An Assessment of the Potential Impacts of Emerald Ash Borer (*Agrilus planipennis* Fairmaire) on Virginia's Municipal Street Trees

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ABSTRACT

Emerald ash borer (*Agrilus planipennis* Fairmaire) (EAB) is an invasive, wood-boring beetle (Coleoptera: Buprestidae) introduced unintentionally to the United States from East Asia that infests and eventually kills native ash trees (*Fraxinus* spp.). First detected near Detroit, Michigan in 2002, EAB had spread to fifteen U.S. states by 2011, killing an estimated 50 million ash trees along the way. EAB was first discovered in Virginia in 2003 and re-infested the state in 2008, raising concerns over impacts that the invasive pest might have on municipal urban forests and street trees. Despite these concerns, little is known about native ash abundance in Virginia's urban forests; as a result, potential EAB impacts have been difficult to project. In this study, street tree assessments were conducted in fourteen Virginia municipalities using i-Tree Streets®, which is a software program developed by the U.S. Forest Service that uses field inventory data to estimate street tree abundance and composition along with the quantity and monetary worth of functional benefits provided by these street trees. In addition to estimating potential losses of functional benefits provided by native ash street trees, information obtained from Virginia Dept. of Transportation was used to estimate the potential cost of removing these trees from the street side. The assessment indicated that there are about 4,600 native ash street trees in the fourteen studied localities and that native ash species comprise about 2% of municipal street tree populations on average. The highest relative abundance of native ash was found in Winchester City (5.8% of all street trees) whereas Richmond City had the greatest number of native ash street trees (estimated at 1,417). In terms of species importance (which

accounts for both the relative abundance and relative size of trees in the population), only two localities (City of Roanoke and Town of Abingdon) had a native *Fraxinus* species among the top-five most important street tree species in the locality. In contrast, every municipality had at least one *Acer* species among the top-five, and eight of fourteen localities had at least one top-five *Quercus* species. Native ash street trees in the studied localities were estimated to provide functional benefits (energy conservation, stormwater mitigation, air pollution abatement, carbon sequestration, and aesthetic contributions) valued at over $535,000 annually, or roughly $38,000 per locality. In addition, carbon stored in these trees (about 17 million kilograms) was valued at nearly $277 thousand. The total estimated cost of removing lost ash trees was estimated at nearly $1.75 million, averaging about $124,000 for each municipality, and replacing the canopy cover and basal area provided by existing native ash street trees would exceed $17 million. In total, the studied localities would incur a gross financial impact of about $20.26 million due to losses of functional benefits and structural assets provided by native ash street trees.

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CHAPTER 1 – INTRODUCTION

The urban forest is the collection of all woody vegetation located within an area of dense human settlement (Nowak et al. 2008) and is a critical component of the natural and artificial infrastructure in every municipality. Trees are the dominant woody vegetation in most urban forests and can be found residing in parks, streetscapes, remnant forests, and other maintained and un-maintained areas where they provide a multitude of benefits for citizens and the environment. In the conterminous United States, it has been estimated that there are nearly 3.8 billion urban trees, their canopies overhanging 27% of urbanized lands on average (Nowak et al. 2001). Street trees – although accounting for a relatively small proportion of the trees in a typical city – are a conspicuous and critical component of most urban forests. Their proximity to people and their importance to our day-to-day lives make them the focal point of municipal forestry programs, encompassing the majority of management efforts and expenditures

Urban forests in general – and street trees in particular – provide a broad range of benefits to urban areas. Many of these benefits are intrinsic to the physiological function of woody plants. Examples of these so-called ecosystem services include stormwater abatement, energy conservation, air pollution removal, and carbon sequestration and storage (McPherson 2003). These services can potentially save hundreds of thousands of dollars in environmental management costs for municipalities annually and create job opportunities for urban forestry professionals that grow, plant, and maintain trees. Urban forests also provide benefits for communities that go beyond the environment. The presence of abundant, healthy trees in the urban environment has also been associated with higher real estate values (McPherson et al.

2005), lower crime rates, improved citizen health, and enhancement of outdoor recreation (Lohr et al. 2004).

There are a number of non-native, invasive pests (plants, pathogens, and insects) that threaten the health of North American urban forests and their provision of benefits. Many pests are brought to North America from foreign continents with similar climate and physiography, yet they often arrive to an environment with few natural enemies and abundant, vulnerable host plants (Liebhold et al. 1995). These circumstances allow pests to flourish in our native ecosystems and overtake the natural flora. In the United States, there is a growing list of insects that have been introduced over the last 60 years (Ball et al. 2007). It is estimated that nearly 2.5 exotic pests are introduced each year into the United States (Aukema et al. 2010). Most of these pests do very little harm to native ecosystems, in part due to environmental controls such as climate, physiography, or predators. However, some become tree pests, causing significant ecologic harm and economic loss to both forestlands and urban forests.

In 1928, an exotic tree pest was first reported in the United States that would eventually cause unprecedented harm to North America's urban forests (Stipes and Campana 1981). American elm (*Ulmus americana* L.) had been widely planted in suburbs and along city streets throughout the Northeast and Midwest during the early 20th century. The species' redeeming qualities (ease of propagation, fast growth rate, urban tolerance, and attractive form) had contributed to its overplanting, leading to near monocultures in some cities. Then Dutch elm disease (*Ophiostoma ulmi* Buisman *and O. novo-ulmi* Brasier), a fungal pathogen, was introduced from Europe, purportedly on veneer logs shipped from the Netherlands to Ohio (Brasier 1990). It was rapidly spread via two coleopterans: the native elm bark beetle (*Hylurgopinus rufipes* Eichhoff) and the smaller European elm bark beetle (*Scolytus multistriatus* Marsham). This insect-disease combination claimed nearly 77 million native elm trees, by the 1970s within urban areas of eastern North America, according to one Washington Post writer (McCombs 2001). This instance shows that low species richness, e.g. street tree monocultures, can drastically increase the susceptibility of urban forests to pest invasions.

Now the urban forests of North America are under attack by a newly-introduced pest, the emerald ash borer (EAB) (*Agrilus planipennis* Fairmaire). EAB was purportedly introduced from Asia in the 1990s through infested wooden packaging materials (Poland and McCullough 2006). Although EAB causes limited ecological and economic harm to ash trees (*Fraxinus* spp.) in its native habitat, the pest has decimated native ash populations in parts of North America where it has been introduced (Sydnor et al. 2007). First detected near Detroit, Michigan in 2002, EAB has rapidly dispersed throughout several Midwestern and Eastern states (Asaro 2006, 2008), and its spread is predicted to encompass 25 Eastern states by 2019 (Kovacs et al. 2010). Since its introduction, EAB has led to the destruction over 53 million native ash trees, with another 38 million predicted to be affected in urban areas alone over the next decade (Kovacs et al. 2010). It is projected that the cost of treatment, removal, and replacement of urban ash trees impacted by EAB will exceed $10 billion by 2019 (Kovacs et al. 2010).

EAB was first discovered in Virginia in 2003 at an elementary school in Fairfax County, brought from Michigan in illegal ash tree nursery stock that was subsequently planted on the school grounds (VDACS 2010a). An aggressive eradication effort was undertaken, and a subsequent trapping survey suggested that the pest had been successfully eliminated from the state. However, in 2008, EAB was found again in three locations in Fairfax County (Asaro 2008). Survey trapping throughout Virginia in 2009 revealed no EAB detections outside Fairfax County (VDACS 2009). About 4,000 survey traps were deployed throughout the state in spring 2010 to continue EAB monitoring efforts (VDACS 2010). At present, there is a federal quarantine on seven counties (Arlington, Clarke, Fairfax, Fauquier, Frederick, Loudon, and Prince William) in the state (APHIS 2008) and a state quarantine on eleven localities (Arlington, Fairfax, Fauquier, Loudoun, and Prince William, and the cities of Alexandria, Fairfax City, Falls Church, Manassas, Manassas Park, and Winchester) (VDACS 2010).

Native ash abundance in Virginia's urban forests is not well documented; therefore, the potential impacts of EAB outbreak on Virginia's municipalities are uncertain. Although urban tree inventories have been performed in various localities throughout the state, these data have not been collected in a standardized manner nor have they been analyzed in a comprehensive fashion. Data from the US Forest Service (USFS) Forest Inventory Analysis (FIA) program suggest that native ash species generally occupy only 1% of the standing tree count in rural forests in Virginia (USFS 2008). While these data might provide insight on native ash abundance in remnant forests within urban areas, ash abundance in managed landscapes may differ significantly because people exert much more control on species composition in managed landscapes. Kirwan et al. (2007) inventoried landscape trees on 105 K-12 school campuses in

Virginia during 2000 – 2005 and found that green ash (*Fraxinus pennsylvanica* Marshall) accounted for only 2% of inventoried trees; no other native ash species exceeded 1%. To the extent that school campuses are a reflection of the communities in which they reside, these findings may imply low abundance of native ash in Virginia's urban forests. However, more comprehensive assessment of these urban forests is necessary to document native ash abundance, which is an integral first step for determining the potential impacts of EAB in Virginia.

Given the severity of EAB impacts incurred by numerous Midwestern cities – both in ecological and economic terms, Virginia localities would be wise to begin preparing for impending EAB outbreaks. Yet Virginia municipalities have practically no information on the abundance of native ash trees in their street tree populations, nor do they have an empirical understanding of the contribution that these trees make to the local community and environment. Without this information, municipalities cannot fully appreciate the potential impacts of an EAB outbreak and thus cannot adequately prepare a response plan. The goal of the current study was to address these information needs by assessing street tree populations in select municipalities throughout the Commonwealth of Virginia. This study had three main objectives:

1. Assess the abundance, composition, and condition of native ash species in street tree populations from select Virginia municipalities.
2. Quantify the relative magnitude and monetary value of benefits provided by ash street trees in these localities.
3. Estimate the potential cost of removal and replacement of these ash street trees in the event of an EAB outbreak.

