis and not published (Nisbet, 2014). Similar research has not been done in Nova Scotia. Considering that the EAB was identified in Nova Scotia so recently, there is a clear gap in knowledge concerning possible implications of its presence there.

The goal of this study is to identify the current density and spatial distribution of ash trees in selected riparian zones on HRM park properties and identify the potential short-term implications of the EAB on canopy cover within the parks. This will provide insight on the magnitude of impact of the EAB on these ecologically sensitive urban areas and allow city planners to prioritize parks at greatest risk for management treatments. The research question is as follows:

What are the implications on vulnerability of the EAB on short-term canopy cover in riparian zones in three HRM parks?

This will be answered by addressing the following objectives:

1. Identify the spatial distribution of ash trees in selected riparian zones in HRM parks.
2. Complete an analysis of proportional loss of canopy in individual parks assuming simultaneous mortality of all ash trees.

The research question was addressed using both a tree census and cruise method to collect tree inventory data during fall 2019. The spatial distribution of the ash trees was determined using GIS spatial analysis. Proportional loss of canopy was identified among cruise points and was qualitatively examined for spatial patterns. Finally, the data were contextualized in relevant literature to determine if riparian function will be impacted.

Chapter 2: Literature Review

The following literature review will provide an overview of the current research findings concerning the EAB, ash tree ecology, and riparian zone significance in the urban forest. The current state of knowledge concerning EAB ecology, dispersal, and various spread projections will be explored. Ash tree ecology will also be explored as well as treatment and vulnerability of ash trees to EAB invasion. Finally, riparian zone significance in the urban forest will be addressed. This literature review will summarize the current state of knowledge and identify important knowledge gaps.

2.1 Emerald Ash Borer Ecology

The EAB is a species of beetle native to East Asia that has quickly become invasive in North America due to favourable climatic conditions and relative vulnerability of North American ash species compared to other ash species, causing severe damage to ash populations (Tanis & McCullough, 2012). The native range of the EAB has been documented to reach from eastern Russia through China, South Korea, and Japan (Orlova-Bienkowskaja & Volkovitsh, 2018). The first identification of EAB in North America occurred in southeast Michigan and Windsor Ontario in 2002 (Robertson et al., 2018). The EAB is believed to have reached North America somewhere between 1998 and 2000 (Herms & McCullough, 2014 Prasad et al., 2010). Ash tree mortality rates have peaked as high as 99.7% in Michigan (Klooster et al., 2018). Hundreds of millions of trees have been killed since first documented arrival in North America, preying on both healthy and dying trees (Herms & McCullough, 2014).

The EAB is an aggressive insect which feeds in high densities on their host trees until they are girdled and die (Wang et al., 2010). Adult EABs lay eggs in the bark of their host trees, leaving D-shaped entrance holes in the upper canopy. Upon hatching, larvae chew through the bark and feed on the phloem and cambium tissues. They continue to feed in galleries, creating S-shaped patterns until the tree is eventually girdled and dies due to inability to transport nutrients (Wang et al., 2010). Although EAB will attack both healthy and declining trees, they are more attracted to trees under stress. Stressed trees were found to have significantly higher densities of EAB (Crook & Mastro, 2010). Attraction to stressed trees can also facilitate a spread to nearby healthy trees (Crook & Mastro, 2010). Overall, the EAB is an invasive pest with the potential to cause severe damage to North American ash trees.

2.1.1 Emerald Ash Borer Spread

EAB has spread throughout North America, largely through anthropogenic means, and is continuing to expand its habitat range. The EAB was believed to have first arrived on wood packaging materials from Asia. Since its first identification in North America in 2002, the EAB has spread at a rate of about 20 km per year travelling throughout North America, excluding long-range dispersal from human vectors (Prasad et al., 2010). Human-driven vectors including personal vehicles, wood packaging materials, and transportation of wood products are a leading driver of EAB spread in addition to its natural flight capacity (BenDor et al., 2006).

Climate change will also facilitate the spread of EAB. Fahrner et al. (2015) found that maximum flight distances for EAB were reached at 27.9 °C. EAB flight distance was found to be positively correlated with temperature and therefore rising global average temperatures will allow the EAB to travel greater distances in shorter amounts of time. Suitable climates for the EAB will continue to shift north over time with rising global average temperature projections. If spread is not slowed, the EAB is predicted to invade 80% of ash species range by 2020 (Liang & Fei, 2014). Anthropogenic causes have had a significant contribution to the spread of EAB. The

EAB continues to spread throughout North America via means of personal vehicles and wood products in addition to their flight capacity and facilitation by climate change.

2.2 Ash Tree Ecology

Ash is an ecologically, socially, and economically significant species in North America. Ash trees are of the most widely distributed genus in North America (Hanberry, 2014). There are five species of ash native to North America including white, black, blue, and green. Green ash (*Fraxinus pennsylvanica)* tends to be the most widespread due to plantations in Canada and the United States. Green ash, as well as the non-native European ash, are also frequently planted as street trees in urban settings (Hanberry, 2014). Ash is a diverse genus that have been found in greater than 25 forest cover types in North America (Cappaert et al., 2005). Ash support a community of about 282 arthropod species, and as ash declines, the populations of at least 50% of these species are projected to decline (Gandhi & Herms, 2010). Ash species are tolerant of both poorly drained and well drained soils, often present in ecologically sensitive areas such as riparian forests and support riparian functions such as sediment stabilization. For riparian organic matter consumers, ash leaves were found to be either the first or second most preferred food source (Kreutzweiser et al., 2018).

Black ash, in particular, is a culturally significant species to First Nations in Canada and the United States and was commonly used for basket-making (Costanza et al., 2017). Additionally, black ash is classified as threatened under COSEWIC due to its niche habitat requirements (COSEWIC, 2018). Ash species are also economically significant. Ash species are valued at USD $282 billion in urban environments in the United States alone (Haur & Peterson, 2017). The EAB is projected to cause a $280 million annual increase in municipal budgets due to tree and stump removal, replanting, pruning, watering, and fertilization. This significant rate of budget increase is projected to occur between the five to eight year period following invasion, and subsequently decrease between the 9-12 year period once the pest infests the majority of existing ash (Haur & Peterson, 2017). Overall, ash species are a significant genus in North America due to their versatile range of benefits.

2.2.1 Ash Tree Vulnerability

A variety of factors determine the vulnerability of ash trees to EAB invasion, including stress level and species. Due to the lack of coevolution, North American ash species are less resistant to EAB invasion than Asian ash species. Amongst North American ash species the relatively rare blue ash species has been found to be more resistant to EAB attack compared to white ash (Tanis & McCullough 2012). In sampled stands, the survivorship rate of blue ash ranged from 63-71%, where the survivorship of white ash ranged from 0-15%. Tluczek et al. (2011) examined the impact of host tree stress of EAB densities and found that partially and completely girdled trees showed four to five times greater adult EAB capture rates. Statistically significant higher EAB larvae densities were found in stressed ash trees compared to healthy ash trees.

Once the EAB has invaded an area, there is little treatment that can be done to save infected trees. Siegert et al. (2017) found girdled trees were more likely to attract EAB populations than healthy trees. Flower et al. (2013) demonstrated that native bark-foraging bird species such as woodpeckers fed preferentially on ash trees compared to non-ash trees, and specifically favoured trees demonstrating canopy decline relative to healthy canopies. The study found that bark-foraging bird species were effective in eradicating about 45% of EAB from canopy-declining stands and 22% in stand with healthy canopies (Flower et al. 2013). However, in similar studies conducted with high EAB densities, these natural predators were not effective in slowing the spread (Leing & Fei, 2014). The EAB is an aggressively invasive insect and infestations are difficult to detect and manage. While there are means of detecting populations and slowing spread, this can vary based on tree vulnerability and on a stand-by-stand basis.

2.3 Riparian Forest Significance

A riparian zone is defined as a terrestrial area adjacent to rivers, streams, lakes, ponds, and wetlands (Watersheds Canada, n.d.). Riparian forests are ecologically significant and sensitive areas where trees play many important functions. It has been found that species richness is significantly higher in riparian zones compared to reference ecosystems (Jansson et al., 2007). This is especially true in riparian zones with groundwater discharges, which dramatically increases vegetation and species richness (36-209% more species rich), due to the decreased drought stress and higher nitrogen availability. Soil fertility was also found to be significantly higher than upland ecosystems studied (Jansson et al., 2007). Riparian zones were found to be unique areas due to periodic flood pulses creating a dynamic and narrow environment with edge effects of both aquatic and terrestrial ecosystems (Junk et al., 1989). Best practices to promote increased ecological function and integrity of riparian zones include improving channel mobility, re-naturalizing flow regimes, and removal of invasive species (Gonzalez et al., 2017). Additionally, trees are riparian ecosystem engineers, playing a significant role in sediment accretion, bank stabilization, and flow resistance (Gurnell & Petts, 2006). Riparian zones have been identified to contain significant ash populations throughout eastern North America and play a key role in riparian zone structure and ecological integrity (Gurnell & Petts, 2006; Nisbet et al., 2015). Riparian zones fulfill a unique niche in forested areas and represent a corridor which demonstrates characteristics of both terrestrial and aquatic ecosystems. Riparian zones are areas of particular ecologic significance and trees play an important role in facilitating their function.

2.3.1 The Urban Forest & Riparian Zones

Riparian zones in the urban forest provide unique ecological functions and ecosystem services that are of importance for conservation. These benefits are similar to those described in rural forests but play an especially important role in light of urban pollution and extra stress on these areas from being within an urban zone (Reisinger et al., 2016). Forested riparian zones in urban areas were found to be important sources of nitrogen mineralization and increase net nitrification rates, decreasing nitrogen pollution into larger waterways (Reisinger et al., 2016). Treed riparian buffer zones have shown to increase stream resiliency to urban disturbance and mitigate negative impacts of urban disturbance such as runoff, erosion, and decreased nutrient cycling. Nitrogen uptake rates were found to be higher than reference ecosystems (Roy et al., 2006). However, canopy cover was found to reduce stream temperatures and subsequently overall N uptake rates (Solins et al., 2018). Urban riparian canopy provides a habitat corridor for taxa such as amphibians, reptiles, birds, and mammals increasing species biodiversity in urban areas (Solins et al., 2018). Overall, riparian zones play a unique role in the urban forest that aids in mitigation of impacts due to development.

