Assessing the Vulnerability of the HRM Urban and Rural Canopy to the Potential Arrival of the Emerald Ash Borer

Environmental Science Honours Thesis

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Abstract

Invasive species are an ever-increasing problem in urban and rural forests, and have the potential to severely decimate tree populations. The Emerald Ash Borer (EAB) is one such invasive species, which targets ash trees. Ash trees are ecologically, socially, and economically valuable trees. The EAB is predicted to move into the Halifax Regional Municipality (HRM), and this of concern to many forest managers. In this study, I assessed the vulnerability of ash trees in the urban and rural HRM to the potential arrival of the EAB. This assessment was conducted using spatial data compiled from different studies and sources. The locations of ash trees were extracted from these datasets to get a spatial distribution of the ash trees in the HRM. An assessment of the distribution was then conducted using the tree metrics provided in the data (e.g. diameter at breast height (DBH), stem count/plot), the spatial distribution of ash trees, and the existing literature on EAB dispersal. The objective of the study was to show how the urban and rural forest will be affected by the EAB. It was found that 2.54% of the trees found in the urban forest were ash trees, and that in the rural forest, 0.02% of the trees in the FRI data, as well 0.14% of trees found in the UFORE and PSP data, were ash. The urban forest may be more susceptible to the EAB given the shorter distances between ash trees, compared to the longer distances between ash trees and groves in the rural forest. However, given the overall low density of ash trees found in the datasets, the movement of EAB into the HRM may not affect the overall canopy cover to a high degree. The spatial distribution will provide forest managers with a clear depiction of which areas may be affected the most. It will also aid in deciding which mitigation practices can be put in place to contain the spread of EAB should it arrive into the HRM.

Chapter 1

Invasive species are an ever-increasing problem in urban forests. They have the potential to severely deplete or even eradicate tree species. The urban forest is a key component of cities and towns and provides many benefits in the form of ecosystem services. These ecosystem services include shading, aesthetic value, flood mitigation, toxin uptake, carbon storage, among others (Duinker et al. 2015). Therefore, the care and management of urban forests is critical for the development of a more sustainable future. This is especially true with the ongoing problem of climate change. The rising temperatures will increase the movement of invasive species and shape the urban forest (Foran et al. 2015, Liang & Fei 2013).

The EAB is an invasive insect from Asia and was first spotted in Michigan in 2002 (Poland & McCullough 2006). Since then it has spread widely and caused extensive damage to the urban forests throughout the US and into Quebec (Natural Resources Canada 2016). The EAB has spread due to several anthropogenic and biological factors. These factors include rapid movement through the transportation of firewood (Herms & McCullough 2014), and spreading due to the lack of predators (Liang & Fei 2013). The EAB is predicted to spread to approximately 80% of the ash population range in North America (Liang & Fei 2013). For these reasons, various experts believe it is of grave concern to the urban and rural forest of the HRM.

Ash trees are a valuable species for a variety of reasons, and the EAB will cause extensive damage on all species in the genus. The EAB bores into the ash trees at the larval stage, chewing through the phloem and cambium, subsequently girdling the tree (Poland & McCullough 2006). The loss of ash trees will cause massive impacts. Canada is expected to lose $0.5 to $1.5 billion due to ash-tree mortality (McKenney et al. 2012). Ash trees are highly significant in riparian buffer areas (Nisbet et al. 2015), and their loss will cause a variety of

ecological impacts (Nisbet 2014, Ulyshen et al. 2011). Black ash is also a highly valuable species to indigenous communities, and thus has significant social importance (Hurlburt 2011, Costanza et al. 2017). In sum, ash trees are a significant species that should be conserved.

One approach to predicting the effects of invasive species on forests is risk assessment. Risk and susceptibility assessments aid in the development of management plans so that appropriate mitigation responses can be put in place (Foran et al. 2015, Fuenteabla et al. 2013). One such study by Foran et al. (2015) assessed the susceptibility of the urban forest in Cambridge, MA, to the EAB. Another study looking at the vulnerability of urban forests to the EAB found that the vulnerability does not decrease with higher species diversity (Berland & Elliot, 2012). Many other studies focusing on the susceptibility and vulnerability of ash trees in North America to the EAB at a spatial level have also been conducted (Liang & Fei 2013, Crocker 2006, Ayersman 2010). However, no such assessment study has been conducted in Halifax, NS. Therefore, there is a need to know how the EAB may spread in Halifax's urban and rural forest.

The goal of this study is to assess the vulnerability of HRM's urban and rural canopy cover to the potential arrival of the EAB. This will be accomplished partly by generating a spatial distribution of ash trees across the entire HRM. This study will aid urban forest managers in the HRM to mitigate and prepare for the potential arrival of the EAB. An image of how the urban and rural forest will be affected can be obtained through a vulnerability assessment.

This study will determine how the HRM's urban and rural canopy may be affected by the potential arrival of the EAB. This research question is further developed through the following points:

1. A spatial distribution of the ash trees in the urban and rural HRM will be determined.
2. A qualitative vulnerability assessment will be conducted using the spatial distribution of ash trees, the spatial analysis, and existing literature on the effect of EAB on ash trees.

The research question will be addressed through a spatial distribution of ash trees in the HRM, coupled with a qualitative vulnerability assessment of the predicted impact. Relevant data will be assembled from a variety of sources held by the Dalhousie University School for Resource and Environmental Studies (SRES), the Canadian Food Inspection Agency (CFIA), and the Department of Natural Resources (DNR). The boundaries of the study are confined to the HRM. The urban boundaries are defined by the Urban Forest Master Plan neighbourhood boundaries (Halifax Urban Forest Planning Team 2013). The rural boundaries are all other areas in the HRM. Through an analysis of the effects of the EAB and the spatial distribution of ash trees, proper management plans can be put into place to facilitate the mitigation of impacts on the urban and rural forest of the HRM.

Chapter 2

This literature review will summarize the current state of knowledge about the emerald ash borer (EAB). The EAB is an invasive species from China that targets ash trees and has spread throughout various regions in North America. This literature review will cover four main topics: the ecology of the EAB, the spread and dispersal patterns of the EAB in North America, the ecology and treatment of ash trees, and lastly, risk and vulnerability assessments on the EAB and similar studies. The literature review will provide a contextual basis and framework for the vulnerability assessment.