CHAPTER 2 – LITERATURE REVIEW

2.1 The Urban Forest

2.1.1 Resource Overview

The urban forest comprises planted and naturally occurring trees within the municipal boundary (Johnston and Shimada 2004). As a unit, this resource is controlled and influenced by constituents of the municipality. Naturally occurring trees in remnant forests are present due to natural selection and regeneration and are usually managed differently than trees planted near buildings and streets, which are typically selected by people for diversity or specific characteristics (e.g. form, tolerance of site conditions, pest resistance, etc.).

Street tree populations also comprise both naturally occurring (such as those located in forest fragments next to roadsides) and planted trees within public rights-of-way. The street tree population is one aspect of the larger municipal urban forest. With nearly four-fifths of the United States population residing in urban areas, it is essential that street tree populations be healthy, diverse, and properly situated so that benefits are maximized and costs are minimized.

However, urban trees are often subjected to inhospitable conditions and vulnerable to attacks from insects and pathogens. Non-native pests are an important consideration for urban forest and street tree management in the United States. These pests are capable of eradicating entire taxa from the urban forest, as was the case with Dutch elm disease (*Ophiostoma novi-ulmi*), which nearly extirpated native elms (*Ulmus* spp.) from urban forests during the 20th century. As the world becomes more dependent on global trade, new threats to urban forests and street trees are emerging at an unprecedented rate (Aukema et al. 2010). Arguably, the greatest pest threat to urban forests and street trees currently in the United States is emerald ash borer (EAB), an invasive exotic insect that threatens to eliminate native ash (*Fraxinus* spp.) from much of the eastern United States.

2.1.2 Resource Abundance and Composition

Nearly 4% of the total land area in the United States has been classified as urban and is estimated to contain nearly 80% of the United States' population (Nowak et al. 2001). The influx of people moving into urbanized areas has caused peri-urban forests to be denuded for buildings, parking lots, and utilities. There are approximately 4 billion trees located in urban areas of the conterminous United States, equaling roughly 17 trees per urban inhabitant (Nowak et al. 2001). These trees provide canopy cover over about 27% of urban lands on average. It has been estimated that there are over 60 million street trees in the U.S. (Kielbaso 1990), which suggests that there is one street tree for every 67 trees beyond the road edge in any given urban forest.

In the Commonwealth of Virginia, urban areas constitute about 8% of total land area, roughly 8,800 square kilometers (Nowak and Crane 2002). There is an estimated 157 million urban trees in Virginia (Nowak et al. 2001); however, there is limited comprehensive information about species composition in Virginia's urban forests. Tree canopy coverage in these areas has been estimated at about 35% (Nowak et al. 2002), but this is based on low-resolution (1-km^2) Landsat data. Researchers at Virginia Tech have recently conducted urban tree canopy (UTC) assessments in 23 localities using high-resolution (1-m^2) NAIP data and have seen great variance in canopy coverage across the state. Lynchburg, Radford, and Arlington lead Virginia municipalities with the highest percentage of area covered in canopy (58%, 53%, and 52% respectively). In contrast, Purcellville and Woodstock have the least amount of canopy coverage at 20% and 22% respectively (Program 2011).

Forest Inventory Analysis (FIA) data from the USFS provide some insight on tree species composition in Virginia's forests (USFS 2010). FIA data from 2007 indicated that the top-five most abundant species (in terms of stem count) were red maple (*Acer rubrum*), loblolly pine (*Pinus taeda*), yellow-poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), and blackgum (*Nyssa sylvatica*) (Rose 2007). *Fraxinus americana* (ranked 28th) and *F. pennsylvanica* (ranked 32nd) were the only native ashes among the fifty most abundant species. In total, FIA data indicated that there were about 178 million native ash trees in Virginia's forests, which accounted for only about 2.2% of all forestland trees (Rose 2007). However, these FIA plots are located on a randomly generated grid, which have very few plots located in urban areas. These estimates are likely not an accurate indication of native ash abundance in urban forests.

Little is known about native ash abundance in Virginia's urban areas. Based on estimates of native ash density and canopy cover on developed lands in sixteen eastern cities (none in Virginia), Kovacs et al. (2010) estimated that there are about 1.3 million ash trees in Virginia and District of Columbia. In a study conducted on 105 K-12 public school campuses across Virginia, researchers found that the ten most important species of landscape trees (>12.5 cm diameter) comprised mostly native taxa within *Acer, Pinus,* and *Quercus* (one notable exception was *Pyrus calleryana* "Bradford") (Kirwan et al. 2007). Only one native ash species (*Fraxinus americana*) was among the thirty-five most important species surveyed on school grounds. It was found on only 10% of school campuses and comprised only 1% of the total school campus tree inventory statewide. Although many localities in Virginia possess street tree inventories that might provide insight on urban tree composition, these data are rarely available to the public in a comprehensive fashion.

2.1.3 Resource Benefits and Values

Urban forests provide numerous benefits with tangible monetary value for citizens within municipalities (Nowak et al. 2008). These benefits are related to both the function and structure of the urban forest. Among the most significant functional benefits of urban forests are real estate value enhancement, temperature moderation and energy conservation, stormwater abatement, air pollution mitigation, and carbon sequestration. Urban forests also have value as structural assets. This structural value is derived from long-term carbon storage as well as the replacement value of the green infrastructure itself. Street trees are often on the front-line of functional and structural benefit provision given their proximity to people and built infrastructure.

Once trees and other vegetation are planted outside a home or office building, the real estate value of the residence can increase by as much as seven to ten percent (Buhyoff et al. 1984). A study in Portland (OR) revealed that street trees located adjacent to residential property increased the home value by $8,900 on average. Also, all trees in Portland yielded nearly $1.35 billion in monetary benefits (Donovan and Butry 2010).

Street trees shade and cool hardscapes of municipalities (Nowak and Dwyer 2007). Ultra-violet radiation is captured by the leaves (nearly 90%), leaving hardscapes cooler and less likely to rapidly degrade (Heisler 1986) and reduces re-radiated heat from these surfaces. By intercepting solar radiation and through evaporative cooling, trees also facilitate energy conservation. When broadleaf trees are planted on the south aspect of a domicile, they cast shade upon the structure during the summer months and effectively cool the area under the tree by about 5 °C (Heisler 1986). If evergreen trees are planted on the north aspect of a landscape, harsh winter winds can be blocked, thus slowing heat loss from the domicile (Heisler 1990, Akbari et al. 1992).

Leaves and stems, in combination with the soil surrounding the root system, also act as 'traps' for stormwater runoff. In a storm event, water quickly runs off compacted soil and impervious surfaces of roads, parking lots, and other hardscapes. If trees are planted near the roadside, more rainfall will be intercepted by the canopy and accompanying root systems, preventing nearly 26% of intercepted rainfall from running off (Neville 1996). The soil immediately surrounding the tree provides the stormwater a place to infiltrate, saturate, and slowly release. Highly urbanized areas can also use trees and structural soils (soil created from several media homogenized together) that allow for water penetration and root growth as stormwater detention infrastructure (Day and Dickinson 2008).

In addition to energy conservation and stormwater abatement, trees improve air quality through pollution mitigation. As leaf gas exchange occurs, air quality will increase by removing gaseous pollutants from the atmosphere through stomata openings (Smith 1990, McPherson 2003). Trees can also filter harmful particulates from the atmosphere. In Chicago, for example, urban trees are estimated to remove nearly 1% of all ozone from the atmosphere (Nowak 1994).

Urban trees also sequester significant amounts of carbon on an annual basis. During the growing season, trees remove carbon dioxide from the atmosphere and use this molecule to synthesize glucose, the primary foodstuff for photosynthetically active plants. Each year, urban trees in the U.S. sequester nearly 23 million tons of carbon, valued at $460 million annually (Nowak and Crane 2002).

Trees in urban forests are also structural assets to municipalities, an infrastructure analogous to the bricks-and-mortar of a factory that creates widgets for revenue. Because trees are perennial plants, they store significant amounts of carbon in the wood that composes branches, trunks, and roots. In total, it is estimated that urban trees in the United States store 700 million tons of carbon, which is valued at over $14 billion (Nowak and Crane 2002). These researchers also estimated urban tree carbon storage in Virginia at nearly 30 million tons, the sixth largest stock of urban tree carbon storage in the U.S.

Much like any other infrastructure asset in a city, urban trees can also be valued using appraisal methods that estimate the cost to replace the assets. With urban trees, this value is often called a compensatory or replacement value. Using the Council of Tree and Landscape Appraisers method, compensatory value has been estimated for street trees in certain municipalities, the entire street tree population of the United States, and for the urban forests of the coterminous United States. Nowak et al. (2002) estimated the value for structural replacement of urban trees within the United States at over $2 trillion. New York City had the highest compensatory value for urban trees amongst investigated municipalities at $5.2 billion. Virginia was estimated to have nearly $100 billion in compensatory value, which was the sixth highest total in the coterminous United States (Nowak et al. 2002).

2.1.4 Resource Threats and Vulnerabilities

Many people for various reasons manage the urban forest: city officials, horticulturists, arborists, and municipal constituents all influence the structure and function of the urban forest. Due to the limited growing space in urban planting sites (soil pits adjacent to roadways, islands in parking lots), poor quality soil (Wray 2003), and urbanization, urban trees may suffer poor health or structural instability. In parking lots and along streets, trees must survive the heat island effect (re-radiated heat from the hardscape), injury from road salt, poor soil conditions, and pedestrian or vehicular traffic. Stressful conditions can make urban trees more susceptible to pests and other disorders that can lead to poor growth and early mortality (Kozlowski 1969). These factors also have the potential to cause safety hazards, poorly developed root systems, stunted growth, lowered vitality, and loss of ecosystem benefits to the municipality. Due to these potential issues, urban trees require careful planning and management to maximize benefits and value. If trees are not protected and managed properly, health issues can overwhelm municipal resources and attribute to premature tree mortality.