2.4 Knowledge Gaps

There is a gap in the literature concerning the ecological significance of riparian zones in the urban forest, as well as the impacts of canopy loss on these functions. There are many studies concerning chemical composition and nutrient cycling in urban riparian zones with little mention on the function of canopy or trees as a whole. Most studies that do mention canopy function are concentrated to European and Asian geographic locations which do not have the same ash species as North America (Li et al., 2014; Gurnell & Petts, 2006). The closest study which addresses the impacts of the EAB on riparian function in relation to canopy loss in a North American context is an unpublished master's dissertation concerning risk to aquatic communities posed by EAB impacts on riparian zones in Southern Ontario (Nisbet, 2014). More studies must be conducted in North America to better understand the function of riparian zones in the urban forest and the impact a loss in canopy will have in the short-term. This study examines the level of threat the EAB poses to urban riparian canopy cover and its short-term implications on the riparian zone and riparian function. Based on the important role of ash trees in riparian zones, determining how the EAB will affect HRM parks will help inform park management policies in the face of the EAB invasion.

2.5 Summary

This literature review has contextualized the ecology and invasion of the EAB in North America and its subsequent impacts on host species. The subject of importance of riparian zones in the urban forest has been highlighted in the review, with an apparent lack of knowledge on the impacts of tree loss on the function of these riparian zones. The need to understand the impacts of the EAB on a localized scale in such ecologically sensitive areas is of increasing importance as the EAB continues to ravage and spread throughout North American urban and rural forests.

Chapter 3: Methods

3.1 Overview of Methods

This project assessed the implications of the EAB on riparian canopy cover in HRM parks. The implications were determined via collection of tree data in three HRM parks, analysis of spatial distribution, and comparison with current literature and regulations. Based on park size and observed ash-tree distribution, both census and cruise-transect-sampling methods were used to determine ash proportions and distribution in the parks. The data were then examined for spatial patterns among points to determine areas of greatest risk along the riparian zone. Finally, the predicted loss of canopy was contextualized in relevant primary and grey literature to determine if riparian function could be impaired.

3.2 Study Area

The study areas were chosen from the criteria of HRM park properties with a water course where ash trees were present in the riparian zone (within 20 m of the watercourse edge). Three HRM parks - Fish Hatchery Park, Moirs Mill Park and Sir Sandford Fleming Park (Figure 1) - were found to satisfy these criteria and were selected for survey.

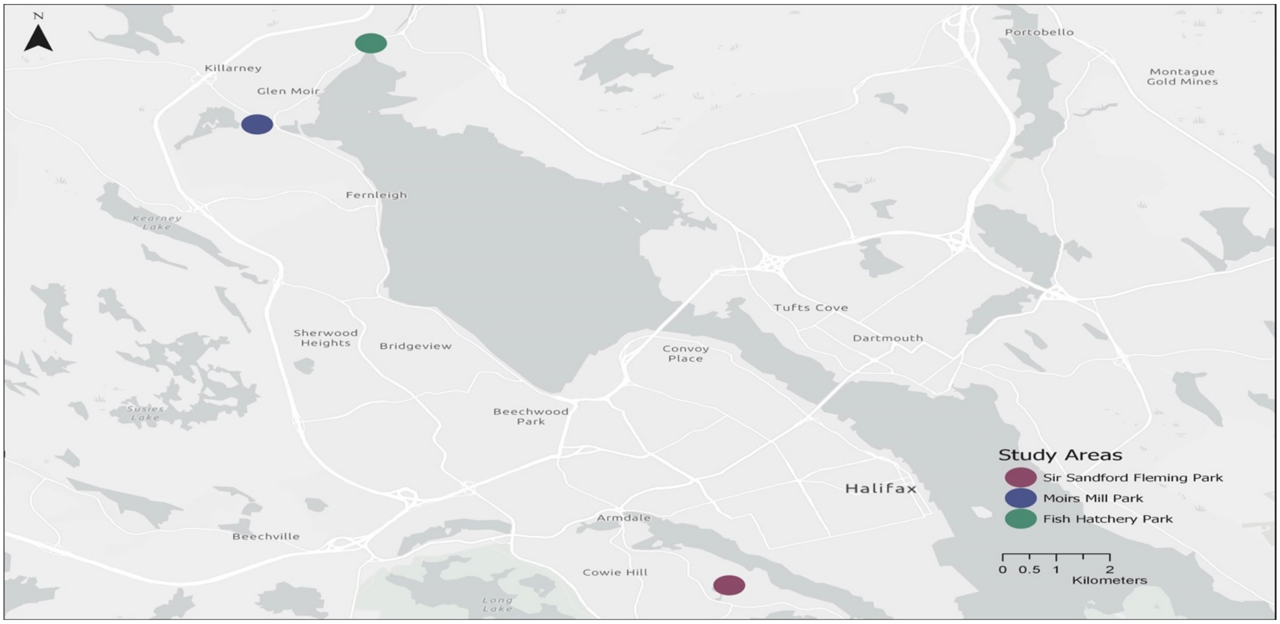


Figure 1. Three study areas selected for riparian canopy assessment (Sir Sandford Fleming Park, Moirs Mill Park, Fish Hatchery Park) in Halifax, Nova Scotia, Canada in fall 2019.

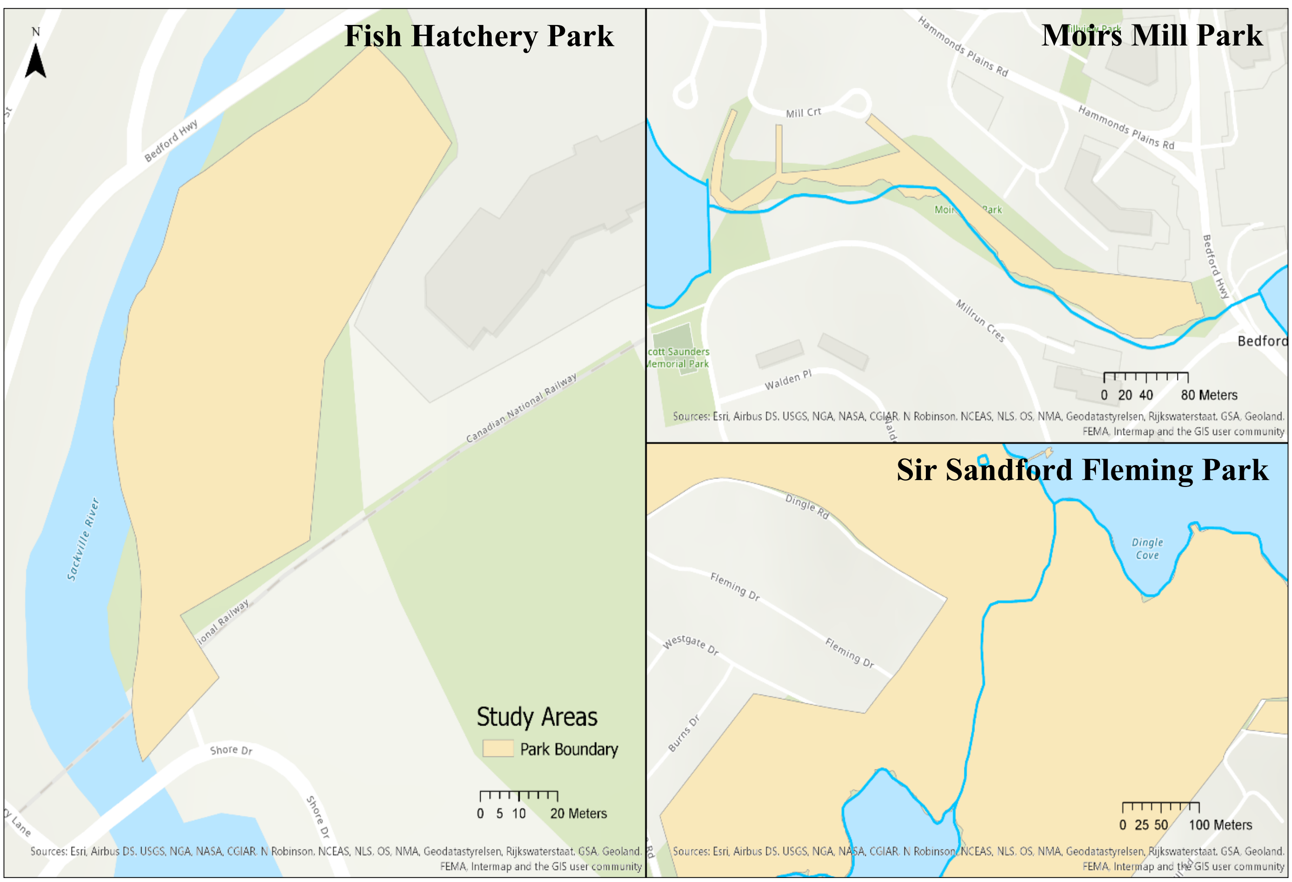


Figure 2. Study areas showcasing HRM park boundaries selected for evaluation in Halifax, Canada. These parks include: (A) Fish Hatchery Park, (B) Moirs Mill Park, and (C) Sir Sandford Fleming Park.

Fish Hatchery Park is 0.64 ha in size and is located next to the Bedford Highway in Bedford. Fish Hatchery Park has 138 m of river front along the Sackville River (Figure 2). Fish Hatchery park extends up to 40 m from the mouth of the Sackville River where it enters into the Bedford Basin. This park is located approximately 1.4 km from DeWolf Park, the nearest known location of EAB infestation at the current time.

Moirs Mill Park is located 23 m from the Bedford Highway in Bedford. The park is 1.35 ha in size (Figure 2). The park is adjacent to an unnamed stream approximately 600 m long, which drains Paper Mill Lake. The park boundary extends to 58 m prior to the mouth of the river that empties into the Bedford Basin. Moirs Mill Park is located approximately 0.53 km from DeWolf Park.

Sir Sandford Fleming Park (also known as Dingle Park) is 38 ha, off Purcell's Cove Road in Jollimore. In the park, there is a 450 m stream that drains Frog Pond and flows into the Northwest Arm. Sir Sandford Fleming Park is approximately 14 km from DeWolf Park (Figure 1C).

3.3 Field Methods

3.3.1 Tree Census

Fish Hatchery Park, due to its small size and the nature of its tree canopy, was selected for a tree census. In this tree census, diameter at breast height (DBH), crown condition, and crown position of every tree (>= 2.54 cm DBH) within the park boundary were measured. Not all trees measured will be in close proximity to the watercourse, which will be factored into analysis of severity of impacts and risk. Basal area was also calculated from DBH. An Arrow 100 Submeter GNSS Receiver was used, in combination with the ArcCollector app on a personal cellular device, to record the coordinates of each tree location.