2.1 Emerald Ash Borer Ecology

The EAB is native to various regions in Asia where it mostly affects dying or aging ash trees. The EAB is native to regions in Asia, specifically Northeastern China, Korea, Japan, Mongolia, Taiwan, and eastern Russia (Poland & McCullough 2006). In its native habitat, EAB will mostly affect dying or aging Asian ash trees (Herms & McCullough 2014). However, it has also been found that the EAB will significantly damage younger trees, especially those that have access to more sunlight (Wang et al. 2010). This could be due to the higher temperatures from more sun exposure which would lead to faster development of EAB (Wang et al. 2010). The EAB has been found to start emerging from ash trees in early May (Wang et al. 2010). They have several predators that, to an extent, control EAB populations in Asia. These predators include woodpeckers, parasitoids, and pathogens (Wang et al. 2010). Similar to the spread of EAB in North America, the spread of EAB in Asia is higher due to anthropogenic factors. The EAB does not decimate Asian trees in their native habitat, in comparison to its decimation of ash trees in North America.

The EAB negatively affects ash trees in all its life stages. EABs have a 1- to 2-year life cycle in which they overwinter as prepupae and emerge in the spring as beetles (Poland &. McCullough 2006, Poland et al. 2015). Eggs are laid underneath and between cracks in bark, and hatch within 2 weeks (Herms & McCullough, 2014). The most damage to ash trees occurs when the EAB is at the larval stage (Herms & McCullough 2014). The larvae eat the phloem and cambium of the tree, consequently girdling it (Poland & McCullough, 2006). This stage lasts from mid-summer into fall after which larvae go into their prepupal stage where they overwinter inside the bark (Herms & McCullough, 2014). Some EAB can overwinter for two years, but most emerge around the middle to late spring (Herms & McCullough, 2014). EAB emerge at slightly different times based on temperature and location (Duarte 2013). While mature EAB also eat the leaves of the ash tree, they do not cause as much damage as when in their larval stage (Herms & McCullough 2014, Wang et al. 2010). EAB infestations are hard to detect because ash trees, unlike other trees, do not secrete sap when infested (Wang et al. 2010). Therefore, the only signs of infestation that appear are the D-shaped exit holes that the fully formed beetles emerge from. At that time the damage is already done to the tree (Wang et al. 2010). In sum, the EAB causes the mortality of ash trees through the different stages of its lifecycle.

2.2 Spread and Dispersal of EAB

The EAB has proliferated through North America so rapidly due to a variety of reasons, mostly anthropogenic. The most-agreed-upon reason for how EAB came into North America is from wooden packing materials, such as crates and pallets (Herms & McCullough 2014). However, the transportation of firewood and nursery stock has been the main contributor to EAB spread throughout North America (Poland & McCullough 2006). Another factor that led to the

quick spread, especially through urban forests, was the high ash tree density (Herms & McCullough 2014). This high density meant low species diversity and allowed for EAB to proliferate quickly (Herms & McCullough 2014). Movement of firewood, urban stress, and lack of diversity have most likely been reasons for the rapid spread of EAB throughout North America.

The EAB has spread into the mid-portion of the US as well as Ontario and Quebec and is predicted to spread throughout the mid-Atlantic region of the US and throughout the eastern regions in Canada as well. The EAB was first spotted in Michigan in 2002, but has since spread throughout parts of the US and Canada (Poland & McCullough 2006). It is continuously spreading and as of December 7th, 2017, its spread has ranged from Winnipeg, Manitoba to Sherbrooke, Quebec (Canadian Food Inspection Agency 2017). This also holds true in the US where states in the mid-Atlantic region have a high risk of being infested (Ayersman 2010).

Several studies have been conducted on the dispersal patterns of the EAB to understand how the EAB will spread into new locations (Sargent et al. 2010, BenDor et al. 2006, Siegert et al. 2015, Siegert et al. 2010). The EAB has the potential to spread across the entire region of ash trees in North America (Herms & McCullough 2014). Subsequently, the EAB is predicted to spread to Nova Scotia (Urban Forest Master Plan Team 2013). The EAB has spread in various parts of the US and Canada, and is predicted to keep spreading throughout the natural range of Ash trees in North America.

Climate change will affect the spread of EAB by increasing its spread and exacerbating the effect of the EAB on ash trees. The impact of invasive species on urban forests will be increased by climate change through extreme weather patterns which will cause additional stress on trees and increase the range of invasive species (Foran et al. 2015). EAB may be slowed in

the northern ranges of ash trees due to the colder temperatures (Desantis et al. 2013). However, with rising temperatures due to climate, this may not occur. Therefore, with rising CO_2 levels, there is heightened need to prepare for the expansion and movement of invasive species into new regions (Liang & Fei 2013), especially with the additional stress that climate change may cause on ash trees. For example, rising temperatures as well as higher precipitation may have a negative effect on the radial growth of ash trees (Finley et al. 2016). This may increase the infestation of the EAB, since EAB is known to infect highly stressed trees more than others (Tluczek et al. 2011). Climate change may seriously increase the effects of EAB dispersal as well as impact on ash trees.

2.3 Ash Trees and Treatment

Different ash tree species vary in their resistance to the EAB, as well other factors such as stress that can affect the tolerance of ash to the EAB. Ash tree species native to Asia are more resistant to the EAB than ones in North America (Rebek et al. 2008). Within North American ash tree species, there is evidence that some may be more resistant to EAB than others (Tannis & McCullough 2012, Poland et al. 2015). Factors such as stress also affect EAB infestation by increasing larval development and density (Tluczek et al. 2011). The condition and stress of the canopy of ash as a whole can be used to assess EAB infestation, with canopies in poorer health having higher rates of infestation (Flower et al. 2013). North American ash trees appear to be less resistant to the EAB than Asian ash tree species; however, within North American ash trees, some trees are more resistant than others as well.

Ash trees are a particularly important tree species to conserve due to their economic, ecological, and social benefits. When the EAB infests a region, relatively rapid wide-scale

mortality occurs; however, total survival rate is yet to be concluded (Morin et al. 2017). Therefore, effort is needed in mitigating the effects of the EAB. The loss of ash trees causes significant economic loss (McKenny et al. 2012). Their loss is also predicted to harm riparian forest habitats (Nisbet et al. 2015), as well as cause numerous indirect and direct ecological impacts (Nisbet 2014, Ulyshen et al. 2011). Furthermore, black ash is a particularly significant species due to its unique properties and cultural significance to indigenous communities (Hurlburt 2011, Costanza et al. 2017). It is critical to prevent the loss of ash trees caused by the EAB due to the trees numerous beneficial values.

2.4 Risk Assessment

Many studies have conducted risk assessments, as well as various other data analysis methods to study the spread of invasive species. Assessing vulnerability is part of a larger risk assessment; it specifically considers the probability that the invasive species will harm the forest (Fuentealba et al. 2013). Moreover, vulnerability assessments on the impact of climate change on forests can be used as a framework for assessing the vulnerability of forests to invasive species (Seidl et al. 2009, Johnston & Williamson 2007, Ordóñez & Duinker 2015, Glick et al. 2011). Risk assessments on different invasive species have also been conducted and found useful. One such study was on the mountain pine beetle that looked at dispersal patterns and the relationships of spread between climate suitability, ecology, and forest stands (Nealis & Cooke 2014). Other data analysis methods have also been conducted to study the spread of invasive species. Early detection methods of the EAB infestation using hyperspectral imagery are being developed to aid in EAB mitigation as quickly as possible (Pontiuse et al. 2008). Risk assessments and other data analysis methods aid in the mitigation of the spread of invasive species to new locations.