As trees are predisposed by environmental and human-related stress factors, inciting factors arise that can lead to tree mortality (Roberts 1977). Fungi, bacteria, and insects can overcome compromised defense mechanisms of trees. Insects, such as foliage feeding beetles and wood boring beetles, can attack stressed trees. As trees become stressed, resources are not allocated to contain or tolerate insect damage. Over time, numerous infestations of a stressed tree can lead to premature mortality. This fact has lead to several urban tree pests becoming severe enough to threaten the structure, function, and value of the urban forest.

Improper cultural practices also negatively affect urban trees. Improper pruning, such as ‘topping,’ creates pathways for decay organisms to enter, weakening the trees’ structure (Gilman and Knox 2005). Trees that have been pruned improperly may respond with epicormic branches that are usually poorly attached and can potentially fail in high winds or heavy precipitation events.

Urban trees can also be inflicted with a condition called heat stress. Trees located in parking lots and along streets encounter the heat island effect; radiant heat from the sun is magnified due to the dark surfaces of the road or pavement. This excess heat may cause trees to have depressed vigor, early senescence, and poor vitality (Roberts 1977, Cregg and Dix 2001). Urban trees affected by heat stress are also predisposed to drought-like symptoms. These characteristics include wilting of leaves, crown dieback, epicormic branching and eventually death.

As the global economy expands and international shipping expands, the threat of accidental introduction of exotic pests becomes greater (Aukema et al. 2010). Insect pests can be introduced to new areas via wooden packing material, nursery stock, firewood transport, and other means. These introductions can prove to be disastrous from both an economical and ecological standpoint.

In a balanced ecosystem, insects co-evolve with host defense mechanisms, natural enemies, and a suppressive climate. These natural control factors serve as a system of checks-and-balances to the pest population. Non-native insect populations establish in new habitats often do so without natural enemies (DeBach 1964). Thus, these non-native pest populations may increase in an unchecked manner and damage local flora. As the population increases, then the number of plants attacked becomes greater, possibly leading to total extirpation of the native species. A lack of species diversity within the forest attributes to the damage more quickly.

Over the last 100 years, the United States has seen several major tree pest infestations resulting from the introduction of exotic organisms and lack of tree diversity (Liebhold et al. 1995, Aukema et al. 2010). Aukema et al. (2010) estimated that on average about 2.5 non-indigenous forest pests are introduced to the United States each year. For example, a popular street tree, American elm (*Ulmus americana* L.), was decimated by an introduced pathogen (*Ophiostoma novi-ulmi*) and insect (*Hylurgopinus rufipes* Eichhoff and *Scolytus multistriatu*s Marsham) combination. This insect/pathogen combination is a classic example of how monoculture plantings and lack of species richness can decimate the urban forest as a whole. Hence, the reason species diversity is paramount to properly managing the urban forest.

There are over 400 species of phytophagous insects that are naturalized to the United States (Raupp et al 2006). These species range from innocuous to devastating. By 1977, it was estimated that DED had contributed to the death or removal of nearly 50 million elm trees (Schlarbaum et al. 1997). The majority of these pest species are inciting factors that contribute to the mortality of pre-disposed trees. Many insect species must co-infest with other insects and pathogens before tree mortality is possible (Mayo et al. 2003). For example, native oaks (*Quercus* spp L.) may need to have both gypsy moth (*Lymantria dispar* L.) and forest tent caterpillar (*Malacosoma disstria* Hubner) infestations in coordination with microclimate stress and soil conditions before tree vitality is expended and health declines rapidly (Mayo et al. 2003).

Many exotic pests introduced to the United States have caused both economic and ecological damage. The recent introduction of EAB also has the potential to dramatically alter the urban landscape as well potentially extirpating all native ash trees within the United States. With the new tools available for street tree assessment, we have the opportunity to anticipate and prepare for the full impact of EAB on Virginia's urban forests.

2.2 Emerald Ash Borer

2.2.1 Pest Overview

Emerald ash borer is a phloem-feeding beetle first named in 1888 in southeastern China, which is within its native range (Fairmaire 1888). Thought to have been first introduced into the United States in the late 1990s, this insect has quickly become the most catastrophic tree pest epidemic in U.S. history (Kovacs et al. 2010). Since its introduction, it has killed nearly 56 million native ash trees in the United States, and costs of mitigation could exceed $25 billion (Aukema et al. 2010).

Since the early 2000s, EAB has spread to over 20 states. Virginia was originally colonized by EAB in 2003; however, after remediation and sanitation by the USFS, the outbreak seemed to have been eradicated. In 2008, another EAB outbreak was found in northern Virginia and another colony was discovered outside Winchester in 2010. It has been estimated that urban areas of Virginia and the District of Columbia contain nearly 1.3 million native ash trees and the cost to treat and remove ash trees impacted by EAB has been placed at $641 million (Kovacs et al. 2010).

2.2.2 Pest Ecology

Emerald ash borer is a metallic-colored, flat-headed, wood-boring beetle (Coleoptera: Buprestidae) that is native to East Asia. Mitochondrial evidence from Michigan State University suggests EAB specimens found in the United States are related to EAB from the Heilongjiang and Tianjin Provinces of China (Wei et al. 2007). In its native range, EAB does not cause significant damage to indigenous ash species and is considered a minor pest. The insect feeds primarily on *Fraxinus* species in the United States (Anulewicz et al. 2007). The adults feed on the foliage of the crown; however, this does little harm to the overall vitality of the tree. The larvae feed during the summer months and create galleries within the stem and branches of the tree. These galleries restrict the flow of water, nutrients, and foodstuffs from the roots to the leaves and effectively girdles the tree. This girdling typically kills the tree within five years.

In early June, adult beetles emerge from the host tree in which they spent their juvenile and pupal stages to disperse, find mates, and reproduce. Adults feed on the foliage of host trees until sexual maturity is reached (Poland and McCullough 2006). The female lays 50 to 90 eggs in cracks, crevices, and furrows on the outer bark of a host tree (Poland and McCullough 2006). The adult beetles usually live for about three to six weeks. In late July and early August, the eggs hatch and the larvae bore through the bark and into the phloem (Anulewicz et al. 2007). As the larvae feed and mature, they create S-shaped galleries within the phloem. The larvae ingest the phloem and grow in size, creating a progressively larger gallery. The enlargement of the gallery can sometimes score the outer surface of the sapwood (Anulewicz et al. 2007). The larvae overwinter in the prepupal stage in shallow excavations in the gallery. Pupation begins in early April and requires approximately two weeks after which adults emerge in early June.

EAB creates many signs and symptoms as it infests the host plant. The first symptom the host shows is thinning of the canopy (Anulewicz et al. 2007). As the larvae feed, they damage the tree’s vascular system, leading to canopy thinning and dieback. As the larvae continue to develop, additional symptoms appear; epicormic sprouts proliferate from the base of the tree and dieback is evident in the uppermost parts of the canopy (Ball et al. 2007). This is the natural survival reaction of the host tree trying to reestablish leaf area for photosynthesis. The final life-stage sign of EAB is the D-shape exit hole in the bark surface that the adult insect creates after pupation and emergence from the tree (Poland and McCullough 2006).

EAB seems to prefer native ashes of North America while some oriental ashes show resistance (Anulewicz et al. 2007). The most common native ash species include white ash (*Fraxinus americana* L.), green ash (*F. pennsylvanica* L.), black ash (*F. nigra* L.), blue ash (*F. quadrangulata* L.), and red ash (*F. profunda* L.). EAB has not been documented to feed and infest any other genus in the Oleaceae family. Host preference by the insect does not appear related to tree age. However, stressed trees, such as those on parking lots, streets, and other stressful growing spaces, seem to be attacked first (Anulewicz et al. 2007). Once attacked and infested, trees usually expire in two to three years. This pest is difficult to control because it often goes undetected during early stages of infestation. Signs and symptoms of the insect often do not become apparent until the second season of infestation (Nzokou et al. 2008), making it difficult to control the pest and rehabilitate the tree.

2.2.3 Pest Introduction and Impacts

In the summer of 2002, EAB was first detected infesting white ash and green ash in the greater Detroit, Michigan area and soon thereafter was found infesting green ash in southern Ontario, Canada (Sydnor et al. 2007). It was most likely brought to North America in wood packing material constructed from wood products containing EAB larvae that was shipped from east Asia without having been properly treated for wood boring insects (Heimlich et al. 2008). Since being discovered, EAB has killed over 30 million ash trees in the Detroit, Michigan area and several surrounding states in the United States and several million ash trees in Ontario, Canada (Anulewicz et al. 2007). More recently, EAB has spread from Michigan, to Wisconsin, Ohio, Indiana, Missouri, Illinois, West Virginia, Pennsylvania, and Virginia (Asaro 2008, Kovacs et al. 2010). EAB has the capability of winged flight from one-half mile to up to three miles after emerging from the host tree (Taylor et al. 2005).

EAB has spread rather quickly from its original area of concentration in Detroit, Michigan. After its initial report in 2002, Ohio was the next state to have a confirmed infestation of EAB in 2003 (Nzokou et al. 2008). Also, in the same year, Maryland and Virginia both had small, isolated pockets of the insect discovered. Each state began an aggressive quarantine of the areas infested. EAB outbreaks in Virginia were thought to be eradicated; however, Maryland still remained quarantined with EAB (Asaro 2006) Indiana was the next state that was confirmed with an EAB attack. In 2006, resurgence was discovered in Maryland, and Illinois became another state with a confirmed infestation. Pennsylvania and West Virginia were then infested in 2007 (Anulewicz et al. 2007). In 2008, three more states were added to the list of confirmed EAB populations: Wisconsin, Missouri, and Virginia (again). In less than seven years, EAB had infested 10 states and two additional Canadian provinces. The quick spread has been helped due to the fact of limited natural control of the insect in North America and that native ash of North America have practically no natural resistance (Anulewicz et al. 2007).