3.3.2 Cruise Method

Moirs Mill Park and Sir Sandford Fleming Park were both too large to complete measurements for all trees in the riparian zone. Considering that only trees in riparian zones are of interest in this study, sampling points were dispersed along a transect line stretching the length of the stream or river within the park boundaries (Figure 2).

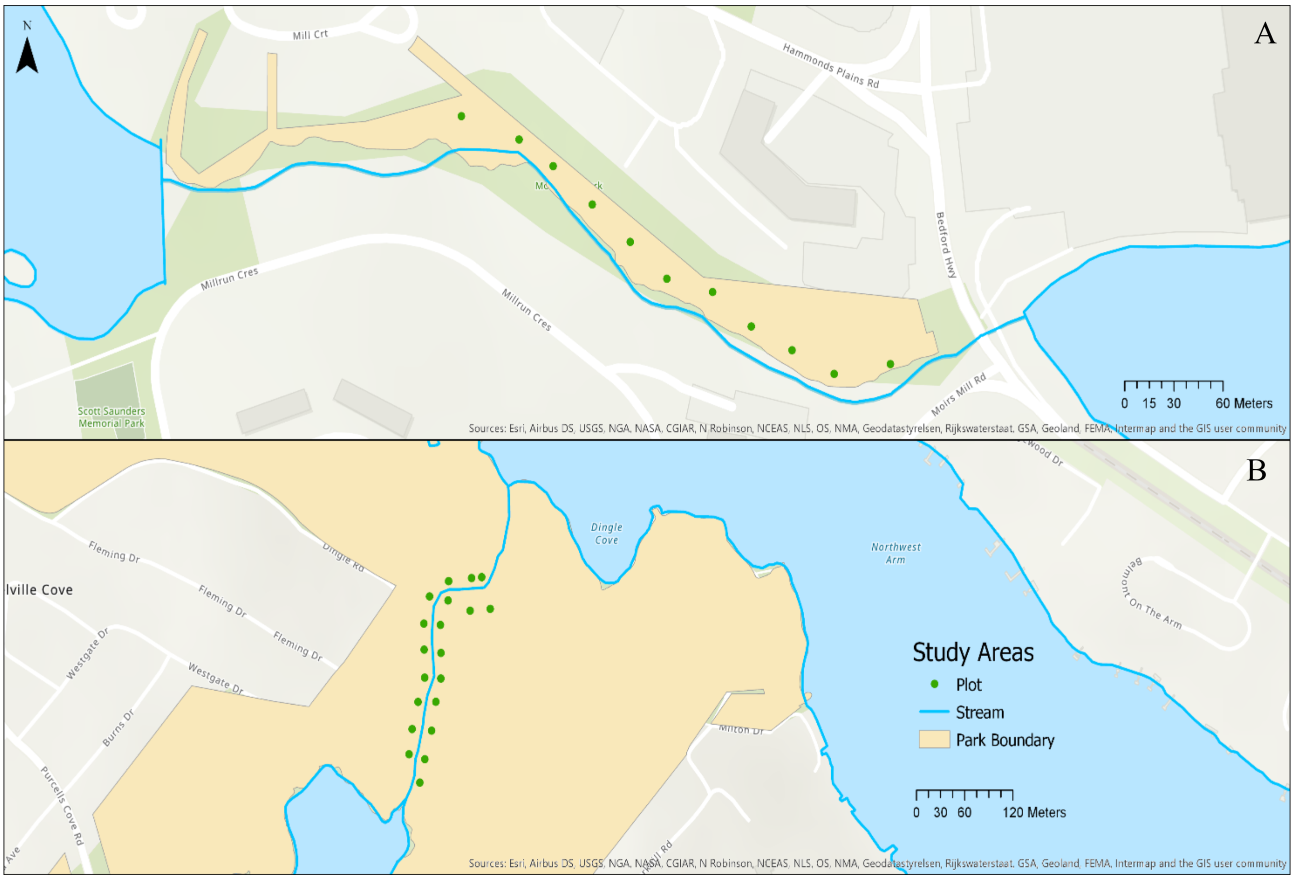


Figure 3. Location of sample plots in (A) Moirs Mill Park and (B) Sir Sandford Fleming Park in Halifax, Canada. Points were sampled in October 2019.

Transects were placed parallel to the stream at a 10 m distance from the edge of the watercourse. For Moirs Mill, one transect line was completed as the park exists only on one side of the river. Eleven points, spaced 30 m apart, were placed along this transect. At Sir Sandford Fleming Park, a transect was placed on both sides of the stream with 10 points spaced 30 m apart on either side of the stream (20 points total).

With transects and points designated, the variable-point (or prism) cruise sample was taken to determine sample size, sample selection, and basal area (BC Forest Service, 2010). Variable-point sampling can be considered a more efficient measurement approach than fixed-plot techniques as variable-point sampling does not require a plot perimeter (BC Forest Service, 2010). Variable-point sampling allows for a coarse species evaluation of the entire length of the watercourse, rather than a detailed evaluation of a portion of the watercourse. Considering that the entirety of the watercourse is of interest, the variable-point method was employed. A 2-BAF (basal area factor) wedge prism was used as it encapsulates the largest number of trees and therefore maximizes potential accuracy, minimizes variance, and provides a detailed evaluation of forest characteristics and basal area (Larsen, 2015). A 2-BAF wedge prism was selected as it included smaller size trees, ensuring measurement of the statistically valid minimum of five trees. Given that the majority of trees within the study areas were around 15-25 cm DBH, a 2-BAF prism was most suitable to capture the statistically ideal number of trees in a sweep, which is about ten trees (Larsen, 2015; BC Forest Service, 2010).

3.4 Measurements

In addition to species identification (genus level when needed) and DBH, the following measurements were taken on each tree counted in all parks:

3.4.1 Crown Position

Relative crown position was qualitatively assessed into one of four categories: suppressed, intermediate, co-dominant, and dominant. Suppressed trees have crowns well below the general level of canopy and receive no direct light. Intermediate trees possess crowns slightly lower than the general level of canopy and receive little direct light from above and none on the sides. Co-dominant trees represent the general level of canopy, receiving full light from above and little on the sides. Finally, dominant trees are generally larger than co-dominant trees, and have crowns extending above the general canopy level (NSDLF 2006). Open-grown trees were considered dominant in the canopy.

3.4.2 Crown Condition

Crown condition was classified as good, fair, and poor based on a qualitative assessment of foliage density, foliage colour, and abundance of fine deadwood. As this became increasingly difficult as autumn progressed, a crown condition of "good" was assumed unless visible deadwood was identified.

3.5 Canopy cover

Locations geo-positioned from the Fish Hatchery survey were extracted from ArcGIS Online and loaded into ArcPro. In combination with ash street tree crown and DBH measurements taken in Halifax and Dartmouth for a previous study (Aryal, 2017), a linear regression equation ($n= 54$, $R^2 = 0.68$, $C_D= 0.1242(DBH) + 4.466 + \varepsilon$) ($\varepsilon$ = error) was derived to determine crown estimates for ash within Fish Hatchery Park under the assumption an existing relationship between DBH and canopy cover as outlined in growth equations which underpin several computer models including i-Tree and OpenTreeMap (McPherson et al.; Westfall & Morin 2012). The equation was derived using ash crown dimensions taken during a previous study (Aryal, 2017). Crown dimension measurements were not conducted in the field due to measurements being taken in late autumn when crowns had begun to recede. Upon completion of ash crown estimates using the derived regression equation, the buffer zone feature in ArcPro was utilized to create circular areas around each tree according to DBH inputted into the equation. These buffer zones illustrate ash canopy cover spatial distribution (Figure 3). The geolocating of ash trees and derivation of crown estimates was completed for Fish Hatchery Park alone because the crowns of the trees in the other two parks are different due to competition for light and space in the forested canopy. Fish Hatchery park is less naturalized, with a mainly open canopy of planted trees distanced from each other.

To assess the proportion of ash canopy present within the total canopy area, three methods of estimating total canopy cover of Fish Hatchery Park were employed to provide a robust estimate and allow comparison of methods. As these methods have their own individual errors, combining and comparing them will provide a more accurate and complete insight into the canopy cover distribution in Fish Hatchery Park. Firstly, an estimate was created using Lidar imagery and normalized difference vegetation index (NDVI) (July 2019, resolution 3 x 3 m). To isolate tree canopy specifically, only NDVI values greater than or equal to 0.69 were selected. Typically, NDVI values greater than 0.6 indicate canopy vegetation (Weier & Herring, 2000). Upon comparison of the canopy estimate with areal imagery, NDVI values greater than 0.69 were assessed to provide the most accurate approximation of canopy cover in Fish Hatchery Park.

Second, a visual estimation of Fish Hatchery canopy cover was determined by manual creation of shapefiles encapsulating areas of canopy cover based on visual evaluation of July 2018 satellite imagery. Lastly, 100 random sample points were placed on July 2018 satellite imagery in ArcPro and evaluated for canopy presence to determine a proportional canopy estimate of Fish Hatchery Park. These points were evaluated to be either “tree canopy” or “no tree canopy” to generate a proportional canopy cover estimate (see iTree Canopy (2011) for details). Two maps were created to demonstrate ash crown area relative to total canopy cover, one using the NDVI estimate, and another illustrating the canopy cover from visual estimation.

3.5.2 Canopy Cover Loss

Each point sampled in Moirs Mill Park and Sir Sandford Fleming Park was analyzed to determine proportion of trees, according to both stem count and basal area, that are ash versus proportion of trees of other species. Under the assumption of 100% canopy cover in the riparian zone, samples from each point were used to estimate percent canopy cover loss at each point. Observations in the field confirmed an approximate 100% canopy cover. Basal area and stem density were also compared to estimate relative tree size. Canopy loss was compared across all points to determine areas of greatest risk along the riparian zone.

3.6 Limitations

The original study area was designed to be restricted to the Bedford Area, as Bedford was the initial location of discovery of the EAB in Nova Scotia (CFIA 2018). There was a limiting criterion of ash trees in the riparian zones of HRM parks in the Bedford area, leaving me with just two parks. A search for other Bedford-area HRM parks with ash in the riparian zone identified none, so I added Sir Sandford Fleming Park in the Northwest Arm.

Identifying and surveying more parks would provide a more comprehensive result as to the full extent of the implications of the EAB in these special ecosystems. Unfortunately, the time limitation of two months for data collection was not adequate to include more parks in the study.