A variety of studies have been conducted on the vulnerability of the urban forest in different areas to the EAB. Assessments show that the infestation of the EAB has serious effects on urban forests (Foran et al. 2015). Studies that look at the spread of the EAB provide useful information for further assessments. For example, it has been found that the highest environmental predictor for the spread of EAB is distance to nearest infestation (Huset 2013). In forests, high species diversity may correlate with lower EAB vulnerability (Berland & Elliot 2013, Greene & Millward 2016). Furthermore, high concentrations of ash trees, as well as ash with larger diameters and subsequently a larger amount of phloem in riparian forests, increase the production of EAB within those regions (Crocker 2006). The spread of EAB may be able to be predicted and measured, which will aid in providing a theoretical framework for the vulnerability assessment that will be conducted in this study.

There are many different types of treatment for EAB, and work is continuously being conducted to find the most effective methods. There is much research going into the trapping of EAB and treatment of ash trees (Natural Resources Canada 2016). Other treatment methods can also be implemented to mitigate the effects of the EAB which consist of managing the forest as a whole. For example, higher species diversity and improved conditions of the urban forest are expected to lessen the effect of an EAB infestation (Greene & Millward 2016). However, new EAB trapping methods still need to be produced (Damon & Victor 2010), especially due to issues that may arise with current treatment methods. One such example is the felling of ash trees near EAB infested ash. It was found that uncut and undisturbed areas with ash trees that were infested with EAB had fewer invasive species than areas that were cut (Hausman 2010). There are many treatment types for the EAB, both for monitoring the steps of EAB as well as methods of tracking the EAB into its next location.

2.5 Conclusion

The EAB is of serious concern to the urban and rural forests of the HRM due to the extensive damage it may cause to its ash tree population. While the EAB affects ash trees through every stage of its life, the most severe impact occurs during its larval stage, as it eats through the phloem and cambium of the tree. In terms of the ash species native to North America, these species are at a higher risk of mortality from the EAB. Furthermore, the stress of growing in urban areas also increases the impact that the EAB has on these ash trees.

The range of the EAB in North America is extensive and predicted to spread throughout the range of ash trees. This is detrimental since ash trees are important and their loss may have significant negative ecological, social, and economic impacts. Therefore, there is a need for risk and vulnerability assessments due to the usefulness they can provide through planning for the distribution and effects of the EAB should it come to the HRM.

Chapter 3

3.1 Overview of Methods

The project assessed the vulnerability of ash trees in the urban and rural forest of the HRM to the potential arrival of the EAB. This assessment was conducted using spatial data compiled from disparate studies and sources. The locations of ash trees were extracted from these datasets to develop a spatial distribution of the ash trees in the HRM. The spatial pattern of the ash trees was then examined to assess the spread potential of the EAB should it come to Halifax. Lastly, a qualitative assessment of the distribution of ash trees in Halifax was made using the tree metrics provided in the data (e.g. DBH, height, condition), the spatial pattern of ash trees, and the existing literature on EAB dispersal.

3.2 Study Area

The HRM was delineated into two sections: the rural and the urban forest (Figure 1, Figure 2). The boundaries of the urban forest were set according to the boundaries used in the Urban Forest Master Plan (UFMP). An advantage of using these boundaries was that it gives clear descriptors of the boundaries of the urban forest. The remaining area was set as the rural forest because the HRM has one major urban centre. The remaining areas mostly consist of small towns and villages and thus can be distinguished as rural.

3.3 Data Aggregation

No primary data collection was conducted in the study. Data were assembled from a variety of sources. The following were the data sets used in the study.

Rural datasets

Permanent Sample Plot (PSP) data: These data were collected by the Nova Scotia Department of Natural Resources (NS DNR). The plots are monitored every five years for the purpose of tracking forest health and biodiversity (Nova Scotia Forestry Division, 2006). Metrics of the individual trees within the plots are recorded and compiled in a spreadsheet. Foster and Duinker (2017) recently analyzed these data.

NS DNR Forest Inventory Data: The forest inventory data comprise a collection of data from air-photo interpretation that characterizes the forests of Nova Scotia at the stand level (Nova Scotia Forestry Division, 2016). The FRI data measure species proportion, height, landclass, and various other metrics per stand group.

Canadian Food Inspection Agency (CFIA) trap data: These data consist of the location and activity of EAB traps set by the CFIA on ash trees in various locations in Nova Scotia.

Urban datasets

2007 UFORE data: These data were the predecessors of the the 2016 I-tree data. The purpose of the data was to make calculations of specific ecosystem services using the UFORE model of the USDA Forest Service. The model uses the data to provide information on the urban forest structure and function. It has a larger study area than the UFMP area, so the data outside of the UFMP boundary were included in the rural data.

2016 I-tree data: These data were collected to assess the ecosystem services of the urban forest using UFORE's successor i-Tree Eco. The dataset consists of 200 tree plots that were established through stratified random sampling (20 plots in each stratum). Metrics such as DBH, species,

and tree height were measured for each tree within the plot (Foster & Duinker, 2017). The plot locations are mapped, and specific tree information is included in a separate excel sheet.

UFMP planting data: These data consist of measurements on trees planted under the Urban Forest Master Plan. They also include a variety of metrics of the individual trees shortly after planting.

Tree spacing study, Aryal study data: These data were collected by master's student Bimal Aryal at the School for Resource and Environmental Studies. The data are spatially resolved only to street segments and consist of the counts and species of the trees in those segments. Various metrics such as DBH are recorded as well.

Managing tree diversity, Nitoslawski study data: These data were collected from four suburban neighbourhoods in the HRM (Nitoslawski & Duinker, 2016). Three types of trees were measured: street trees, forest remnant plot trees, and private residential property trees. Within those different types, species and metrics such as DBH were recorded.

Ashburn golf course data: This is a small dataset that consists of a tree inventory of the Ashburn Golf Course conducted by former Dalhousie MREM student, Shauna Doll.

Waterfront data: This dataset consists of a tree inventory of a section of the Halifax waterfront (Foster et al. 2014). The purpose was to gain a a better understanding of the conditions of trees so as to improve the urban forest along the Halifax waterfront. The metrics recorded include species, DBH and canopy condition.