In 2003, EAB was detected in Fairfax County, Virginia, adjacent to the Maryland border. The insect was introduced to the northern Virginia area via infested nursery stock, which had not been properly inspected (Asaro 2006). After a quarantine effort by the Animal and Plant Health Inspection Service (APHIS), the United States Forest Service (USFS) and the Virginia Department of Forestry (VDOF), the insect was thought to be eradicated from the area in early 2004. However, in 2008 the insect was again detected in Fairfax County(Asaro 2008) and was discovered near Winchester city in 2010.

To slow the spread of the insect, the Virginia Department of Agriculture and Consumer Services, along with the above agencies, have quarantined seven counties (Arlington, Clarke, Fairfax, Fauquier, Frederick (found in traps), Loudon, and Prince William) and six independent municipalities (Alexandria, Fairfax, Falls Church, Manassas, Manassas Park, and Winchester) (VDACS 2010b). Under this quarantine, it is unlawful to transport known native ash, which has not been properly treated, from a quarantined county into a county not quarantined.

The introduction of EAB to the United States has the potential to drastically alter the composition and ecology of urban and rural forests. Areas where native ash are naturally abundant or have been heavily planted will be affected the most. Ball et al. (2007) estimated that approximately 36% of street trees of select communities in South Dakota are populated with native ash species. Sydnor et al. (2007) estimated there are 4.3 million urban ash trees in Ohio. Native ash are particularly good street trees due to their fast growing nature and ability to tolerate stressful environments. Also, native ash trees are tolerant to alkaline soils (which are prevalent in the Midwest of the United States). Kovacs et al. (2010) estimated that managing native ash trees impacted by EAB on developed lands (treatment, suppression, and removals) will cost over $10 billion and will affect close to 38 million trees nationwide by 2019. In Virginia, Kovacs et al. (2010) estimates that over 1.3 million native ash trees will be affected at a cost of $641 million.

Two different strategies have been used to suppress EAB populations. The first method of suppression is through quarantine, which revolves around regulating the intrastate and interstate transport of native ash biomass (VDACS 2010b) from counties known to harbor EAB to counties that do not. In the EAB quarantine, regulatory organizations, such as the USFS, VDOF, and Virginia Department of Agriculture and Consumer Services (VDACS), monitor the spread of the insect using bait traps and intentionally girdled ash trees set up on a 0.5 mile by 0.5 mile gird to detect any spread of the insect (Ball et al. 2007). Although these traps are not universally applied, it identifies if the insect is moving from flight or with the help of humans.

The second strategy is to use conventional pest management techniques such as pesticides and natural enemies to prevent and suppress EAB outbreaks. Several insecticides have been found that effectively control EAB when injected into the tree trunk or drenched around the root system. Currently, standards dictate the use of imidacloprid, a systemic insecticide, to control population levels (Nzokou et al. 2008). However, new products such as TREE-age (emamectin benzoate) and Safari (dinotefuran) have shown promise in recent studies (Herms et al. 2009). These pesticides are most effective when applied to trees prior to EAB infestation. Moreover, limited research has been conducted on pesticide efficacy with trees over 63 cm trunk diameter (Herms et al. 2009).

Researchers are currently studying the efficacy of the entomopathogen *Beauveria bassiana* and releasing Asian stingless wasps that parasitize and kill EAB in its native range. Three parasitic wasps are being evaluated for their efficacy in controlling EAB larvae. Research has previously shown these wasps, especially *Tetrastichus planipennisi* (Yang), have the potential to kill 50% of EAB larvae before adults emerge (Yang et al. 2006) *Beauveria bassiana* (Bals. - Criv.) is a naturally occurring fungus that is being studied to determine its control efficacy on EAB if released prior to parasitic wasps (Castrillo et al. 2010).

2.3 Street Tree Assessment and i-Tree Streets

2.3.1 Assessment Overview

Past estimates have indicated that there are nearly 60 million street trees located within rights-of-way in the United States (Kielbaso 1990). Street trees are critical components of the urban forest; these trees shade sidewalks and streets (causing less UV light degradation), help calm traffic, reduce glare from headlights, reduce traffic noise, beautify the landscape, and help reduce the amount of water that runs off from hardscapes (Maco and McPherson 2003). To properly manage this resource and maximize its benefits to society, a resource assessment is periodically conducted as part of the resource management cycle (Miller 1997). A street tree assessment provides a detailed view of the street tree population (McPherson 2003). When performed correctly, street tree assessments provide critical information on tree health as well as the benefits trees provide for the municipality (Nowak et al. 2006). These assessments provide a wealth of information about street tree abundance, composition, and condition, aid in calculating street tree benefits and costs, as well as detailing vulnerabilities that may be present (McPherson 2003). This information is valuable to planners, arborists, and urban forest managers of the municipality so there is a better understanding of resource management needs.

Until the last decade, street tree assessments had focused solely on evaluating tree abundance, composition, and condition (i.e., the structure of the street tree population). As scientists have learned more about the function of street trees (and the relationship between structure and function), quantification of these benefits has been incorporated into the assessment process. The USFS pioneered this contemporary paradigm of street tree assessment to develop a software program called i-Tree Streets (Nowak et al. 2008). It is a peer reviewed, state-of-the-art program that has the capability to not only evaluate structural attributes of the street tree population, but also the quantity of benefits along with their monetary worth. This information on function and value expands understanding of urban forest benefits and provides evidence for resource conservation, investment, and management.

2.3.2 Street Tree Inventories

The street tree inventory is a management tool that has been used since the advent of urban forestry. Quite simply, a tree inventory is a record of the location and characteristics of individual trees within a well-defined group (Bond and Buchanan 2006). These data form the basis for planning and managing the urban forest (Miller 1997). Three types of tree inventories are commonly conducted by urban foresters: complete, partial, and sample inventories (Bond and Buchanan 2006). In a complete inventory, all trees located within a defined population or geographic area are enumerated. This creates an inventory that is highly precise, but requires extensive time and money to complete. A partial inventory is simply a complete inventory for a subset of a tree population or geographic area (e.g., only trees of a particular species or only on primary streets). In contrast to complete and partial inventories, a sample inventory procures data for a representative portion of the population and then uses statistical methods to calculate population level attributes. This method is much quicker and cost-effective than a complete inventory; however, the resulting statistics will possess some degree of uncertainty depending on sample intensity and variation in attributes of interest within the tree population.

Once an inventory has been completed, these data can be analyzed using software programs designed for various purposes. Inventory software programs are used to assess abundance and composition and to identify management needs for the street tree population (Smiley and Baker 1988, Bond and Buchanan 2006). These characteristics of the urban forest are valuable data that allow for determination of urban forest health and planning of management. Among these inventory analysis programs, i-Tree Streets has the distinction of being the only program that uses street tree inventory data to assess multiple aspects of street tree structure, function, and value.

2.3.3 i-Tree Streets

Currently, the most widely available street tree assessment software in the United States is i-Tree Streets (hereafter referred to as Streets). This peer-reviewed, empirically derived application was developed through a partnership of the USFS and the Davey Resource Group. Streets uses a complete or sample inventory to estimate street tree abundance and composition within a municipality. Sample inventories are created by generating a random sample of street segments within a municipality (typically 3–12% sample fraction) and then field measuring all trees residing on the sampled streets for key attributes (e.g. species, condition, and trunk diameter). These inventory attribute data are then used as inputs for empirical models that estimate (by species) stem count, leaf area, canopy coverage, and quantity and monetary value of functional benefits for the entire street tree population within the municipality (i-Tree 2008).

To predict street tree dimensions, Streets uses empirical models developed through research that began in the San Joaquin Valley of California in the summer of 1998 and then continued in other model cities throughout the United States (Peper et al. 2001, McPherson 2003). Researchers used regression analysis to build empirical models between trunk diameter and age (R^2 = 0.85), height (R^2 = 0.86), crown diameter (R^2 = 0.92), crown height (R^2 = 0.86), and leaf area (R^2 = 0.91). The original study entailed collecting field data from 12 different street trees species commonly found in several neighborhoods in the San Joaquin Valley. The neighborhoods were rated as young (those created between 1970 an 1990) and old (created pre-1970) neighborhood. This stratification allowed researchers to determine the age of the trees.

To develop empirical models for other areas of the United States, Streets researchers in Davis, California divided the United States into 16 regional climate zones based on Sunset's National Garden Book. Within each of these zones, a model city was selected and 800 street trees were randomly selected and intensively sampled. Sampling consisted of 40 trees of the 20 most common species growing in the region. Municipal records were solicited for each tree to determine its age and then other attributes were measured in the field (e.g. DBH, leaf area, crown height, tree height, etc.) (Peper et al. 2001, McPherson 2003).

Economic analyses of the street tree inventory through Streets uses specific values to determine the monetary value for functional benefits provided by each tree. Values from energy conservation were calculated from savings of electrical power and natural gas and were obtained from regional electric power companies. Stormwater runoff benefits were priced from local department of works as to budget augmentations and deflations from areas of high tree populations to low tree populations. Retention and detention ponds were taken into consideration, as well as sanitary treatment of stormwater. Median home sales price was also obtained from localities to allow for aesthetic benefit appraisal using the Council of Tree and Landscape Appraisers guide. Carbon storage and carbon sequestration values were defined from www.eCO2.com (i-Tree 2008).

In the past five to ten years, street tree inventories and assessments have become more popular and prevalent (Maco and McPherson 2003). Urban planners and municipality leaders are being urged to perform these analyses to permit more informed decisions about urban forest management. These inventories and assessments provide much needed information that can be used for policy decision-making and for bolstering funding for the urban forest (Peper et al. 2001). In particular, i-Tree Streets enables urban forest managers to better understand not only the structure of street tree populations, but also interpret the monetary value of the functional benefits provided by street trees. In addition to typical applications for resource management, i-Tree Streets can also be a valuable tool for identifying urban forest vulnerabilities to invasive pests and understanding the full scope of impacts that these pests might have on an urban forest, including losses of structural and functional values.