There is also a limitation concerning species-level ash identification. As data collection ran into the fall season, many ash were near-leafless and only a genus level identification was recorded. This limited the detail of analysis that could be completed and the implications that could be drawn. There is a similar limitation for crown condition identification, as all trees were classified as good unless visible deadwood. This classification does not consider trees in declining condition that may not have visible deadwood present, limiting the detail of analysis that could be conducted.

Chapter 4: Results

The following section describes the distribution of ash trees for three parks within the HRM: first, a map depicting canopy spatial distribution of Fish Hatchery Park, followed by ash frequency, basal area, and species distribution for Moirs Mill Park and Sir Sandford Fleming Park.

4.1 Fish Hatchery Park

The majority of ash trees within Fish Hatchery Park exist on the edge of the Sackville River (Figure 3). Ash individuals residing in the northern area of the park were not captured by the NDVI canopy cover estimation but were accounted for in the qualitative visual estimation of canopy cover (Figure 3). Overall, NDVI estimated a larger canopy area, while the visual estimation produced a more dispersed tree canopy. Ash trees in closer proximity to one another produced similar canopy projections. The isolated ash tree in the centre of the park resulted in the largest crown projection. 75% of ash trees measured were classified as co-dominant or higher (dominant).

Figure 3. Map of Fish Hatchery Park, Nova Scotia crown cover estimates by (A) NDVI (Normalized Difference Vegetation Index), and (B) visual estimation. Imagery used to derive estimates was taken in July, 2019 (A), and August, 2018 (B).

Overall, the NDVI and visual estimation of canopy area produced similar results, with 41.3% and 39.9% total canopy cover estimations respectively, with ash crown being shown to account for 28.8% and 32.9% of total canopy (Table 1). The random sample estimation (n = 100) produced the largest estimation (45% canopy cover).

Table 1. Canopy cover estimates by NDVI (Normalized Difference Vegetation Index), visual estimation, and random sample estimation in hectares (ha) and as a proportion (%) in Fish Hatchery Park, Nova Scotia. "Ash Canopy Cover (%)" percentage represents the proportion of total canopy that consists of ash tree canopy.

	NDVI Estimation	Visual Estimation	Random Sample Estimation
Total Canopy Cover (ha)	0.2687	0.2594	-
Total Canopy Cover (%)	41.3	39.9	45.0
Ash Canopy Cover (ha)	0.086	0.086	-
Ash Canopy Cover (%)	28.8	32.9	-

4.2 Moirs Mill & Sir Sandford Fleming Parks

The genus distribution of sampled plots within Moirs Mill Park consists of a nearly equal proportion of maple, hemlock, and ash (Figure 4). Non-conifer species exhibit higher stem counts relative to conifer species within the plots.

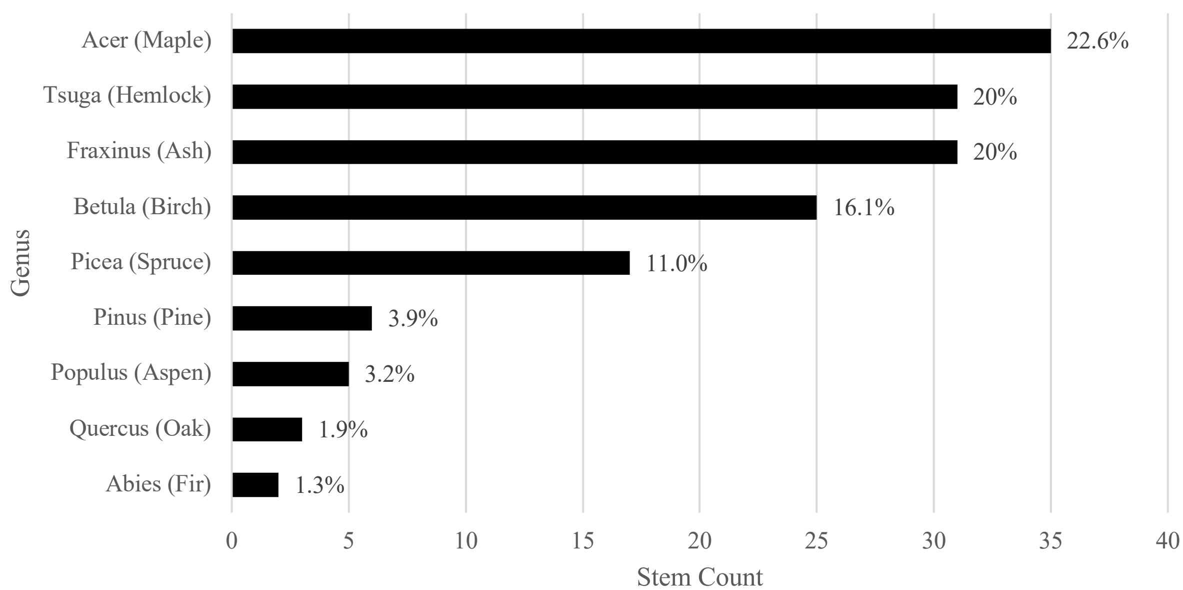
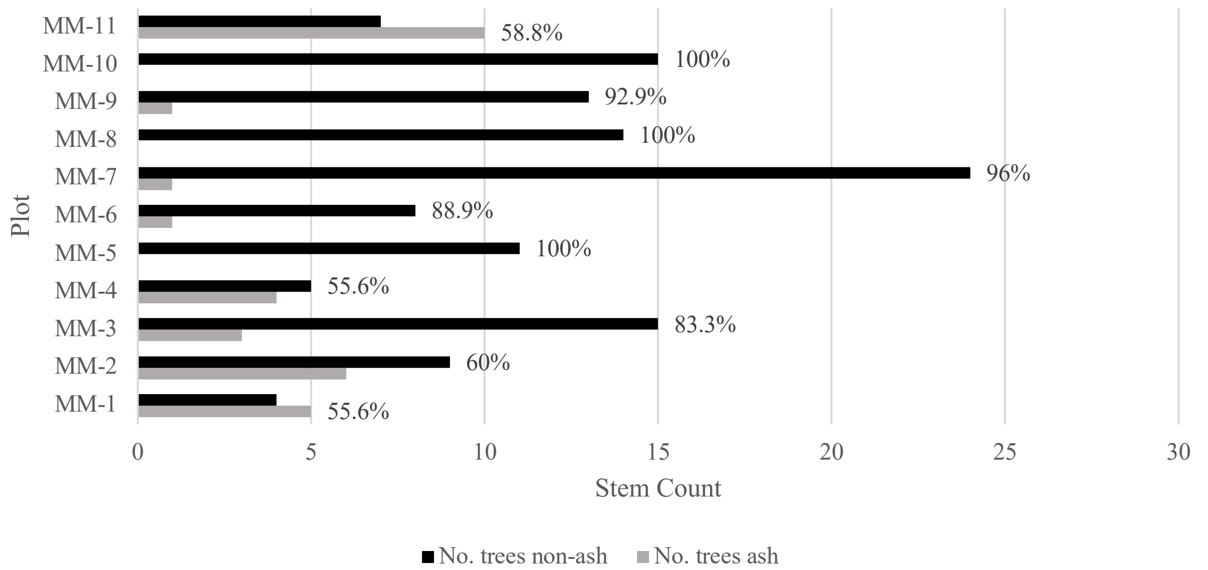


Figure 4. Genus distribution across all 11 plots in Moirs Mill Park, Nova Scotia, in October 2019. Percentages adjacent to bars indicate proportion of total stem count each genus accounts for.

In Moirs Mill Park, ash were present in eight of the 11 plots (Figure 5). The largest number of ash trees were present near the mouth of the river (plots MM-1 – MM-4), and towards where the stream meets Paper Mill Lake (MM-11). Distribution by basal area and counts produced results of similar proportions, with the exception of plots MM-2 and MM-4. Total mean DBH for Moirs Mill Park was 20.5 cm ± 13.3 cm. Mean ash DBH was 20.1 ± 8.0 cm. Approximately 71% of ash trees were classified as co-dominant.

A



B

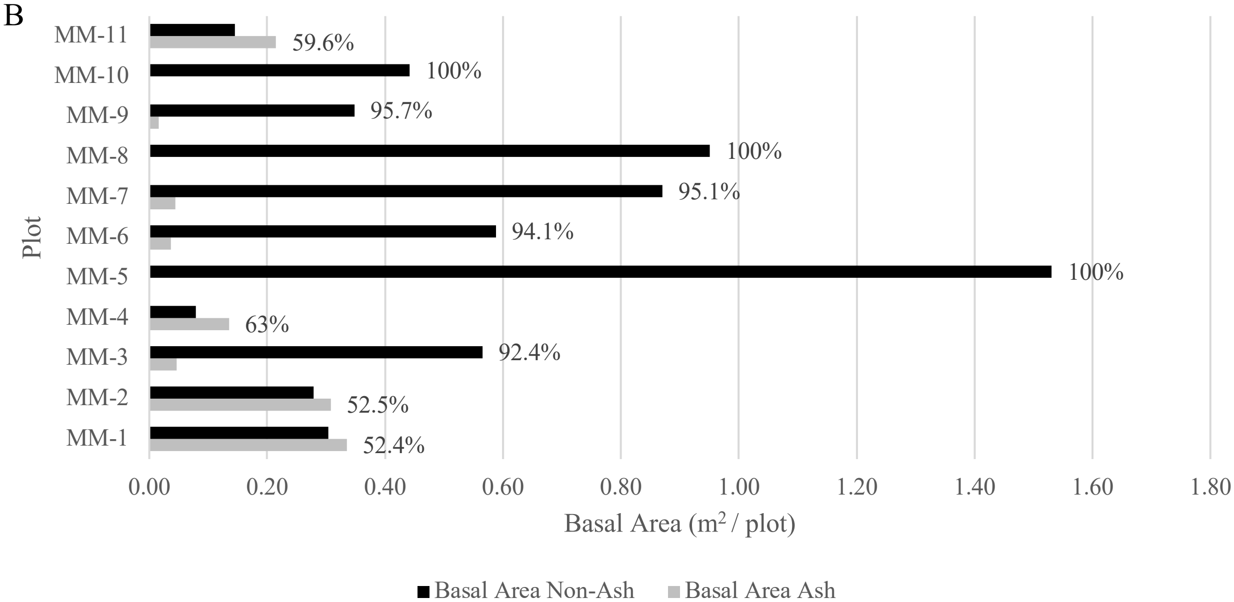


Figure 5. Species distribution of ash (Fraxinus genus) versus non-ash by (A) frequency and (B) basal area (m^2) over 11 plots in Moirs Mill Park, Nova Scotia in October 2019. Data labels represent proportion of basal area per plot.