3.4 Data reliability and validity

A validity issue presents itself in relation to variable precision of the different datasets. For example, some datasets consist of plot data, while others are collected at the individual tree

level. This may cause problems with the coherence of the distribution of ash trees. This will then affect how accurate the assessment will be. There is no way to mitigate this issue because it is not possible to change the precision of the data. However, being consistent with the analysis across all datasets may manage the issue. Issues with accuracy were unlikely to appear given that each dataset is accompanied by information on how the data were collected and who collected them.

The main concern with the reliability of this project was how well the data represents the true amount of ash trees in the HRM. Therefore the true effect of the EAB should it arrive to the HRM is difficult to quantify. Though there is little that can be done to mitigate this issue, the intensity of sampling in each region was factored into the study so as to provide a reference point to how much sampling occurred. This provides a foundation to understand the total number of ash trees found in the data.

3.5 Data Analysis

3.5.1 ArcGIS Analysis

The location of ash trees from the various datasets was extracted using as consistent a set of methods as possible. However, due to the variation of the datasets, some divergences were inevitable. The appropriate species identification from the provided spreadsheet was joined to the particular shapefile (if it is not already included in the shapefile itself). The locations of the ash trees were then selected using the select-by-attributes feature. Those locations were then exported into a new shapefile.

The urban forest delineated by the HRM UFMP area was further split into the 10 communities within the UFMP area. The datasets were aggregated per community. Intensity of ash trees within the datasets were not included in the spatial distribution to reduce clutter within

the map. In turn, the map was accompanaied by a corresponding table with the amount of ash trees found in each datasat per community. Basal area was also calculated for the i-Tree data using the DBH of the trees found. Furthermore, a table with the total count of trees found in the urban forest was created for the purposes of assessing the urban forest as a whole

The rural forest was aggregated into rates. Each dataset was rated based on its natural breaks, whether it was low, medium or high in its quantity of ash trees. The PSP plots and UFORE plots were rated based on basal area sum. The CFIA plots were placed in areas with ash trees present, and therefore were given a standard rate of medium. The FRI data were rated based on proportion of ash in each polygon, and those polygons were then converted to points for the purposes of this display. There were two reasons to display the rural data in this manner. Firstly, this manner displays the various datasets in a cohesive manner, allowing readers and users of this data to easily detect where ash is most present. Secondly, this manner allows more location-sensitive data to be displayed without the specific location being identified. Lastly, two tables were created that display the total amount of ash found in the FRI, as well as UFORE and PSP data, respectively.

3.5.2 Assessment portion

The vulnerability assessment did not involve any data analysis but rather is a qualitative assessment that pinpoints areas of concern in comparison to others. The vulnerability was assessed in two main steps. The first looked at the amount of ash found in the urban and rural forests. This incorporated both the amount of ash found as well as the intensity of sampling within each region. The second portion consisted of taking account of the existing literature and the spatial distribution of ash within the HRM to assess how the ash may cope with the incoming

of the EAB. This incorporated the biology of EAB dispersal, as well as the various anthropogenice factors of EAB movement.

3.6 Limitations

There is no complete tree inventory of the urban and rural forest of HRM, so to assess the vulnerability of HRM's ash trees to EAB, a diverse range of datasets had to be used. Ash that is unaccounted for in the datasets will, of course, be missed in the spatial distribution. Therefore, the assessment may have underestimated the extent of the effect that may occur.

Another limitation was the inability to fully quantify the uncertainty of the assessment. Therefore, the extent to which this assessment is accurate is difficult to determine. Another problem that arose was that since the assessment was not quantitative, different interpretations can lead to alternative conclusions from the analysis portion. However, with a spatial distribution of the ash, the data are readily available and can be re-used to improve the vulnerability assessment over time as more information arises.

Chapter 4

4.1 Rural Forest

The following is an account of the distribution ash trees found across rural and urban HRM. The rural data will be described first, to give an account of the data for HRM as a whole, before going into a finer examination of the urban forest.

As Figure 1 shows, ash seem to be distributed unevenly across rural HRM, with relative concentrations furthest from the ocean. In Figure 1 it can be seen that ash is centralized in clusters along the northwest region of the HRM, as well as along the borders of the urban forest.