CHAPTER 3 – RESEARCH METHODS

3.1 Study Site Selection

This study was conducted over a three-year period (2008–2010) in fourteen municipalities within Virginia. To facilitate selection of these study sites, Virginia was first divided into nine regions based roughly on existing geo-physical and socio-political boundaries (Figure 3.1.1).

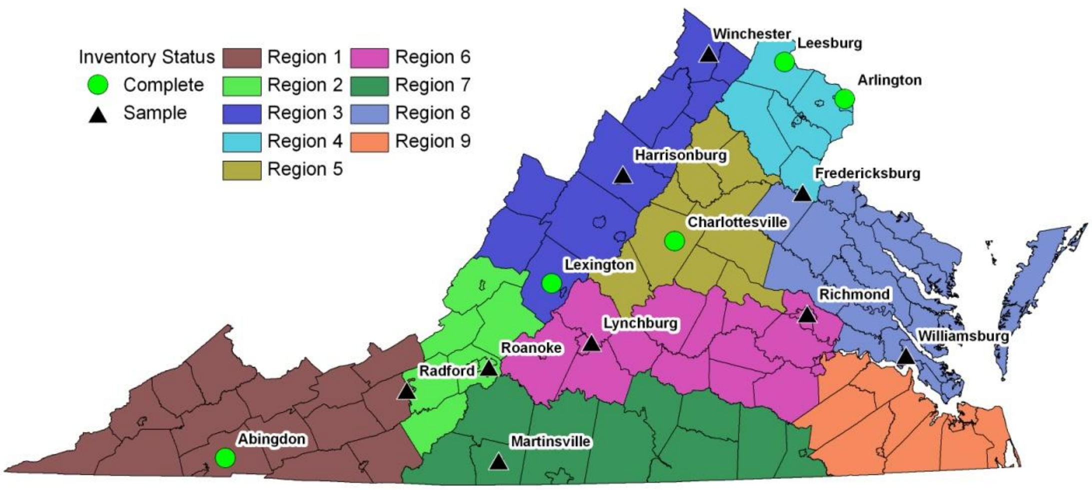

Figure 3.1.1: Virginia state map depicting the nine regions delineated for the street tree study and the locations of fourteen study sites symbolized by street tree inventory type (complete or sample inventory).

Within these nine regions, 132 localities were identified as study site prospects based on their designation as (1) an independent city, (2) a Tree City USA designee, or (3) a county seat (Figure 3.1.2). Contact information for the municipal agent responsible for street trees (i.e., forester, horticulturist, or engineer) in each locality was then obtained. Primarily focusing on independent cities and Tree City USA designees, about 60 municipalities were contacted to determine if a street tree inventory already existed (Table 3.1.1).

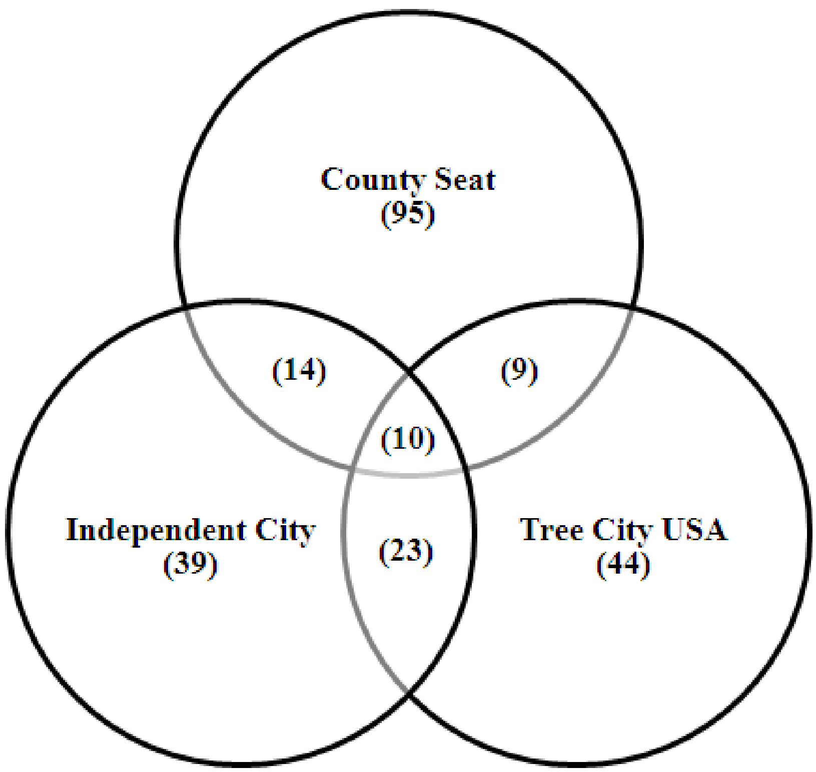

Figure 3.1.2: Venn diagram showing the composition of Virginia municipalities that were candidates for either obtaining existing street tree inventories or conducting new street tree inventories during the street tree study. Count is shown in parentheses.

Table 3.1.1: Street tree inventory status of Virginia municipalities that are a Tree City USA (TCUSA), county seat, or independent city based on contact efforts from 2008 to 2011.

	Existing Street Tree Inventory Status			
Locality Type	Yes	No	Unknown	Total
County Seat (Not TCUSA)	0	4	68	72
Independent City (Not TCUSA)	0	6	10	16
Tree City USA				
Independent City	9	12	2	23
Not Independent City	5	16	0	21
Total	14	38	80	132

Based on these contact efforts, existing inventory data were obtained from nine localities that met our selection criteria: (1) inventory less than 10 years old, (2) inventory containing street trees only, and (3) inventory representative of all street trees within municipal boundaries. To enhance geographic representation across the nine study regions, we also conducted sample street tree inventories in nine additional localities. These selections were not part of a statistical sampling design, but were rather purposefully chosen to broadly assess street tree populations across the state and provide management information for key municipal stakeholders. Geographic and demographic attributes for all fourteen municipalities included in the study are shown in Table 3.2.1.

3.2 Data Collection

In addition to compiling existing inventories from nine localities, sample street tree inventories were also conducted in nine Virginia municipalities. These inventories were performed, assessed, and reported using i-Tree Streets (Streets). The Streets inventory protocol entails random sampling of discrete street segments within a locality to enumerate and characterize all trees growing adjacent to the sampled street segments.

Table 3.2.1: Geographic and demographic attributes of fourteen Virginia municipalities selected for the street tree study.

Municipality	Tree City USA	Land Area[a] (sq. miles)	Population[b] (#)	Population Density (# / sq. mile)	Urban Street Miles[c]
Abingdon	Yes	8.3	8,004	964	55
Arlington	Yes	26.0	217,483	8,365	40
Charlottesville	Yes	10.3	42,218	4,099	135
Fredericksburg	Yes	10.5	23,193	2,209	82
Harrisonburg	Yes	17.6	45,137	2,565	138
Leesburg	Yes	11.6	40,927	3,528	105
Lexington	Yes	2.5	6,901	2,760	25
Lynchburg	Yes	49.4	73,933	1,497	377
Martinsville	Yes	11.0	14,635	1,330	100
Radford	No	9.8	16,184	1,651	71
Richmond	Yes	60.1	204,451	3,402	823
Roanoke	Yes	42.9	94,482	2,202	455
Williamsburg	No	8.5	12,729	1,498	50
Winchester	Yes	9.3	26,322	2,830	100

[a]Source: U.S. Census Bureau – Virginia QuickFacts, 2000 Estimate.

[b]Source: U.S. Census Bureau American Factfinder, 2009 Estimate.

[c]Source: Virginia Department of Transportation – Urban Street Miles Table of State Highway System, December 2009.

In preparing to conduct the sample street tree inventories, digital geospatial data (i.e., corporate limit boundaries, private parcel boundaries, and public street centerlines) were first obtained for each municipality. In the Streets protocol, public street centerlines (hereafter referred to as street segments) serve as the inventory sampling units. These street segments generally extend from cross-street to cross-street, but multiple abutting segments can be encountered on long stretches of road lacking intersections. For each municipality, a 4–12%

random sample of all street segments within the corporate boundaries was inventoried (Table 3.2.2). The specific sampling intensity chosen for each locality was based on its land area, land use mix, and development density with the goal of generating a total street tree population estimate with a standard error not more than 10% of the population estimate. Smaller, less-urban localities were sampled at higher intensities due to greater variation in street tree abundance amongst their street segments. In some localities, street segment sampling was stratified by land use in an effort to further reduce sampling error.

Table 3.2.2: Description of street tree inventories obtained in fourteen Virginia municipalities selected for the street tree study. Complete inventories did not employ a street segment sampling procedure; thus this information is not provided for complete inventories.

Municipality	Inventory Type	Total Street Segments	Sampled Street Segments	Percent Sampled	Date Inventoried
Abingdon	Complete	–	–	100	2007
Arlington	Complete	–	–	100	2003
Charlottesville	Complete	–	–	100	2009
Fredericksburg	Sample	1,304	127	9.7	2008
Harrisonburg	Sample	1,771	214	12.1	2010
Leesburg	Complete	–	–	100	2004
Lexington	Complete	–	–	100	2008
Lynchburg	Sample	4,175	375	9.0	2009
Martinsville	Sample	1,103	133	12.1	2009
Radford	Sample	1,784	212	11.9	2008
Richmond	Sample	14,249	553	3.9	2009
Roanoke	Sample	6,245	309	4.9	2008
Williamsburg	Sample	949	124	13.1	2010
Winchester	Sample	1,611	162	10.1	2008

For each municipality, ArcMap 9.3.1 (ESRI Inc., Redlands, CA) GIS software was used to select a random sample of street segments at the designated sampling intensity. Using ArcMap's Field Calculator tool, every street segment within the city was assigned a randomly generated number between 0 and 1. The tabular dataset was then sorted in ascending order by the assigned random numbers and a sample was selected from the sorted list of street segments equal to the designated sampling intensity (e.g., if a locality had 1,000 street segments and the sampling intensity was 10%, then the first 100 street segments in the randomized, sorted list were selected).