As Table 2 shows, five of the 11 plots sampled within Moirs Mill Park will exist below the regulation of 20 m^2/ha of living basal area within 20 m of a watercourse (riparian zone) greater than 50 cm in bedwidth should the EAB eliminate all ash trees sampled, as outlined by the *Wildlife Habitat and Watercourses Protection Regulations* under the *Forests Act* of Nova

Scotia (2001). However, all but two of these plots were already below regulation prior to projection without ash trees. Plots near the mouth of the river (plots MM-1 – MM-4) would be most impacted by EAB arrival, as a significant proportion of these plots consist of ash trees. In plots with an ash presence, ash accounts for a significant proportion of the sample. Additionally, observations from the field suggest that, should the EAB eliminate all ash trees, plots MM-1, MM-2, and MM-11 have significant potential to be in violation of the *Habitat and Watercourses Protection Regulations* prohibiting creating canopy openings of larger than 15 m within the riparian zone (2001).

Table 2. Ash metrics in relation to basal area, as well as total trees measured, across 11 plots in Moirs Mill Park, Nova Scotia, in October 2019. Values in red indicate the basal area for that plot is below regulation outlined by the Nova Scotia Forests Act (1989). "Remaining basal area" refers to a situation where the EAB has arrived and eliminated all ash trees sampled. SD = Standard Deviation.

Plot ID	# trees per plot	# ash trees	% ash	Basal area per plot (m^2)	Basal area (m^2/ha)	Remaining basal area (m^2/ha)
MM-1	9	5	55.6	0.64	18	8
MM-2	15	6	40.0	0.59	30	18
MM-3	18	3	16.7	0.61	36	30
MM-4	9	4	44.4	0.21	18	10
MM-5	11	0	0.0	1.53	22	22
MM-6	9	1	11.1	0.62	18	16
MM-7	25	1	4.0	0.91	50	48
MM-8	14	0	0.0	0.95	28	28
MM-9	14	1	7.1	0.36	28	26
MM-10	15	0	0.0	0.44	30	30
MM-11	17	10	58.8	0.36	34	14
Mean	14	3	21.6	0.67	28	23
SD	4.8	3.2	23.3	0.37	9.6	11.4

Plots sampled within Sir Sandford Fleming Park consisted largely of non-conifer species of birch and maple, with a lesser presence of the conifers hemlock and spruce (Figure 6). Ash represents a relatively low proportion of genus distribution compared to other present genera and other parks sampled. The woodlands present in Sir Sandford Fleming Park contain a smaller ash presence and a greater proportion of conifer species than Moirs Mill Park (Figure 4).

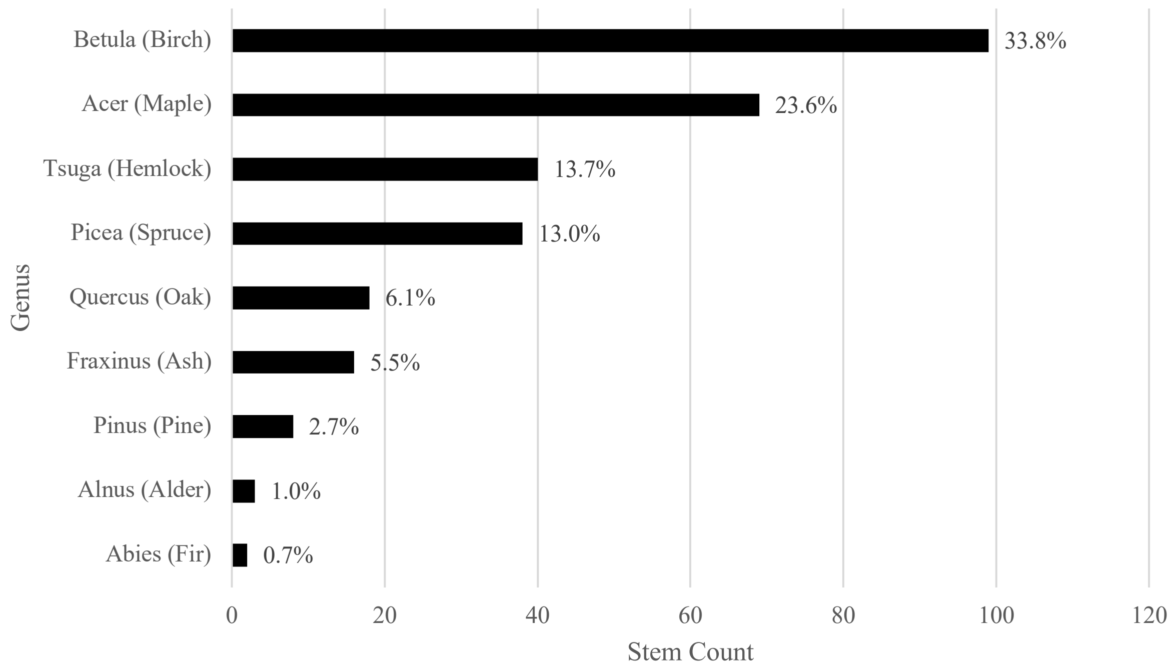


Figure 6. Genus distribution across all 20 plots in Sir Sandford Fleming Park, Nova Scotia, in October 2019. Percentages adjacent to bars indicate proportion of total stem count each genus accounts for.

As shown in Figure 7, ash were present in less than half (eight) of the 20 plots sampled. The distribution of ash across plots is spatially dispersed, with a small clump near the middle of the stream course (SF-E-8 – SF-E-9). Beginning with plot SF-E-2, the following plots are across from one another on opposites sides of the stream (e.g. SF-E-2 is opposite SF-W-1). Distribution by basal area and frequency produce similar patterns to one another, which are largely consistent across all plots. Of the 16 ash trees measured, 81% were classified as co-dominant or higher.

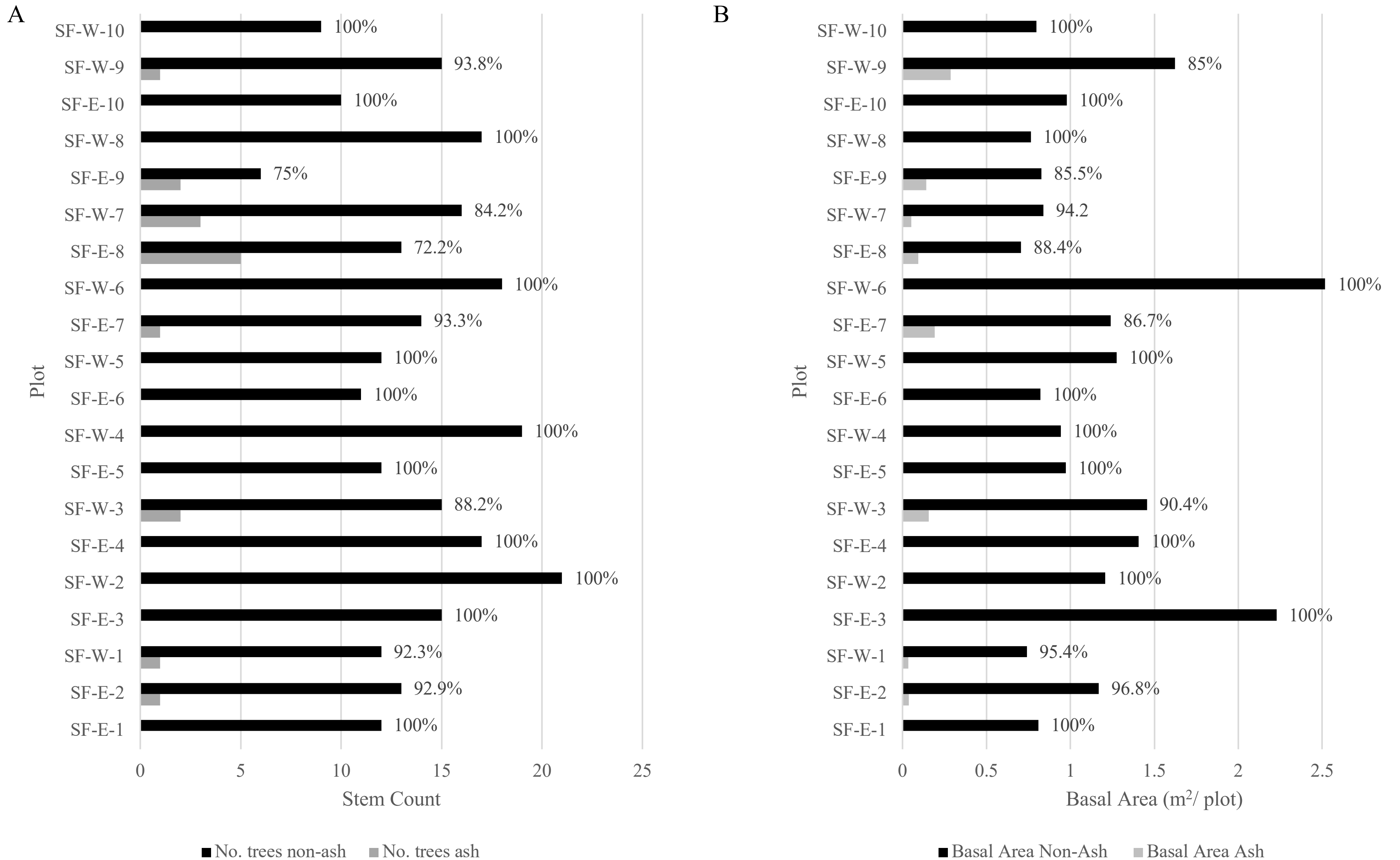


Figure 7. Species distribution of ash (*Fraxinus* genus) versus non-ash by (A) frequency and (B) basal area (m2/ha) over 20 plots in Sir Sandford Fleming Park, Nova Scotia in October 2019. Plots with labels beginning with "SF-E" indicate the plot is on the east side of the stream, and "SF-W" indicates the plot is on the west side of the stream.

Should the EAB arrive at Sir Sandford Fleming Park and eliminate all ash trees sampled, only one (SF-E-9) out of the 20 plots sampled would fall below the living basal area regulation outlined by the *Wildlife Habitat and Watercourses Protection Regulations* (2001) (Table 3). However, this plot is already currently below regulation. An additional plot (SF-W-10), is also below regulation, but there are no ash present and it is therefore not predicted to change with the arrival of the EAB. Ash does not make up a significant proportion of any plot sampled.