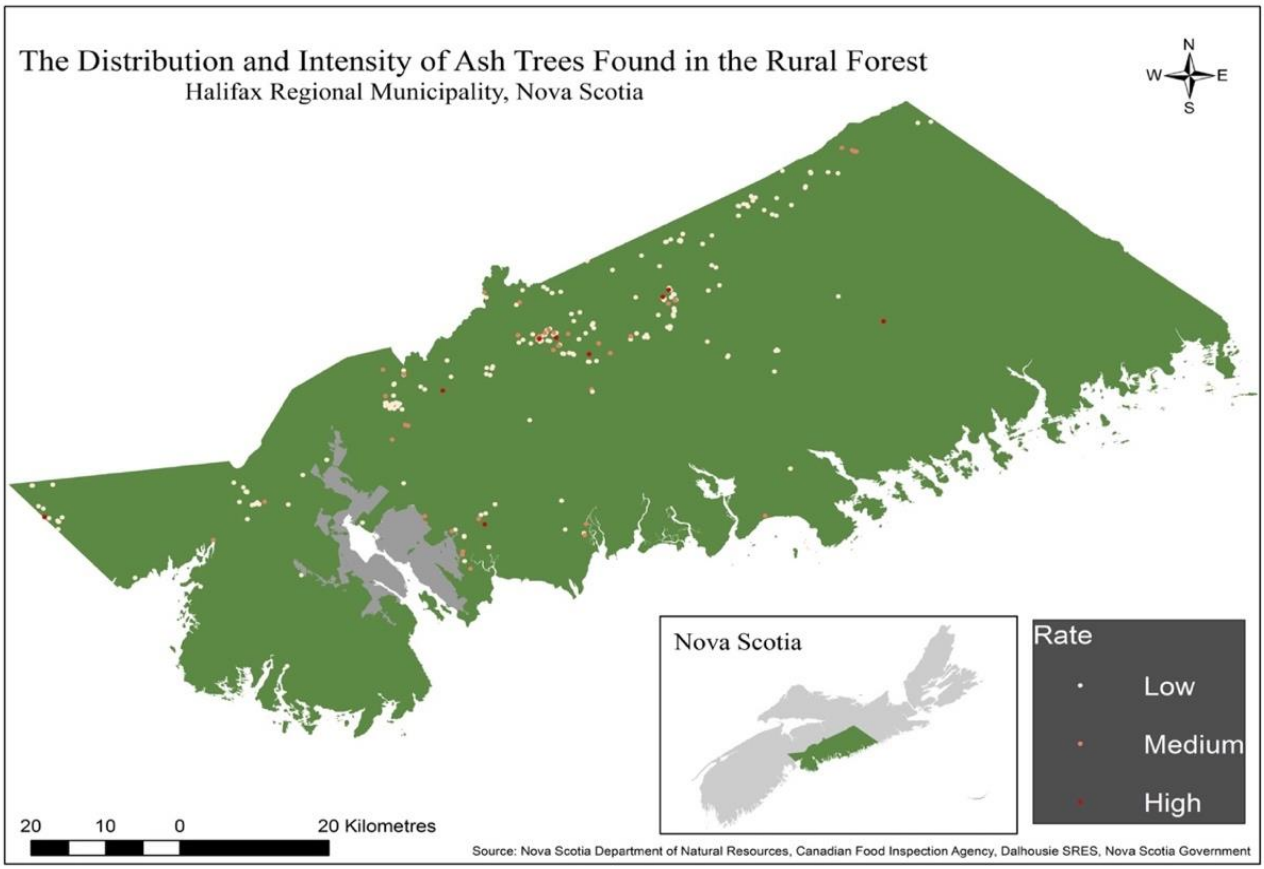

Figure 1 The distribution and intensity of ash trees found in the rural forest of the Halifax Regional Municipality. Each rural data set - CFIA traps, FRI data, UFORE data, and PSP data - is represented as points, with a rating system for the intensity of ash trees found within the point.

There are almost no ash trees on the coastline of the HRM. The data indicate that merely 0.14% of the trees sampled were ash trees (Table 1). The FRI data indicate that, using the effective-hectares approach, ash was found in 0.02% of the total area of the FRI (Table 2).

Table 1 Compositional proportions of ash present in the sampled datasets in the rural forest of the Halifax Regional Municipality. CFIA traps and FRI data could not be included in this analysis.

Rural Data	Total Trees	Total Ash Trees	Percent Ash
PSP	34837	37	0.11
UFORE (rural)	5505	19	0.35
Total	40342	56	0.14

Table 2 Effective hectares of ash in the FRI data. Effective hectares are calculated as proportion-weighted areas of ash trees in each FRI polygon. Percent ash refers to total number of effective hectares as a proportion of the total hectares in the FRI data regardless of species.

Proportion in FRI	Hectares	Effective Hectares
0	736902.8	0.0
0.1	563.6	56.4
0.2	258.8	51.8
0.3	107.6	32.3
0.4	15.2	6.1
0.5	1.3	0.6
0.6	7.5	4.5
0.7	2.7	1.9
0.8	1.1	0.9
0.9	1.4	1.3
Total ha	737861.9	155.7
Percent Ash (%)	N/A	0.02

4.2 Urban Forest

As evident in Table 3, the only urban-forest dataset that provided consistent coverage of the entire urban core (i.e., UFMP study area) was the i-Tree study. It included 20 plots in each of the UFMP's ten communities (Figure 2). In the i-Tree data, no ash were found in the far west (FBLT) and far east (Cole Harbour, Eastern Passage) communities. Ash populations are clearly more prevalent in the central and northern communities, especially in Beaverbank and Spryfield.

Table 3 The data and subsequent metrics on ash trees, as well as total trees measured, of each community in the urban forest of the Halifax Regional Municipality

Data	Community	Dartmouth	Halifax Peninsula	Beaver Bank	Bedford	Spryfield	Sackville	Ashburn/ Armdale	Cole Harbour	Eastern Passage	FBLT
UFORE	Number of plots with ash	2	0	0	0	0	0	0	0	0	0
UFORE	Total number of plots	24	7	2	6	4	12	1	4	11	6
i-Tree	Number of plots with ash	1	1	3	3	3	3	1	0	0	0
i-Tree	Total number of plots	20	20	20	20	20	20	20	20	20	20
i-Tree	Average Density (# trees/ha)	25	75	275	108	25	58	25	0	0	0
i-Tree	Basal Area sum	0.004	0.137	0.431	0.331	0.239	0.178	0.019	0.000	0.000	0.000
UFMP contract	Number of ash	0	20	0	0	0	0	0	10	9	10
UFMP 311	Number of ash planted	3	3	0	3	0	2	0	0	0	0
UFMP	Total number planted	368	2041	0	33	20	178	0	855	846	907
Aryal study	Number of ash	9	46	0	0	0	0	0	0	0	0
Aryal study	Total measured	246	1916	0	0	0	0	0	0	0	0
Nitoslawski study	Number of ash	72	0	0	119	0	0	0	0	0	0
Nitoslawski study	Total measured	1080	0	0	1362	0	0	0	0	0	0
FRI	Percent of community in FRI database	27.6	0	0	0	0	0	0	0	0	0
FRI	Effective hectare	9.85	0	0	0	0	0	0	0	0	0
Waterfront	Number of ash	0	4	0	0	0	0	0	0	0	0
Waterfront	Total number measured	0	463	0	0	0	0	0	0	0	0
Ashburn golf course	Number of ash	0	0	0	0	0	0	~300	0	0	0
Ashburn golf course	Total number measured	0	0	0	0	0	0	~17000	0	0	0