The sampled street segments were then overlaid on high-resolution aerial imagery of the locality and visually inspected for sampling suitability. Those segments that were atypically short or long, that had atypical right-of-way boundaries, or that had been recently impacted by land development (street closure or widening, building construction, etc.) were removed from the sample and replaced with additional, randomly-sampled segments. To ease identification of right-of-way boundaries in the field, a geoprocessing tool of ArcMap's 3D Analyst called Near Analysis was used to calculate the perpendicular distance from edge of street pavement to edge of right-of-way along the extent of each sampled street segment. Paper field maps were then created based on this analysis depicting the sampled street segment, its adjacent right-of-way boundaries, and its calculated right-of-way widths overlaid on aerial imagery of the vicinity. These maps were used to orient the field crews, determine whether trees were within the public right-of-way, and document the location of inventoried trees.

Field data were collected using a hand-held computer (Dell Axim x51v, Dell Inc., Round Rock, TX) running the Streets field application. For the purpose of this project, a tree was defined as any self-supporting, woody plant either greater than 2.44 meters tall or possessing a single stem within 0.30 meters of ground line. This definition captured large trees, large shrubs that function as trees, and small juvenile trees. All trees residing within the public right-of-way (naturally-occurring trees, publicly-planted trees, and privately-planted trees) were inventoried. Along streets retaining a forested character (i.e. an unmaintained ground cover), only trees greater than 10.2 cm diameter breast height (DBH) and not more than 6.1 meters from the edge of pavement were inventoried.

The following attributes were measured and documented for each inventoried tree:

- Tree ID number
- Street segment number
- Street name and address
- Botanical name
- Trunk diameter (DBH) class
- Age class
- Structural condition
- Functional condition (health)
- Land use
- Site type

The definitions and value ranges for these attributes are reported in Appendix A. The only quantitative measures taken in the field were tree distance from edge of pavement and DBH measurement, which were measured using a diameter tape or calipers. All other attributes were qualitative, categorical measures.

3.3 Data Analysis

Existing and sample street tree inventory data sets were thoroughly reviewed for missing and out of range values prior to analysis. Incomplete or erroneous records (e.g., a tree having a trunk diameter of 1,000 inches), were corrected when possible or discarded from the data sets. Data sets were then uploaded to a computer running the Streets desktop application, and a project file was created for each locality. Streets calculates the quantity and monetary value of tree benefits based on meteorological and tree-modeling data collected from reference cities throughout the US. For this study, calculations for all localities were based on the South Climate Zone, which uses Charlotte, NC as its reference city. After designating the climate zone, municipal attributes such as land area and human population were defined in the project set-up for each locality. Default values provided by STREETS for pricing ecosystem services and tree benefits were used for all municipal assessments. Finally, a species matching procedure was performed in the project set-up to define any tree species tallied in a project locality that did not exist in the South Climate Zone data base. For example, Chinese tupelo (*Nyssa sinensis*)—which is not in the modeling data base—would be best matched as blackgum (*Nyssa sylvatica*) because they are similar in stature, form, and growth rate.

Once the project files were prepared, Streets assessment reports were generated for each locality. Two report types were provided by the Streets application: a resource structural analysis (which calculates street tree abundance, composition, condition, and land-use occupation) and a benefit-cost analysis (which calculates gross ecosystem services, replacement value, and aesthetic/real estate contributions). To assign a value to the structural resource, the Council of Tree and Landscape Appraisers (CTLA) guidelines for tree valuation were followed. In this procedure, the estimated value of the tree is determined with a formula that uses the base price of a replacement nursery tree, and adds value to the base price by factoring in the tree (being appraised) species, condition of the tree, location of the tree, and the diameter of the tree (CTLA 2000). These characteristics of the tree allow for an accurate representation of the value of the tree that is being appraised.

To analyze the discrepancies between native ash removal by EAB, all native ash trees were coded as being private trees, except in those municipalities where native ash were present in high enough numbers to not warrant such a specific code. Added to these impacts were estimates of potential costs to remove and replace, based on the trunk formula method of the CTLA, the native ash component of each street tree population. Based on conversations with contractors from the Virginia Department of Transportation, the range for removing trees varies from $98 per tree under 15 cm to nearly $1,400 per tree for tree diameters that reach over 100 cm. By summing these costs with the lost ecosystem and aesthetic/real estate benefits, an estimate of the total economic impact of EAB outbreak for each locality was calculated.

In addition to estimating EAB economic impacts, correlation analyses were performed in an effort to better understand the relationship of native ash abundance in municipal street tree populations to the characteristics of municipalities in Virginia. Several bio-physical and sociodemographic variables hypothesized to be related to native ash abundance were tabulated for each locality in the study and compiled into a single database for the fourteen study sites. After screening the attribute data to ensure normal distribution, pairwise correlation analysis between each municipal attribute and native ash relative abundance (% of total tree population) was performed using JMP statistical software (SAS Institute Inc., Cary, NC). All statistical tests were performed at the α=0.05 significance level.

CHAPTER 4 – RESULTS

4.1 Structure, Function, and Value of Municipal Street Trees

Street tree population estimates ranged from 868 in Lexington to 46,792 in the state capitol of Richmond (Table 4.1.1). Across all municipalities, street tree populations averaged 14,798 trees. On a per-capita basis, Arlington had the least amount of street trees per person (0.09). In contrast, Fredericksburg had the highest per-capita street tree abundance (0.90). On average, each municipality contained 0.32 street trees per capita.

Table 4.1.1: Street tree population attributes and associated values for fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. For municipalities assessed with sample inventories, standard errors of the attributes and values are shown in parentheses.

Municipality	Street Tree Population	Top-Five Importance Value[a,b] (%)	CO_2 Storage Quantity[b] (kg)	CO_2 Storage Value ($)	Structural Replacement Value[c] ($)
Abingdon	1,193 (n/a)	50	1,818,574	30,070 (n/a)	2,829,814 (n/a)
Arlington	20,355 (n/a)	45	18,886,325	312,279 (n/a)	35,616,064 (n/a)
Charlottesville	5,988 (n/a)	32	12,642,818	209,045 (n/a)	28,892,459 (n/a)
Fredericksburg	20,792 (1,675)	42	20,237,274	334,617 (28,402)	48,940,606 (4,154,024)
Harrisonburg	6,985 (662)	26	6,937,249	114,705 (10,875)	12,889,427 (1,222,059)
Leesburg	3,088 (n/a)	30	2,339,793	38,688 (n/a)	5,297,012 (n/a)
Lexington	868 (n/a)	43	2,081,745	34,421 (n/a)	3,231,249 (n/a)
Lynchburg	26,820 (2,088)	31	27,260,859	450,749 (34,978)	60,480,330 (4,693,236)
Martinsville	3,566 (433)	39	3,463,036	57,260 (6,959)	7,075,858 (859,900)
Radford	12,724 (1,157)	33	16,973,730	280,655 (25,519)	32,728,620 (2,975,864)
Richmond	46,792 (3,645)	57	117,638,538	1,945,115 (151,529)	211,889,829 (16,506,728)
Roanoke	43,371 (3,433)	35	76,340,639	1,262,267 (99,917)	33,096,528 (2,619,825)
Williamsburg	5,640 (550)	40	7,566,145	125,104 (12,199)	19,061,256 (1,858,617)
Winchester	8,990 (774)	57	18,184,381	300,673 (25,875)	33,228,972 (2,859,631)

[a]Sum of the importance values for the five most important species (i.e., largest importance values) in the municipality. Importance value is calculated by summing the estimated leaf area, canopy cover, and stem count for all street trees, then determining the relative percentage that each species accounts for these metrics, and then averaging the values of the three metrics for each species.

[b]i-Tree Streets does not compute a standard error for the estimate of importance value or carbon storage quantity.

[c]Structural replacement value is calculated by i-Tree Streets using the trunk formula appraisal method of the Council of Tree and Landscape Appraisers whereby the worth of an existing tree is based on the cost to plant a quantity of nursery trees of equal trunk basal area.

Species diversity of the municipal street tree populations was investigated using a metric called importance value (IV), which accounts for both the relative abundance and the relative biomass of each species in the population. IV is calculated by first summing the stem count, leaf area, and canopy cover (area under tree dripline) for all species and then determining the percentage of each summed attribute that is accounted for by each species. The percentage value for each of the three structural attributes is then averaged for each species to derive the IV (McPherson et al. 2005). Like relative abundance of stems, the higher the IV, the more common the species is in a particular urban forest. However, IV gives a better indication of the dominance of particular species because it accounts for both tree number and tree size. Summing IVs for the five most important species in a locality provides insight on structural diversity and therefore population stability and resiliency (McPherson et al. 2005).

In the studied localities, summed top-five IVs ranged from 26% (out of 100% total species IV in street tree population) in Harrisonburg to 57% in Richmond and Winchester (Table 4.1.1). Top-five IVs averaged 40% across all fourteen localities. Looking more closely at the species comprising the top-five IVs, the genera *Acer* and *Quercus* dominated the important species across all municipalities (Table 4.1.2). Every municipality had at least one *Acer* species among the top-five, and eight of fourteen localities had at least one top-five *Quercus* species. In four localities, total IVs of the *Acer* species among the top-five exceeded 20%: Abingdon, Fredericksburg, Richmond, and Winchester. Both Charlottesville and Richmond had *Quercus* species among the top-five with IVs totaling nearly 30%. In total, there were 25 instances of *Acer* species among the top-five IVs and 12 instances of *Quercus* species across the studied localities. Only two localities (Abingdon and Roanoke) had a *Fraxinus* species among the top-five most important species. Nine localities exhibited a situation where one species garnered over 10% of the total IV.