Table 3. Ash metrics in relation to basal area, as well as total trees measured, across 20 plots in Sir Sandford Fleming Park, Nova Scotia, in October 2019. Values in red indicate the basal area for that plot is below regulation outlined by the Nova Scotia Forests Act (1989). SD= Standard deviation.

Plot ID	# trees per plot	Basal area (m^2)	# ash trees	% ash	Basal area (m^2/ha)	Remaining basal area (m^2/ha)
SF-E-1	12	0.81	0	0.0	24	24
SF-E-2	14	1.21	1	7.1	28	26
SF-W-1	13	0.78	1	7.7	26	24
SF-E-3	15	2.23	0	0.0	30	30
SF-W-2	21	1.21	0	0.0	42	42
SF-E-4	17	1.41	0	0.0	34	34
SF-W-3	17	1.61	2	11.8	34	30
SF-E-5	12	0.97	0	0.0	24	24
SF-W-4	19	0.94	0	0.0	38	38
SF-E-6	11	0.82	0	0.0	22	22
SF-W-5	12	1.28	0	0.0	24	24
SF-E-7	15	1.43	1	6.7	30	28
SF-W-6	18	2.52	0	0.0	36	36
SF-E-8	18	0.80	5	27.8	36	26
SF-W-7	19	0.89	3	15.8	38	32
SF-E-9	8	0.97	2	25.0	16	12
SF-W-8	17	0.76	0	0.0	34	34
SF-E-10	10	0.98	0	0.0	20	20
SF-W-9	16	1.91	1	6.3	32	30
SF-W-10	9	0.80	0	0.0	18	18
Mean	14	1.2	1	5.4	29	28
SD	3.7	0.5	1.3	8.6	7.3	7.2

Crown condition distribution was aggregated for Moirs Mill and Sir Sandford Fleming as a significant majority of the conditions were classified as “good”. The proportion of ash classified as "poor" is four times higher relative to proportion of non-ash under the same classification. Trees classified as "fair" and "good" obtain similar proportions across ash and non-ash populations.

Table 1. Crown condition distribution between ash (*Fraxinus* genus) and non-ash species in Moirs Mill and Sir Sandford Fleming Park, Nova Scotia, in October 2019. Percentages represent a proportion of total ash or total non-ash trees measured in the two plots.

	Proportion Ash	Proportion Non-ash	**# Trees Total**
Poor	4.2%	1%	**6**
Fair	6.3%	7.5%	**33**
Good	90%	91.%	**409**
# Trees Total	**48**	**400**	**448**

Chapter 5: Discussion

The canopy of Moirs Mill Park was found to be modestly threatened by EAB arrival. Due to the inconsistency between proportion of stem count (Figure 5A) and proportion of basal area (Figure 5B) in plots MM-2 and MM-4, it can be inferred that the ash trees present in those plots are larger than average. Although tree size has not been found to be a factor in host selection, larger trees may represent a greater proportion of canopy and represent co-dominant or dominant crown positions, therefore having a higher magnitude of impact if eliminated by the EAB (Siegert et al., 2010).

The EAB also preferentially feeds on stressed ash trees (Tluczek et al., 2011). Plots MM-1 to MM-4 are in closest proximity (within 100 m) to urban infrastructure stress factors such as poor soils, vandalism, and road salts. When these factors are considered along with the natural competition occurring in the riparian park ecosystem, plots MM-1 to MM-4 are assessed at greatest risk for short-term canopy loss. Three of the 11 plots sampled were below the minimum 20 m^2/ha regulatory basal area for riparian zones (Table 2) (Government of Nova Scotia, 2001). The basal areas of these plots, along with two plots currently above regulation, will be further reduced as a result of EAB arrival. Considering that plots containing significant ash proportions represent roughly 120 m (4 plots, 30 m spacing) of the 428 m watercourse and ash account for 20% of all trees measured, short-term canopy coverage is likely to be impacted, but at a modest degree of magnitude.

Given the current species distribution present in Moirs Mill (Figure 4), existing species will fill gaps in riparian canopy in a relatively short amount of time (within 10 years) (Rankin & Pickett, 1989). It is likely that the gap in canopy will be filled by a combination of tree regeneration and a lateral expansion of existing crowns. Red maple is a fast-growing early successional species which accounts for the majority of trees sampled and is likely to be the primary species established in the canopy gap after ash mortality in the short term (Rankin & Pickett, 1989). However, shade-tolerant species are also capable of closing small canopy gaps (Muscolo et al., 2014). Over time, the shade-tolerant and long-lived species present, such as yellow birch and eastern hemlock, may fill gaps in the canopy caused by the EAB through regeneration and canopy expansion (Mosseler et al., 2003).

Sir Sandford Fleming Park possessed the smallest concentration of ash trees sampled. The 20 plots sampled only contained 16 ash trees across eight plots out of a total of 293 trees measured (5.5%) (Figure 6). Four plots accounting for 10 of the 16 ash trees measured (SF-E-2, SFW-1, SF-E-8, SF-W-7) are located across the watercourse from each other on parallel transects. The clumped distribution of ash present streamside, located near the centre of the stream, decreases resilience in these areas and increases risk to riparian canopy coverage. However, when evaluating proportion of ash frequency (Figure 7A) against proportion of ash basal area (Figure 7B), it was found that ash present in plots SF-E-8, SF-W-7, and SF-E-9, were relatively small trees, thus decreasing the risk to short-term canopy coverage. EAB invasion resulting in ash mortality would not decrease the basal area of any plot sampled below regulation established by the Government of Nova Scotia (2002), only further reducing one plot already existing below regulation. One additional plot exists under regulatory basal area but contains no ash and therefore would not be impacted by EAB arrival (Table 3). Riparian woodlands sampled in Sir Sandford Fleming Park are dense and contain a small proportion of ash. It is unlikely that riparian canopy will be impacted should the EAB arrive and eliminate the ash population in this park.

The difference between ash populations in these two parks may be attributed to stream size and succession stage. The stream present in Moirs Mill was much wider with greater stream flows, creating an ecosystem more richer in moisture. This impacts which species will be present and the rate at which they can grow. Considering ash species typically prefer more moist environments (Burns & Honkala, 1990), the higher concentration of ash in Moirs Mill Park is not unexpected. Furthermore, the forest within Sir Sandford Fleming Park has been protected since 1908 when it was donated as park, while Moirs Mill Park has only been given park status in recent decades (Halifax Regional Municipality, 2019). The amount of time free of disturbance allowed Sir Sandford Fleming Park to reach a late successional stage dominated by large hemlocks and yellow birch, while Moirs Mill has yet to reach the full extent of this stage.

Three independent estimation methods approximated the total canopy coverage of Fish Hatchery Park to exist between 39% and 45% (Table 1). Imagery provided by ESRI taken in August 2018 was used to conduct both the random sampling and visual estimation. Errors in estimations may be due to the use of separate Lidar imagery taken in July 2019 to complete the NDVI estimate. Furthermore, the pixel size (3 m x 3 m) of the imagery used for the NDVI estimate was larger than optimal for an analysis at this scale. Pixel values with NDVI values greater than 0.69 were considered trees, which may also account for discrepancies as some canopy may have lower NDVI values and will not be captured, or some grasses may have high NDVI values and will be captured. Visual estimation and random sampling, although also limited, were introduced to account for any areas of canopy overlooked by NDVI and provide additional approximation methods to achieve a robust estimate.

Ash trees in closer proximity to the watercourse exhibit greater risk to riparian canopy and should be considered with a higher magnitude of importance. In Fish Hatchery Park, 55% of ash trees measured exist within close proximity (20 m) of the river, placing a greater risk to river shading should all ash be eliminated. However, ash did not account for a significant proportion of total trees streamside, accounting for nine of the 54 trees measured within 20 m of the river (16.7%), reducing the potential for increased risk. In sum, ash trees in closer proximity to the watercourse have the potential to create greater impacts, but the density of ash was not dominant enough to be relevant.

Fish Hatchery Park is functionally different than the two other parks sampled as its tree population, at least on the level ground, consists mainly of dispersed, planted trees. This environment will respond differently to loss of canopy than the naturalized woodlands in the other parks. Tree regeneration is unlikely to happen if ash trees are lost from this park. Furthermore, the loss of the large ash in the centre of the park cannot be mitigated in the short term (Figure 3). Ash were estimated to account for between 28% and 33% of total canopy cover in Fish Hatchery Park. Losing nearly a third of the canopy cover in a non-naturally treed park with significant portions of open canopy will decrease riparian canopy cover in this park. Factors such as total ash canopy estimations, a mean ash DBH of 19.71 cm, and 75% of ash sampled being classified as co-dominant or higher suggests that ash are an important factor in riparian canopy cover in this park. In the short term, EAB arrival and subsequent elimination of all ash trees may have a moderate impact on riparian canopy cover in Fish Hatchery Park.

An additional consideration when evaluating the implications of the EAB on riparian canopy is the potential for canopy loss to degrade riparian function in the microclimate. The loss of canopy can impact a variety of factors pertinent to riparian health. Loss of canopy has been shown to increase riparian health indicators such as daily maximum temperatures, stream temperature, rates of evaporation in riparian microclimates, and wind speeds (Guenther et al., 2012). The factor most widely researched and significantly impacted by canopy loss is stream temperature. Gaps in the canopy allow for increased incoming solar radiation, increasing average stream temperatures. Increased stream temperatures can decrease oxygen availability in aquatic habitats and alter fish metabolism (Dodds, 2002).

Riparian canopy can also absorb and intercept precipitation, stabilizing streambanks. Riparian vegetation has been shown to increase soil strength of streambanks by 2 to 18 kPa (Simon & Collison, 2001), and deciduous canopy typically intercepts between 10 to 20 percent of annual precipitation (Coppin & Richards, 1990). Ash leaf litter is also valuable to the riparian ecosystem, as it has been shown to be a preferred food source for aquatic macroinvertebrates (Nisbet, 2014). These impacts are more extreme in narrow streams less than 3 m in bankfull width (Moore et al., 2005). Overall, riparian canopy can influence a variety of factors influencing riparian function, particularly in narrow streams.