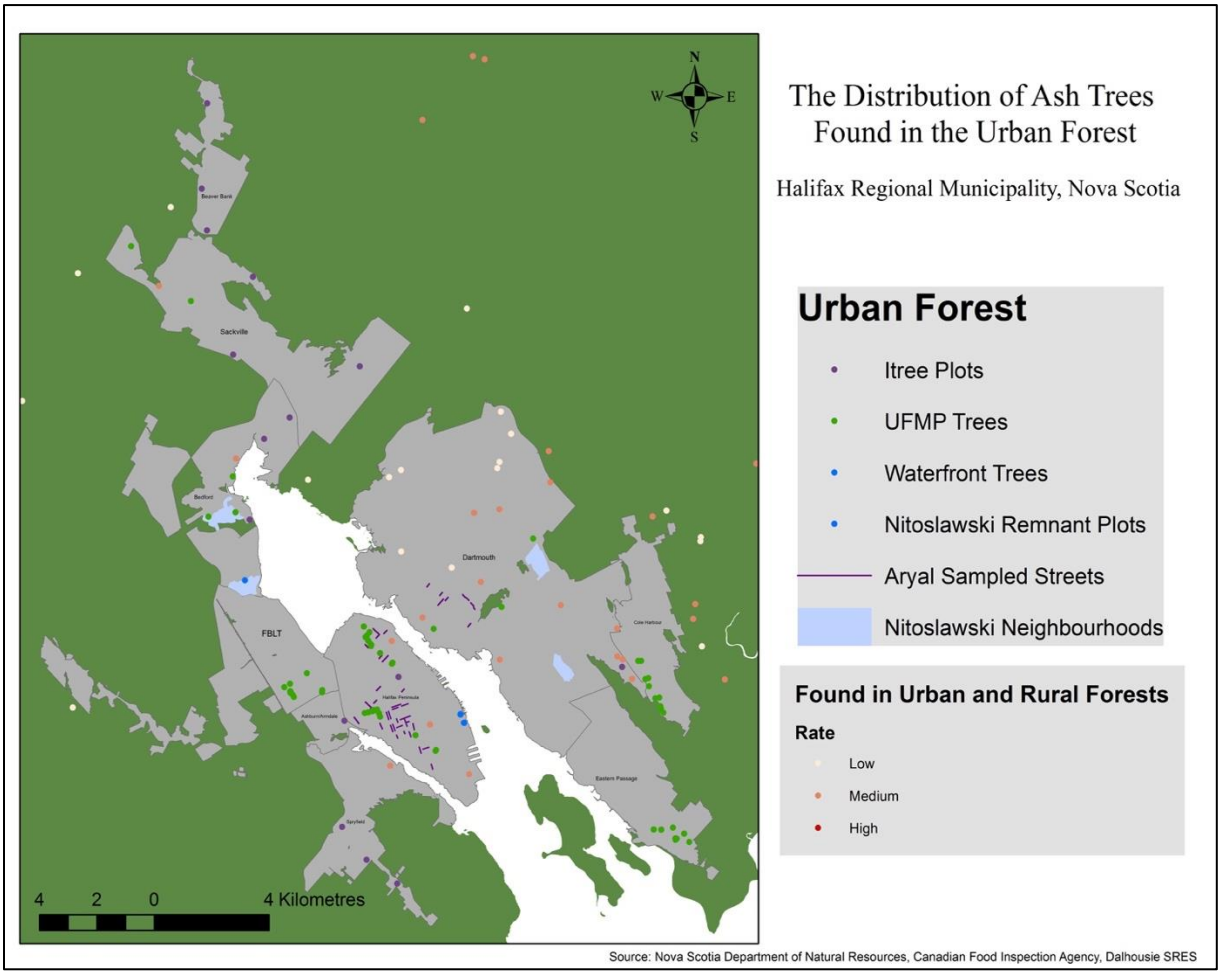

Figure 1 The distribution of ash trees found in the urban forest of the Halifax Regional Municipality. Each urban dataset is included as unique markers. Data overlapping with the rural forest (CFIA traps, UFORE data, FRI data) are based on the rating system used for the rural forest distribution to maintain consistency across figures.

As evident in Figure 2, one can observe how in some communities, such as the Halifax Peninsula and Bedford, there is a roughly even distribution of ash trees, while in some communities, such as Eastern Passage and Cole Harbour, the ash trees are clustered in one location (Figure 2). This latter pattern is explained by the fact that ash trees were part of the planting mix as some streets in these communities received intensive planting attention since UFMP implementation began in 2013.

Dartmouth and the Halifax Peninsula had the highest number of ash trees found as well as the highest sampling intensities (Table 3). Across all UFORE plots in the urban forest, only Dartmouth was found to have ash trees. It also had the highest number of UFORE plots, 24 (Table 3). Dartmouth also contained the highest rate of sampling, with all sampling that occurred having found ash (Table 3). Relative to the rest of the communities, Dartmouth did not have a high basal area or density in the one i-Tree plot found to contain ash trees (Table 3). The Halifax Peninsula had a high sampling intensity, and was found to have a high amount of ash. It only contained one i-Ttree plot with ash; however, that plot was found to have a high basal area sum and density relative to other communities when accounting for the number of plots found with ash (Table 3). The Halifax Peninsula also had the highest number of ash trees planted under the UFMP, 23 (Table 3). In sum, these two communities had a high amount of sampling and high number of ash trees found, relative to the remaining communities.

Beaver Bank had three i-Tree plots with ash, with the highest average density and basal area sum (Table 3). Bedford had the second highest basal area sum, followed by Spryfield and then Sackville (Table 3). A relatively higher sampling rate occurred in Bedford, as two neighbourhoods were measured in the Nitoslawski study (Figure 2); of the 1362 trees measured, 119 trees were ash trees. Two sets of data found ash trees in Ashburn, one i-Tree plot and an inventory of the trees found in the Ashburn golf course (Table 3).

Cole Harbour, Eastern Passage, and Fairview/Beechville-Lakeside-Timberlea (FBLT) all had very similar counts and distributions of ash (Figure 2). All communities contained UFORE and i-Tree plots, but none of these plots contained ash trees (Table 3). The only ash trees found in these communities were trees planted under the UFMP. Furthermore, the number of ash trees planted and total numbers of trees planted were rather similar, with each community having 9-10

ash trees planted, and a total tree count between 855-907 trees (Table 3). In Table 4, it can be seen that the total percentage of ash amongst all trees measured was 2.54%. The study conducted by Nitoslawski et al. had the highest percentage of ash trees found, while the UFORE plots had the lowest percentage of ash trees found (Table 4).

Table 4 The total number of trees measured, total number of ash trees measured in each dataset, and the subsequent percentages of ash present in the sampled datasets in the urban forest of the Halifax Regional Municipality. CFIA traps and FRI data are not included because it is unknown how many ash trees are around the CFIA traps, and the FRI data are in proportions, not counts.

Urban Data	Total Trees	Total Ash Trees	Percent Ash
UFORE (UFMP)	918	6	0.65
Itree	3670	61	1.66
Aryal study	2091	55	2.63
Nitoslawksi study	2442	191	7.82
UFMP	5259	60	1.14
Waterfront	463	4	0.86
Total	14843	377	2.54

Chapter 5

Dartmouth and Halifax contained the highest number of ash trees. This may be due to the higher sampling intensity in the two communities. For example, when comparing the Nitoslawski study, similar percentages were found in Dartmouth, 6.66%, and Bedford, 8.73%. However, Dartmouth is unique in that there are FRI plots containing ash in the community; 27.6% of the community area contains natural stands with 9.85 effective ha of ash (Table 3). Furthermore, when looking at the Aryal study data, 3.66% of the 246 trees measured were ash trees. When comparing solely i-Tree plots, with all other communities, Dartmouth has relatively low ash content in comparison to the remaining communities (Table 3). However, compared to all other communities, it has been the most sampled, and has the highest amount of EAB traps. Overall, this means that in Dartmouth it can be said with lower uncertainty than other communities that the amount of ash found is a good representation of the amount of ash in the canopy of Dartmouth. Seven EAB traps seems to represent a reasonable amount for EAB detection.

Halifax had a high amount of sampling, and had diverse amounts of ash within those samples. Some sets of data contained relatively higher amounts of ash than other communities, such as UFMP and i-Tree, while other studies such as the Aryal study data and Waterfront Development did not. There was one i-Tree plot, and it had a higher basal area sum and density for ash trees than other communities with one plot each. There were also 23 ash trees planted under the UFMP, higher than any of the other communities (Table 3). However, when looking at the Aryal study data, 2.4% of the measured trees were ash, and in the Waterfront Development data, only 0.86 % of the trees were ash. Regardless of the amount, most of the trees sampled in the Halifax core were public street trees, meaning that private property trees and trees in natural

forest stands were not accounted for. This means that public street trees may have been overaccounted for in the study. Therefore, further sampling of private properties and remenant plots may need to occur, to ensure that the distribution of ash trees are understood in these areas.