Table 4.1.2: Five most important street tree species populating fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Importance values are calculated by averaging the % leaf area, % canopy cover, and % stem count that each species accounts for in the total tree population.

Municipality	#1 Important Species (IV)	#2 Important Species (IV)	#3 Important Species (IV)	#4 Important Species (IV)	#5 Important Species (IV)
Abingdon	*Acer saccharum* (17)	*Fraxinus americana* (10)	*Pinus strobus* (9)	*Acer saccharinum* (7)	*Cornus florida* (6)
Arlington	*Quercus phellos* (13)	*Acer rubrum* (13)	*Quercus palustris* (9)	*Acer saccharinum* (5)	*Quercus alba* (5)
Charlottesville	*Acer rubrum* (8)	*Pinus strobus* (7)	*Cornus florida* (6)	*Quercus alba* (6)	*Acer saccharum* (5)
Fredericksburg	*Acer rubrum* (15)	*Acer saccharum* (14)	*Liquidambar styrac.* (5)	*Quercus phellos* (5)	*Pyrus calleryana* (4)
Harrisonburg	*Ulmus pumila* (6)	*Pyrus calleryana* (5)	*Quercus phellos* (5)	*Juglans nigra* (5)	*Acer platanoides* (5)
Leesburg	*Morus rubra* (8)	*Acer rubrum* (6)	*Cornus kousa* (6)	*Quercus palustris* (5)	*Pyrus calleryana* (5)
Lexington	*Acer saccharum* (13)	*Ulmus americana* (9)	*Platanus occident.* (8)	*Juglans nigra* (7)	*Acer negundo* (5)
Lynchburg	*Acer rubrum* (8)	*Liriodendron tulipif.* (8)	*Robinia psuedoacac.* (6)	*Cornus florida* (4)	*Acer saccharum* (4)
Martinsville	*Quercus alba* (9)	*Acer rubrum* (8)	*Cornus florida* (8)	*Quercus falcata* (8)	*Liriodendron tulipif.* (6)
Radford	*Pinus strobus* (11)	*Acer rubrum* (6)	*Acer saccharinum* (6)	*Ulmus pumila* (6)	*Acer platanoides* (5)
Richmond	*Quercus phellos* (15)	*Quercus palustris* (15)	*Acer saccharum* (13)	*Acer rubrum* (8)	*Zelkova serrata* (6)
Roanoke	*Acer saccharum* (14)	*Ulmus pumila* (7)	*Fraxinus americana* (5)	*Acer platanoides* (5)	*Pinus strobus* (5)
Williamsburg	*Lagerstroemia* spp. (10)	*Quercus phellos* (10)	*Acer rubrum* (8)	*Pinus taeda* (7)	*Juniperus virginiana* (5)
Winchester	*Acer saccharum* (21)	*Acer platanoides* (13)	*Platanus occident.* (12)	*Ulmus pumila* (6)	*Pyrus calleryana* (5)

Based on the assessed municipalities, Virginia's street trees are in favorable health overall (Table 4.1.3). Averaged across localities, 89% of the street trees were rated as fair or good health. Street trees were rated as good health most frequently in Winchester (59% of trees) and least frequently in Martinsville (22% of trees). Radford had the highest proportion of street trees rated as dead or dying (3.4%) whereas only 0.1% of Charlottesville's street trees were rated as such. On average, about 10% of the street tree populations were rated as dead, dying, or poor health.

Table 4.1.3: Condition of street trees in fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Condition ratings were determined through visual evaluation of tree health and vitality only. Values are expressed in terms of relative abundance (% of total stem count in municipality) for each condition class.

	Condition Rating (% of Total)			
Municipality	Dead	Poor	Fair	Good
Abingdon	0.6	11.0	59.4	29.0
Arlington	2.0	13.6	65.3	19.1
Charlottesville	0.1	4.4	49.6	45.9
Fredericksburg	2.3	11.9	31.3	54.5
Harrisonburg	1.3	9.9	69.7	19.1
Leesburg	2.2	11.4	36.5	49.9
Lexington	1.3	10.1	21.3	67.3
Lynchburg	1.5	6.6	45.8	46.1
Martinsville	1.4	7.7	69.1	21.8
Radford	3.4	8.2	34.4	54.0
Richmond	2.6	10.0	35.1	52.3
Roanoke	2.4	12.0	41.7	43.9
Williamsburg	0.8	5.5	39.8	53.9
Winchester	0.3	3.1	37.8	58.8

Monetary worth of the structural assets in these street tree populations was assessed using i-Tree Streets' calculations of carbon storage and of replacement value. Total street tree carbon storage in these localities was estimated at over 332 million kilograms, a structural asset valued at nearly

$5.5 million (Table 4.1.1). Carbon storage ranged from about 1.8 million kilograms in Abingdon (valued at $30,000) to about 117 million kilograms in Richmond (valued at nearly $2 million). Average street tree carbon storage in each municipality was over 23 million kilograms and averaged $392,000 in value. Total replacement value of street trees in these localities ranged from $2.8 million in Abingdon to $211 million in Richmond (Table 4.1.1). The average replacement value across all municipalities was about $38 million and the total replacement value for all municipalities was over $535 million.

Functional benefits of street trees assessed in the studied localities included aesthetic enhancements, stormwater interception, energy conservation, carbon dioxide sequestration, and air pollution reduction. For ecological benefits (other than aesthetics), i-Tree Streets calculated both the quantity of benefits (referred to as "resource units" in Table 4.1.4) as well as the monetary worth of these benefits. Because costs of street tree management could not be obtained from every locality for this study, monetary worth of all functional benefits are reported on a gross basis (i.e., the management costs were not deducted from the benefits valuation). Aesthetic benefits of street trees in the studied localities were appreciable, totaling over $9.4 million. On average, each locality's street trees were estimated to provide over $675,000 in aesthetic enhancements to real estate. Richmond had the highest aesthetic benefit, close to $3 million, and Abingdon had the least at $26,000.

Street trees mitigate stormwater runoff by intercepting rain that might otherwise fall upon impervious surfaces and flow to stormwater drainage systems, thereby reducing the volume of water that must be handled by this infrastructure. On average, over 129,000 cubic meters of stormwater were estimated to be intercepted annually by street trees in each locality, valued at over $339,000 (Table 4.1.4). An Olympic-sized swimming pool contains roughly 2,500 cubic meters of water. Richmond had the highest annual benefit from stormwater mitigation at $1.64 million whereas the lowest annual benefit was estimated for Abingdon at about $25,000. In total, street trees in these Virginia municipalities were estimated to mitigate over 1.8 million cubic meters of stormwater annually at a gross value of close to $5 million.

Energy conservation estimates were based on street tree reductions in natural gas usage for heating in winter and electricity usage for air conditioning in summer. Richmond had the largest overall energy conservation benefit, with street trees conserving over 33,000 gigajoules of electricity annually at a value of about $521,000 (Table 4.1.4). Abingdon had the smallest energy benefit with street trees conserving 552 gigajoules of electricity annually at a value of $9,000. Average annual energy conservation benefits provided by street trees in these municipalities were estimated at about 7,000 gigajoules, with gross value of over $118,000. In total, street trees in these localities were estimated to conserve over 100,000 gigajoules of energy annually at a value of nearly $1.7 million. One barrel of crude oil (159 liters) contains about 6.1 gigajoules of potential energy.

Carbon dioxide sequestration estimates included both net sequestration by trees (photosynthesis minus respiration) and avoided emissions from power plants due to street tree reduction of building energy usage. On average, over 2 million kilograms of carbon dioxide were estimated to be sequestered annually by street trees in these localities at a gross value of nearly $35,000 (Table 4.1.4). Street trees in Richmond sequestered the highest amount of carbon dioxide annually, around 9 million kilograms (valued at about $153,000), whereas street trees in Abingdon sequestered just over 139,000 kilograms (valued at about $2,000). In total, street trees in these localities sequestered over 29 million kilograms of carbon annually at a gross value of about $486,000. The average American is responsible for about 19,000 kilograms of carbon dioxide emissions annually.

Estimates of air pollution reduction accounted for pollutants both intercepted by trees (O_3, NO_2, SO_2, and PM_{10}) and avoided emissions (NO_2, SO_2, PM_{10}, and VOCs) from power plants due to street tree reduction of building energy usage. From this reduction calculation was subtracted the emissions of biogenic volatile organic compounds (VOCs) by street trees. Thus in some instances, the net air pollution reduction (in resource units) by street trees was a negative value, meaning that trees were net emitters of air pollution. Depending on species composition and size distribution and on pricing of individual air pollutants, it was possible to have a negative value for both the resource units and monetary worth (or one or the other) in individual localities. Richmond's street trees were the only net emitters of air pollution (1,422 kilograms annually), yet eight out of fourteen localities had net negative values for pricing of air pollution benefits. Roanoke's street trees removed the largest amount of pollutants (over 5,000 kilograms), but had negative value of over $21,000. Fredericksburg had the second highest annual pollution reduction at 3,292 kilograms and also had a positive monetary impact at about $13,000, which was the highest of any locality.

Summed across all functional benefit types, street trees in these fourteen Virginia localities were estimated to provide over $16 million in gross annual benefits (Table 4.1.5), or roughly $1.1 million per locality. Richmond's street trees provide the highest gross annual benefits, valued at nearly $5 million, whereas Abingdon's street trees only provide about $62,000 in benefits. On a per-tree basis, annual street tree benefits averaged about $74 per tree, ranging from about $32 per tree in Martinsville to about $109 per tree in Winchester. On a per-capita basis, street tree benefits averaged about $23 per person, ranging from about $5 per person in Leesburg to about $73 per person in Fredericksburg.