Should the EAB eliminate all ash trees within Fish Hatchery Park, riparian function will not significantly be altered. Fish Hatchery Park accounts for 138 m of the Sackville River, 40 m from its mouth which flows into the Bedford Basin. The Sackville River is 40 km long and drains an area of 155 km^2 with high stream flows and velocity (Sackville Rivers Association, n.d.). Bankfull width in Fish Hatchery Park ranges between 5 m and 20 m, increasing as it nears the Basin. This is a major river system which drains a large area, increasing its resilience to disturbance in small portions of the river. Additionally, the portion of the Sackville River adjacent to Fish Hatchery is largely unshaded. The large width and unshaded nature of the river reduces the significance of canopy cover on riparian function. Ash canopy did not account for a significant proportion of canopy measured within 20 m of the river (riparian zone), minimizing bank stabilization and stream temperature impacts. The ash canopy in Fish Hatchery Park is not large enough in area to produce an impact on the microclimate of a large and fast waterbody such as the Sackville River.

Moirs Mill Park is at a low risk of decreased riparian function as a result of EAB invasion by factors including increased stream temperature, evaporation, and wind temperatures. The factor of greatest concern to Moirs Mill Park is bank stabilization. The Moirs Mill riparian zone approximates 100% canopy coverage, making it more susceptible to riparian function impacts due to loss of canopy than largely unshaded waterbodies. Ash were found to be a dominant tree type accounting for 20% of all trees sampled and 15.7% of total basal area. A study conducted in British Columbia reported that removal of 50% basal area from a riparian zone with a 1.5 m stream width was found to increase stream temperature by 2 degrees Celsius (Guenther et al., 2012). Although stream widths ranged from 5 m to 15 m, two of the eleven plots sampled (MM-1 & MM-11) were projected to lose at least 50% of their basal area as a result of EAB invasion (Table 1). This generates a degree of risk to factors such as stream temperature but is unlikely to make an impact in this park due to large stream width. The stream within Moirs Mill has a high velocity and drains a substantial lake. Most ash present exist within 1 m of the stream, with some clumps of ash growing in the active channel. Streambank soil strength in Moirs Mill may be decreased as a result of ash elimination, increasing bank erosion and stream sediment concentration. Overall, riparian function is at a low risk of being impacted by the EAB, with bank stabilization being the factor of greatest concern.

The EAB does not pose a threat to riparian function as a result of canopy loss in Sir Sandford Fleming Park. Stream width of Sir Sandford Fleming Park is the lowest of those surveyed, averaging 1.5 m bankfull width. The narrow stream width places this park at greatest risk for degraded riparian function as a result of EAB-induced canopy loss. However, only 16 ash trees were present across 20 plots and 293 total trees (Figure 7). The ash present in Sir Sandford Fleming were also relatively dispersed, further decreasing risk of significant canopy openings and resulting riparian impairment. The low density and basal area of ash present prevent the EAB from having significant negative implications on riparian function in Sir Sandford Fleming Park.

An important limitation of the study is the lack of primary crown dimension data. For Fish Hatchery Park, a regression equation to estimate crown cover was derived from HRM street tree data due to the planted and spaced nature of tree distribution in that park. A similar estimate could not be made in the naturalized woodland ecosystems present in the other two parks. Furthermore, the r^2 value indicates that 68% of the variation in DBH and crown diameter is explained by their relation, indicating some degree of error in the crown estimates made. Obtaining crown measurements would increase the accuracy of results and allow for stronger inferences to be made. Although inferences regarding canopy cover can be made by evaluating basal area, exact estimates of total ash canopy cannot be estimated. Overall, lack of crown estimates taken in the field were a limitation that should be accounted for in future studies of this kind.

Another limitation, specific to Moirs Mill Park, includes the park boundary itself. Only one side of the stream is located within HRM park boundary, the other side belonging to various residential property owners. There is potential for large canopy openings greater than the 15 m minimum regulation to be created should ash presence account for a similar distribution of stem count and basal area on the unsampled side. Additionally, only 360 m of the 530 m watercourse could be sampled due to upstream sample points being within private property. Overall, the park boundaries of Moirs Mill Park limited the ability to comprehensively evaluate and sample the entire riparian zone.

The lack of species-level identification poses another limitation to the study. Ash species have been shown to have varying resistances to the EAB, with blue ash being the most resistant of North American ash species (Tanis & McCullough, 2012). Green ash have also been shown to be particularly susceptible due to lower relative amounts of volatile compounds (Pureswaran & Poland, 2009). Furthermore, black ash is a species valued and utilized by Indigenous communities for purposes such as basket-making (Diamond & Emery, 2011). Obtaining species-level identification would allow for a finer-detailed analysis by evaluating the relative susceptibility of each species on the context of its distribution. Species-level identification would also allow areas of greatest risk to culturally significant species to be identified and prioritized. In sum, genus-level identification limits the analysis to a coarse evaluation of implications.

Overall, this study provided a coarse evaluation of the implications of the EAB on canopy cover and riparian function in three HRM parks. There is an opportunity for a more detailed and insightful evaluation in future studies. Given the important role of ash trees in riparian zones, all parks within the HRM should be evaluated with species-level identification and detailed crown measurements. Ideally, additional measurements surrounding the riparian microclimate such as stream temperature, velocity, air temperature, and wind speed should be taken to provide a baseline prior to any EAB-induced impacts. Fixed-area plots should also be established to provide finely-detailed areal ash densities. An expansion of this study has the potential to address prominent knowledge gaps in the impact of riparian canopy on both mechanical and hydrologic riparian function. Furthermore, the distribution of ash within the riparian zones of all HRM parks is not known. This study found prominent ash distributions in two of three parks evaluated, suggesting that further studies should be conducted to achieve a more complete evaluation of the implications of the EAB on riparian zones within all HRM parks.

Chapter 6: Conclusion

This study aimed to understand the implications of the EAB on short-term riparian zone canopy cover in three HRM parks. The study sought to address the research question: what are the implications of the EAB on short-term canopy cover in riparian zones in three HRM parks? To address this, total crown cover, ash crown cover, ash spatial distribution, genus distribution, and basal area distribution were identified. Additionally, data were contextualized within relevant primary and grey literature to determine if riparian function would be impaired. The study examined the significance of impacts to both canopy coverage and riparian function that would be caused by the EAB should invasion and elimination of all ash occur.

Fish Hatchery Park contains an estimated 39 to 45% total canopy coverage. Ash canopy coverage accounted for approximately 30% of total crown area in the park (Table 1), with the majority of ash present riverside (Figure 3). Upon reviewing relevant literature and considering stream size in relation to ash presence, it was found that the EAB will have a moderate impact on canopy coverage, but a negligible impact on riparian function in Fish Hatchery Park. Given that regeneration is less likely to happen in this park, it is recommended that further tree-planting be considered in preparation for potential ash canopy eradication.

Moirs Mill Park and Sir Sandford Fleming Park were evaluated using cruise transect lines. One hundred percent riparian canopy coverage was assumed. In Moirs Mill Park, four of the 11 plots were found to contain significant ash presence (Figure 5). Five of these plots are currently below basal area regulation, and two are at risk of violating canopy opening regulation (Table 2). When evaluating these factors in addition to stream characteristics, it was found that the EAB will moderately impact canopy coverage and have a low impact on riparian function in Moirs Mill. In Sir Sandford Fleming, ash were found in only eight of the 20 plots and no plots were found to be reduced below regulatory basal area (Table 3). The small presence of ash indicates that neither canopy coverage nor riparian function will be impacted by possible EAB arrival in this park. Considering the natural regeneration likely to occur in Moirs Mill Park and lack of ash presence in Sir Sandford Fleming Park, management interventions by the HRM may not be necessary in these parks.

Prior to this study, the significance of the ash population within HRM park riparian zones was not known. Furthermore, the implication of the loss of ash canopy on riparian function in these parks had also not been considered. The recent identification of the EAB in Nova Scotia has created a need to fill the knowledge gap of the implications of this invasive species on canopy coverage in locations specific to Canada's urban forest and more specifically, in riparian zones of the urban forest.

To best serve urban forest managers and decision-makers, future studies should include as many parks as possible to achieve a more comprehensive evaluation of the impacts. The HRM contains many more parks than those sampled in this study, which could potentially have a greater presence of ash and are therefore at greater risk. Additionally, future studies could better address risk to riparian function by capturing stream data and integrating it into analysis. Ash canopy and stream characteristics vary by location, creating a need for more parks to be surveyed to produce a comprehensive evaluation of the risk to HRM parks.

References

Aryal, B. (2017). Economic and biophysical implications of alternative street-tree spacings in Halifax, Canada (Unpublished master's dissertation). Dalhousie University, Halifax, NS.

BC Forest Service. (2010). *Cruising Manual*. Retrieved from ftp://ftp.for.gov.bc.ca/HVA/external/!publish/Web/Manuals/Cruising/2010/CruisingManualMasterAmend2.pdf

Bendor, T., Metcalf, S., Fontenot, L., Sangunett, B., & Hannon, B. (2006). Modeling the spread of the Emerald Ash Borer. *Ecological Modelling, 197(1-2)*, 221-236.

Canadian Food Inspection Agency. (2017). *Emerald Ash Borer - Latest Information*. Retrieved from: https://www.inspection.gc.ca/plants/plant-pests-invasive-species/insects/emerald-ash-borer/latest-information/eng/1337287614593/1337287715022

Cappaert, D., McCullough, D., Poland, T., & Siegert, N. (2005). Emerald Ash Borer in North America: A research and regulatory challenge. *American Entomologist, 51(3)*, 152-165.

Coppin, N., & Richards, I. (Eds.). (1990). *Use of vegetation in civil engineering* (23-36). Butterworths: Ciria.

Costanza, K., Livingston, W., Kashian, D., Slesak, R., Tardif, J., Dech, J., . . . Siegert, N. (2017). The precarious state of a cultural keystone species: Tribal and biological assessments of the role and future of black ash. *Journal of Forestry, 115(5)*, 435-446.

Crook, D., & Mastro, J. (2010). Chemical ecology of the emerald ash borer. *Journal of Chemical Ecology, 36(1)*, 101-112.

Diamond, A., & Emery, K. (2011). Black ash (Fraxinus nigra Marsh.): Local ecological knowledge of site characteristics and morphology associated with basket–grade specimens in New England (USA). *Economic Botany, 65(4)*, 422-426.

Duinker, P., Ordóñez, C., Steenberg, J., Miller, K., Toni, S., & Nitoslawski, S. (2015). Trees in Canadian cities: Indispensable life form for urban sustainability. *Sustainability, 7(6)*, 7379-7396.

Fahrner, S., Lelito, J., & Aukema, B. (2015). Influence of temperature on the flight capacity of emerald ash borer Agrilus planipennis and its parasitoid, Tetrastichus planipennisi: Implications to biological control. *BioControl, 60(4)*, 437-449.