Beaver Bank is at the northernmost point of the urban forest in the HRM. Considering that the data show a trend where areas with more sampling intensity appear to contain more ash, then Beaver Bank, a poorly sampled community, may have more ash than it appears. Three i-Tree plots were found to contain ash, and those plots combined contained the highest average density of ash trees across the whole city, as well as the highest basal area sum, i.e., 275 stem/ha, and 0.431 m^2/ha (Table 3). The second highest i-Tree plot with ash was in Bedford, with 3 plots that have an average density of 108 stem/ha, and total basal area sum of 0.331 m^2/ha (Table 3).

When comparing Beaver Bank and Bedford, Beaver Bank may only appear to have fewer ash trees because of the lack of sampling, compared to other communities with the same amount of sampling, but with fewer ash found (e.g. are Cole Harbour, Eastern Passage, Fairview/BLT). This can be seen when looking at the Nitoslawski study data in Bedford; one can see that of the 1362 trees measured in public, private, and remnant plots, 119 were ash trees (Table 4), which is 8.73% of trees sampled. Two neighbourhoods were studied, and of the 119 trees measured, only one ash tree was in a remnant plot, and the remaining were on private residential property or along streets. In sum, Bedford and Beaver Bank had similar i-Tree plot data; however when Bedford was sampled further, it was found to have a much higher presence of ash. Beaverbank may have a similar number of ash trees in its neighbourhoods due to its similarity to Bedford.

Sackville and Spryfield are similar to both Beaver Bank and Bedford in the amount of ash present, but to a slightly smaller degree. Both have three i-Tree plots containing ash. Sackville has a higher average density of ash trees in the i-Tree plots, at 58 stem/ha, compared to

25 m^2/ha in Spryfield. However, Spryfield has a higher basal area sum at 0.239, compared to 0.178 m^2/ha (Table 3). Sackville also contains two ash trees planted under the UFMP, and both neighbourhoods have one EAB trap each. Relative to all other communities, Spryfield and Sackville are similar to each other. As a whole, Beaver Bank, Bedford, Sackville and Spryfield had relatively high amounts of ash trees, especially when looking at the i-Tree plot data, but overall they were sampled less than some other communities.

Sackville, Spryfield, Beaver Bank, and Bedford potentially need more EAB traps, and subsequently more monitoring. Halifax and Dartmouth are dense urban cores and are more easily monitored than suburban areas. However, EAB may live undetected in suburban areas before spreading to a point where the effects are visible. This is particularly pertinent in Beaver Bank, which contains no EAB traps. Along with the high possibility of finding more ash in Beaver Bank, when looking just outside the community, a rural plot contains ash (Figure 2). This is important to note because, when looking at the ash distribution, Beaver Bank is a busy access point from the rural forest to the urban forest (Figure 2). If EAB were to move in Beaver Bank, it may be susceptible to more ash losses than initially expected. Furthermore, it may be an area where the EAB may proliferate without detection. It seems justified to intensify sampling for ash in this community.

Cole Harbour, Eastern Passage, and Fairview/BLT had relatively low sampling and low amounts of ash. These areas had no i-Tree or UFORE plots with ash trees. They did have roughly the same amount of ash trees planted for the UFMP (9 or 10 ash trees in each, from a total of 846-907 trees planted) (Table 3). This may indicate that these communities may have more street ash trees on account of ash being planted in the streets in previous years that were not recorded in any type of inventory or study. Overall, it can be said that these communities most

likely have less ash than others, especially when considering that i-Tree was a stratified random sampling of the 10 communities (20 plots each). Therefore, they may be less susceptible to a loss in canopy cover should the EAB arrive.

According to the i-Tree report, approximately 7435000 trees are in the urban forest of the HRM (Foster & Duinker, 2017). Considering the percentage of ash found among the 14843 trees, 2.54%, measured in the urban forest (Table 4), then there may be approximately 189 thousand ash trees in the urban forest of Halifax. According to the total monetary benefits of trees in the urban forest of Halifax, the total benefit of the estimated amount of ash trees in the urban forest, 189 thousand, would be $39116 or $126238, based on the different calculation methods used in the i-Tree report, the current price of carbon, and the social cost of carbon (Foster & Duinker, 2017).

Ash trees in the rural forests appear to be quite scarce, accounting for only 0.02% of the total FRI (Table 2), and 0.14% of the trees sampled in PSP data and UFORE rural data (Table 1). In terms of canopy loss, the rural forest could lose only a tiny portion of its overall canopy cover to the EAB. However, ash trees were not evenly distributed across the HRM (Figure 1). They cluster along the northwestern border of the HRM. This could entail a loss of some canopy cover in specific regions if the EAB moves through, since ash trees are clustered in a smaller area. Moreover, these areas with high ash can be areas where the EAB may proliferate and disperse into due to the proximity of the ash trees. There are several ash stands (Figure 1) along the boundary of the urban and rural forest. This connection can be a causeway for the EAB to move from the rural forest to the urban forest, if the rural forest becomes infested before the urban forest. EAB may also proliferate due to the higher basal area sum and average density in i-Tree plots found in Beaver Bank and Bedford (Table 3, Figure 2). Lastly, there may be black ash in

the natural stands of the rural forest, which is a threatened species, and valuable to Indigenous communities (Hurlburt 2011, Costanza et al. 2017, NS DNR 2017). In sum, though the rural forest may not lose much canopy cover overall, significant damage may occur at a finer scale.

A variety of studies have researched the dispersal of the EAB through forests. A study modeling the spatial spread and dynamics of the EAB found that rapid spread depends on an initial anthropogenic introduction, coupled by clusters of ash where the EAB hits its carrying capacity and is then forced to move to a new location, assuming a spread of 1.4 km/year (Bendor et al. 2006). A study conducted in Maryland, where extensive regulatory practices were placed to contain the spread of EAB from an infected nursery, found that the EAB spread at an average rate of 1 km/year, and up to 1.37 km/year (Sargent et al. 2010). Another study found that EAB-infected trees were found up to 638 m from the initial infected site after one year, in a heterogeneous distribution of ash trees amongst canopy cover (Siegert et al. 2010). Based on these studies, a rate of 1 km/year was used to estimate the approximate dispersal distance of the EAB should it arrive to the HRM.

When applying the 1 km/year metric to the urban forest, it appears that the EAB may spread at a higher rate in the urban forest than the rural forest. For example, when comparing the distances between ash trees found in high sampling intensity communities, such as the Halifax Peninsula and Dartmouth, ash trees are found much closer to each other, and within 1 km radii. When considering that it was found that the more sampling conducted, the more ash trees found, it is possible that those same proximities are found in the Beaver Bank, Bedford, Spryfield, and Sackville communities. Furthermore, this can especially be seen when taking into account the Nitoslawksi study data, which found a relatively high number of ash in relatively smaller study sites (Figure 2, Table 3). The pattern of small distances and high clusters of ash trees can also be

seen in the UFMP data (Figure 2). Furthermore, given the sparser distribution of ash in the rural forest of the HRM, should the EAB be introduced or move into the area, it may take a much longer time to spread throughout the rural forest due to the longer distances between ash trees (Figure 3). In sum, the shorter distances and clustering of ash trees found in the urban forest suggest that EAB will spread faster in the urban forest than the HRM rural forest.

There are some factors that may affect both the rural forest and urban forest. The EAB is mainly introduced into new areas due to anthropogenic causes such as the transportation of firewood (Herms and McCullough, 2014). The transportation of firewood occurs in Halifax, especially given that it is the core of the HRM, where the majority of the population lives, with high traffic moving into and out of the city. The movement of firewood in the rural forests is also commonplace. Therefore, it cannot be said whether anthropogenic movement of firewood will affect the rural or urban forest more. However, it can be said that the anthropogenic introduction of EAB through firewood may be more likely than EAB dispersing naturally into the HRM. Furthermore, it has been found that tree size is not a factor affecting EAB dispersal at long distances (more than 200 m) (Siegert et al.2010). However, stressed trees are known to be more susceptible (Siegert et al. 2010, Tluczek et al. 2011). Both the urban and rural forests of the HRM deal with stress. Urban trees are mostly affected by factors such as higher pollution rates, poor soils, road salt, and vandalism, while rural trees are mostly affected by competition due to high tree density. Therefore, it can also be said that the impact of EAB on highly stressed trees may affect both rural and urban forests.

A lack of a comprehensive tree inventory that encompasses both the urban and rural forest was arguably the biggest limitation in the study. The FRI is spatially comprehensive for the rural forest but neither the accuracy nor precision of the data are sufficient to get a

comprehensive distribution of the ash trees in the rural forest. Therefore, the study had to rely on other previously collected data, where the studies were not evenly distributed but scattered around the urban and rural forest. A lack of tree inventory means it is unknown where all the ash trees are in the urban and rural forest, and therefore the extent of how much the canopy cover may be affected is essentially unknown.

The urban forest was split into communities for comparative purposes; thus, the data from the various studies were aggregated based on the communities. One of the benefits of such an aggregation is that it brings consistency in relation to the i-Tree data, where each community has 20 plots, and so allows for comparison between the communities, and different comparisons between the studies. However, separating the remaining data based on somewhat arbitrary community boundaries can lead toward losing the overall picture of the urban forest. This, in turn, can lead to false presumptions of the entire community, based on smaller studies within the community. The boundaries may also lead to underestimation of the number of ash within the various regions of the urban forest, such as locations of ash trees just outside the borders of communities in the rural forest unincorporated into the assessment of the communities. For the purpose of this study, however, dividing the data based on community allowed for a more thorough comparison, based on the different regions and characteristics of the urban forest in the HRM.

Another limitation in the study was aggregating the data, and the consequent ability to provide an accurate description of both the urban and rural forests. It was found that when attempting to display the data in a map by ash intensity in the urban forest, the map was overly cluttered. Due to the trend seen in which more ash trees were found with higher sampling intensity in the region, it was found that aggregating the ash trees from all data per community

provided an inaccurate display of regions with high or low ash-tree densities. Furthermore, the finer details of the data, such as the differences found in i-Tree data compared to the remaining data, were lost in this display. In sum, a balance between displaying the data in a manner that still provides sufficient information on the locations of ash trees needs to be found. Therefore, the map of the distribution of ash trees in the urban forest (Figure 2) was supported by a table of the intensities at which the ash trees were found (Table 3)

Chapter 6

In this study, the distribution of ash trees in the urban and rural forest of HRM were mapped using various studies and sources describing the canopy cover of the HRM forest. In the rural forest, a map was created that displayed the various datasets aggregated into one standardized point system based on the intensity of ash trees found in the individual data points (Figure 1). It was found that 0.02% of the FRI data were ash trees (Table 2), and that in the UFORE and PSP data, 0.14% of the trees measured were ash trees (Table 1). A map was also created for the urban forest, the purpose being to display the location of the various urban forest datasets (Figure 2), and further information on the intensities of ash trees is found in Table 3. Overall, it was found that 2.54 % of total measured trees in the urban forest were ash trees (Table 4).

In the assessment, it was found that the urban forest is more at risk for both the introduction and spread of the EAB. The urban forest was found to have relatively higher densities of ash trees, as well as closer clustering of ash trees, making dispersal much easier than in the rural forest. This is especially true when considering it has been found that the EAB may travel at approximately 1 km/ year. However, anthropogenic movement and propensity of EAB to move into highly stressed trees may affect both the rural and urban forest. This is because residents in both rural and urban areas transport firewood. As well, urban trees may be stressed due to higher pollution, while rural trees may be stressed due to more competition. In sum, however, it is important to note that given the apparent overall low density of ash trees, EAB may not have an altogether serious impact on the canopy cover of the HRM.

The distribution of ash trees presented in this study may be useful for forest managers and researchers looking to protect the population of ash trees, and overall canopy cover of the

HRM. Understanding the distribution of ash trees will allow for placement of traps in areas with more risk of EAB dispersal, such as areas with high urban stresses, and areas with higher densities of ash trees. Furthermore, the ash-tree distribution in HRM's urban forests facilitates a rapid dispersal of the EAB. Therefore, with the EAB coming towards Nova Scotia in future years, it is important that forest managers know where buffer zones of EAB infestation can be placed, the areas of high ash-tree density, and where EAB will most likely be detected.

For future research, it may be useful to assess the areas with high ash density for the level of stress on the trees, relative distances between ash trees, and range of ash tree species present. In the rural forest, this may be useful since relative distances between ash stands are not at a fine enough scale to assess how the EAB may disperse. Furthermore, black ash, a threatened, unique, and culturally significant species may be found in those high-density areas of ash in the rural forests (Hurlburt 2011, Costanza et al. 2017, NS DNR 2017). These stands should be monitored to conserve the species and protect it from further losses. In the urban forests, the areas where sampling for ash trees should be conducted more intensively are Beaver Bank, Bedford, Spryfield, and Sackville. These areas were found to have higher average densities and total basal area sums in the i-Tree plots. Therefore, they may be at a higher risk of EAB proliferating in those areas without detection, especially due to the lower number of EAB traps found in those areas.

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