Flower, C., Knight, K., Rebbeck, J., & Gonzalez-Meler, M. (2013). The relationship between the emerald ash borer (Agrilus planipennis) and ash (Fraxinus spp.) tree decline: Using visual canopy condition assessments and leaf isotope measurements to assess pest damage. *Forest Ecology and Management, 303*, 143-147.

Gandhi, K., & Herms, J. (2010). North American arthropods at risk due to widespread Fraxinus mortality caused by the alien Emerald Ash Borer. *Biological Invasions, 12(6),* 1839-1846.

Gonzalez, E., Felipe-Lucia, M., Bourgeois, B., Boz, B., Nilsson, C., Palmer, G., & Sher, A. (2017). Integrative conservation of riparian zones. *Biological Conservation, 211*, 20-29.

Government of Nova Scotia. (2001). Forests Act: Wildlife Habitat and Watercourses Protection Regulations. Revised Statutes of Nova Scotia, c. 179.

Guenther, S., Moore, R., & Gomi, T. (2012). Riparian microclimate and evaporation from a coastal headwater stream, and their response to partial-retention forest harvesting. *Agricultural and Forest Meteorology*, *164*, 1-9.

Gurnell, A., & Petts, G. (2006). Trees as riparian engineers: The Tagliamento River, Italy. *Earth Surface Processes and Landforms, 31(12)*, 1558-1574.

Halifax Regional Municipality. (2019). *Emerald Ash Borer*. Retrieved from https://www.halifax.ca/transportation/streets-sidewalks/urban-forestry/emerald-ash-borer

Halifax Regional Municipality. (2019). *Sir Sandford Fleming Park*. Retrieved from https://www.halifax.ca/recreation/parks-trails-gardens/trails/sir-sandford-fleming-park

Hanberry, B. (2014). Rise of Fraxinus in the United States between 1968 and 2013. *The Journal of the Torrey Botanical Society, 141(3),* 242-249.

Hauer, R., & Peterson, W. (2017). Effects of emerald ash borer on municipal forestry budgets. *Landscape and Urban Planning, 157*, 98-105.

Herms, D., & McCullough, D. (2014). Emerald ash borer invasion of North America: History, biology, ecology, impacts, and management. *Annual Review of Entomology, 59(1),* 13-30.

iTree Canopy. (2011). *Technical Notes*. Retrieved from https://canopy.itreetools.org/resources/iTree_Canopy_Methodology.pdf

Jansson, R., Laudon, H., Johansson, E., & Augspurger, C. (2007). The importance of groundwater discharge for plant species number in riparian zones. *Ecology, 88(1)*, 131-139.

Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. *Canadian Journal of Fisheries and Aquatic Sciences*, *106(1)*, 110-127.

Klooster, W., Gandhi, K., Long, L., Perry, K., Rice, K., & Herms, D. (2018). Ecological impacts of emerald ash borer in forests at the epicenter of the invasion in North America. *Forests, 9(5)*, 250-264.

Kreutzweiser, D., Nisbet, D., Sibley, P., & Scarr, T. (2019). Loss of ash trees in riparian forests from emerald ash borer infestations has implications for aquatic invertebrate leaf-litter consumers. *Canadian Journal of Forest Research, 49(2)*, 134-144.

Larsen. (2015). *Forest Measurement and Inventory*. Retrieved from http://oak.snr.missouri.edu/for2542/Handouts/sampling.pdf

Li, Y., Chen, Z., Lou, H., Wang, D., Deng, H., & Wang, C. (2014). Denitrification controls in urban riparian soils: Implications for reducing urban nonpoint source nitrogen pollution. *Environmental Science and Pollution Research, 21(17)*, 10174-10185.

Liang, L., & Fei, S. (2014). Divergence of the potential invasion range of emerald ash borer and its host distribution in North America under climate change. *Climatic Change, 122(4)*, 735-746.

Macpherson, M., Kleczkowski, A., Healey, J., Quine, C., & Hanley, N. (2017). The effects of invasive pests and pathogens on strategies for forest diversification. *Ecological modelling*, *350*, 87-99.

Mathewson, B. (2014). The relative abundance of the juvenile phase of the eastern red-spotted newt at Harvard Forest prior to the arrival of the hemlock woolly adelgid. *Southeastern Naturalist, 13*, 117-129.

McKenney, D., Pedlar, J., Yemshanov, D., Lyons, B., Cambell, K., & Lawrence, K. (2012). Estimates of the potential cost of emerald ash borer (Agrilus planipennis Fairmaire) in Canadian municipalities. *Arboriculture and Urban Forestry*, *38(3)*, 81-91.

McPherson, E. G., van Doorn, N. S., & Peper, P. J. (2016). Urban tree database and allometric equations. *Gen. Tech. Rep. PSW-GTR-253. Albany, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station. 86*, 253.

Moore, R., Spittlehouse, D., & Story, A. (2005). Riparian microclimate and stream temperature response to forest harvesting: a review. *JAWRA Journal of the American Water Resources Association, 41(4)*, 813-834.

Morin, R. S. (2010). The distribution of ash in North America. In*: Michler, Charles H.; Ginzel, Matthew D., eds. 2010. Proceedings of symposium on ash in North America; 2010 March 9-11; West Lafayette, IN. Gen. Tech. Rep. NRS-P-72. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station: 16 (72)*, 16-16.

Mosseler, A., Lynds, J., & Major, J. (2003). Old-growth forests of the Acadian Forest Region. *Environmental Reviews, 11*, S47-S77.

Muscolo, A., Bagnato, S., Sidari, M., & Mercurio, R. (2014). A review of the roles of forest canopy gaps. *Journal of Forestry Research, 25(4)*, 725-736.

National Research Council Committee on Riparian Zone Functioning Strategies for Management. (2002). *Riparian areas : Functions and stragegies for management*. Washington, D.C.: National Academy Press.

Nisbet, D. (2014). Indirect ecological impacts of emerald ash borer and associated ash decline in southern Ontario riparian areas: Risks to aquatic communities (Masters dissertation). University of Guelph, Guelph, ON.

Nisbet, D., Kreutzweiser, D., Sibley, P., & Scarr, T. (2015). Ecological risks posed by emerald ash borer to riparian forest habitats: A review and problem formulation with management implications. *Forest Ecology and Management, 358*, 165-173.

Orlova-Bienkowskaja, M., & Volkovitsh, J. (2018). Are native ranges of the most destructive invasive pests well known? A case study of the native range of the emerald ash borer, Agrilus planipennis (Coleoptera: Buprestidae). *Biological Invasions, 20(5)*, 1275-1286.

Poland, T., Chen, Y., Koch, J., & Pureswaran, D. (2015). Review of the emerald ash borer (Coleoptera: Buprestidae), life history, mating behaviours, host plant selection, and host resistance. *Canadian Entomologist, 147(3)*, 252-262.

Prasad, A., Iverson, M., Peters, L., Bossenbroek, R., Matthews, M., Davis-Sydnor, P., & Schwartz, J. (2010). Modeling the invasive emerald ash borer risk of spread using a spatially explicit cellular model. *Landscape Ecology, 25(3),* 353-369.

Pureswaran, Deepa S., & Poland, Therese M. (2009). Host Selection and Feeding Preference of Agrilus planipennis (Coleoptera: Buprestidae) on Ash (Fraxinus spp.). *Environmental Entomology, 38*, 757-765.

Rankin, W., & Pickett, S. (1989). Time of establishment of red maple (Acer rubrum) in early oldfield succession. *The Bulletin of the Torrey Botanical Club, 116(2)*, 182-186.

Reisinger, A., Groffman, P., & Rosi-Marshall, E. (2016). Nitrogen cycling process rates across urban ecosystems. *Microbiology Ecology 92(12)*. 1-11.

Robertson, W., Robinett, M., & McCullough, M. (2018). Soil moisture response to white ash mortality following emerald ash borer invasion. *Environmental Earth Sciences, 77(9)*, 1-14.

Roy, A., Freeman, M., Freeman, B., Wenger, S., Ensign, W., & Meyer, J. (2006). Importance of riparian forests in urban catchments contingent on sediment and hydrologic regimes. *Environmental Management, 37(4)*, 523-539.

Ryall, K. (2013). Detecting and monitoring emerald ash borer populations. *Natural Resources Canada, Canadian Forest Service-Great Lakes Forestry Centre, Sault Ste. Marie, ON. Frontline Express*, *67*.

Sackville Rivers Association. (n.d.). *Sackville River Watershed*. Retrieved from http://sackvillerivers.ns.ca/sackville-river-watershed/

Siegert, N., McCullough, D., Williams, D., Fraser, I., Poland, T., & Pierce, S. (2010). Dispersal of Agrilus planipennis (Coleoptera: Buprestidae) from discrete epicenters in two outlier sites. *Environmental Entomology, 39*, 253-265.

Simon, A., & Collison, A., (2002). Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. *Earth Surface Processes and Landforms, 27*, 527-546.

Solins, J., Thorne, J., & Cadenasso, M. (2018). Riparian canopy expansion in an urban landscape: Multiple drivers of vegetation change along headwater streams near Sacramento, California. *Landscape and Urban Planning, 172*, 37-46.

Steenberg, J., Millward, A., Nowak, D., Robinson, P., & Ellis, A. (2017). Forecasting urban forest ecosystem structure, function, and vulnerability. *Environmental Management, 59(3)*, 373-392.

Tanis, S., & McCullough, D. (2012). Differential persistence of blue ash and white ash following emerald ash borer invasion. *Canadian Journal of Forest Research, 42(8)*, 1542-1550.

Tluczek, A., McCullough, D., & Poland, T. (2011). Influence of host stress on emerald ash borer adult density, development, and distribution in *Fraxinus pennsylvanica* trees. *Environmental Entomology, 40(2)*, 357-366.

Wang, X., Yang, Z., Gould, J., Zhang, Y., Liu, G., & Liu, E.. (2010). The biology and ecology of the emerald ash borer, *Agrilus planipennis*, in China. *Journal of Insect Science, 10(128)*, 128.

Ward, D. (2004). 14 Threats to the Urban Forest. *Alternatives Journal, 30(3)*, 22.

Weier J., & Herring, D. (2000). *Measuring Vegetation*. Retrieved from https://earthobservatory.nasa.gov/features/MeasuringVegetation/measuring_vegetation_1.php

Westfall & Morin. (2012). *Canopy Cover Estimates for Individual Tree Attributes*. Retrieved from https://www.nrs.fs.fed.us/pubs/gtr/gtr-nrs-p-105papers/39westfall-p-105.pdf