Table 4.1.4: Gross annual benefits of street trees in fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Resource units are shown in shaded lines and their monetary worth is shown in un-shaded lines. i-Tree does not compute a standard error for resource units or for values calculated from complete inventories.

	Aesthetic Benefits (n/a)		Stormwater Interception (m^3)		Energy Conservation (*GJ*)		Net CO_2 Sequestration (kg)		Net Air Pollution Red. (kg)	
Municipality	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000
Abingdon	26		25		9		2		0.2	
resource units			9,387		552		139,278		175	
Arlington	1,699		280		102		31		-5.1	
resource units			106,946		6,708		1,887,688		1,411	
Charlottesville	350		168		58		17		0.4	
resource units			64,111		3,746		1,007,278		1,131	
Fredericksburg	1,193	101	329	28	132	11	37	3	13.2	1.1
resource units			125,729		8,663		2,234,501		3,292	
Harrisonburg	207	20	103	10	40	4	12	1	0.5	0.1
resource units			39,250		2,607		731,176		754	
Leesburg	147		34		14		4		-0.1	
resource units			12,886		901		238,273		231	
Lexington	48		27		9		3		-0.4	
resource units			10,208		552		156,323		131	
Lynchburg	624	48	412	32	159	12	48	4	-1.4	-0.1
resource units			157,698		3,276		2,881,381		2,745	
Martinsville	39	5	50	6	19	2	6	1	0.2	0.1
resource units			19,038		1,220		336,633		348	
Radford	319	29	243	22	88	8	27	2	1.9	0.2
resource units			92,790		5,733		1,620,344		1,776	
Richmond	2,827	220	1,642	128	521	41	153	12	-148.7	-11.5
resource units			627,773		33,729		9,270,269		-1,422	
Roanoke	1,070	85	1,062	84	373	30	109	9	-21.1	-1.7
resource units			406,039		24,271		6,597,462		5,197	
Williamsburg	296	29	111	11	40	4	11	1	-4.7	-0.5
resource units			42,487		2,603		665,547		408	
Winchester	617	53	262	23	91	8	26	2	-11.5	-1.0
resource units			100,113		5,923		1,546,924		773	

Table 4.1.5: Gross annual benefits (ecosystem services and aesthetic/real estate enhancements) provided by street trees in fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Values calculated from complete inventories do not have a standard error.

	Total Annual Benefits		Annual Benefits Per Tree		Annual Benefits Per Capita	
	$	SE	$	SE	$	SE
Abingdon	61,589	–	51.63	–	7.69	–
Arlington	2,107,244	–	103.52	–	9.69	–
Charlottesville	592,338	–	98.92	–	14.03	–
Fredericksburg	1,703,803	144,617	81.95	6.96	73.46	6.24
Harrisonburg	361,757	34,299	51.79	4.91	8.01	0.76
Leesburg	198,568	–	64.30	–	4.85	–
Lexington	85,003	–	97.93	–	12.32	–
Lynchburg	1,241,733	96,358	46.30	3.59	16.80	1.30
Martinsville	113,117	13,747	31.72	3.85	7.73	0.94
Radford	678,477	61,691	53.32	4.85	41.92	3.81
Richmond	4,995,158	389,135	106.75	8.32	24.43	1.90
Roanoke	2,593,302	205,278	59.79	4.73	27.45	2.17
Williamsburg	453,955	44,264	80.49	7.85	35.66	3.48
Winchester	983,585	84,646	109.41	9.42	37.37	3.22

4.2 Native Ash Composition in Street Tree Populations

Three species of native ash (*Fraxinus americana, F.nigra,* and *F. pennsylvanica*) were inventoried in the fourteen studied Virginia localities. *F. americana* was inventoried in 14 out of 14 municipalities, *F. pennsylvanica* in 12 out of 14, and *F. nigra* in 2 out of 14. In the studied localities, tree species of the genus *Fraxinus* were very uncommon in the street tree population. Combined relative abundance of native ash species ranged from 0.1% in Williamsburg to 5.84% in Winchester and averaged 2.0% across all localities (see Table 4.2.1). Importance values were also calculated using Streets. Only two municipalities had native ash as one of the top five most important species. Abingdon's native ash street trees tallied approximately 10% of the most important species. Although having low relative abundance of native ash, Richmond (3%) and

Roanoke (2.6%) still had appreciable ash populations estimated at 1,417 and 1,112 trees, respectively. Overall, an estimated 4,558 native ash trees reside in the street tree populations of the fourteen studied localities.

Table 4.2.1: Abundance and importance of native ash (*Fraxinus* spp.) street trees in fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. For municipalities assessed with sample inventories, standard error of native ash abundance is shown in parentheses. Values calculated from complete inventories do not have a standard error.

Municipality	Native Ash Abundance (count)		Native Ash Relative Abundance[a] (% of Total)	Native Ash Importance Value[a,b] (% of Total)
Abingdon	42	(n/a)	3.5	11.3
Arlington	275	(n/a)	1.4	1.0
Charlottesville	113	(n/a)	1.9	2.9
Fredericksburg	257	(113)	1.2	1.3
Harrisonburg	66	(31)	0.9	3.5
Leesburg	68	(n/a)	2.2	4.3
Lexington	16	(n/a)	1.8	2.9
Lynchburg	490	(136)	1.8	3.4
Martinsville	17	(11)	0.5	0.2
Radford	160	(53)	1.3	1.5
Richmond	1,417	(406)	3.0	3.8
Roanoke	1,112	(605)	2.6	5.8
Williamsburg	8	(7)	0.1	0.3
Winchester	517	(169)	5.8	4.7

[a]i-Tree Streets does not compute a standard error for the estimate of relative abundance or importance value.

[b]Sum of the importance values for native ash species in the municipality. Importance value is calculated by summing the estimated leaf area, canopy cover, and stem count for all street trees, then determining the relative percentage that native ash species accounts for these metrics, and then averaging the values of the three metrics for native ash species.

Of the fourteen municipalities, six assessments showed that the majority of the street tree population was greater than 30 cm in diameter (Table 4.2.2). Only four municipalities showed the majority of the street tree population as being less than 15 cm in diameter. Relative Performance Index (RPI) is a comparison between a single tree species and all other species in the street tree population based on relative condition rating in the inventory. If the RPI is below 1.00, then trees of that taxon are generally in poorer condition than the average tree in the population. If the RPI is above 1.00, then trees of that taxon are in better condition than the average tree in the population. On average, native ash street trees are performing just slightly below other street trees (Table 4.2.2) with an average RPI of 0.97. Harrisonburg's native ash street trees are performing the best of all studied localities (RPI of 1.02) and three localities (Abingdon, Lexington, and Richmond) share the same RPI for native ash street trees of 0.90.

Table 4.2.2: Native ash (*Fraxinus* spp.) street tree attributes and associated values for fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Values calculated from complete inventories do not have a standard error.

		Trunk Diameter Class[b] (% of Native Ash Trees)			CO_2 Storage Quantity[b]	CO_2 Storage Value		Structural Replacement Value[c]	
Municipality	RPI[a]	0–15 cm	15–30 cm	> 30 cm	kg	$	SE	$	SE
Abingdon	0.90	38.1	2.4	59.5	352,326	5,826	–	377,095	–
Arlington	1.00	71.3	17.5	11.2	144,492	2,389	–	261,188	–
Charlottesville	0.98	8.8	29.2	62.0	436,627	7,219	–	776,218	–
Fredericksburg	1.00	71.6	8.2	20.2	319,896	5,289	2,337	580,988	256,669
Harrisonburg	1.02	0	37.9	62.1	431,951	7,142	3,325	461,712	214,929
Leesburg	1.00	14.7	54.5	30.9	134,241	2,220	–	312,645	–
Lexington	0.90	12.5	31.3	56.2	73,632	1,217	–	114,989	–
Lynchburg	1.00	20.4	43.3	36.3	1,325,653	21,919	6,073	2,290,352	634,556
Martinsville	0.94	100	0	0	749	12	8	4,675	3,088
Radford	1.00	26.3	31.3	42.4	242,013	4,002	1,334	492,915	164,289
Richmond	0.90	9.1	20.0	70.9	5,250,422	86,814	24,856	9,038,222	2,587,807
Roanoke	1.00	29.1	16.4	45.5	7,111,173	117,581	63,911	1,603,326	871,490
Williamsburg	1.00	0	0	100	28,528	472	440	41,217	38,430
Winchester	1.00	57.6	19.1	23.3	871,278	14,406	4,710	1,341,406	438,524

[a]Relative Performance Index (RPI) > 1 indicates that native ash are in better condition than the typical street tree in the municipality and < 1 indicates that native ash are in poorer condition than the typical street tree in the municipality. RPI shown is the weighted average for all native ash species in the municipality.

[b]i-Tree Streets does not compute a standard error for the estimate of trunk diameter class or carbon storage quantity.

[c]Structural replacement value is calculated by i-Tree Streets using the trunk formula appraisal method of the Council of Tree and Landscape Appraisers whereby the worth of an existing tree is based on the cost to plant a quantity of nursery trees of equal trunk basal area.

To gain insight on factors that might influence native ash abundance in Virginia's street tree populations, multivariate correlation analyses were performed between native ash relative abundance (% of total street tree count) and sixteen municipal characteristics ranging from the date of foundation to demographic and environmental information. Within the fourteen municipalities studied, two significant correlations ($P<0.05$) were found. A positive, significant correlation was found between relative ash abundance and Years as a Tree City USA. A negative, significant correlation was determined for relative native ash abundance and average January temperature (see Table 4.2.3)

Table 4.2.3: Pairwise correlation analyses between native ash relative abundance (% of total tree population) and select municipal attributes hypothesized to be related to ash abundance in fifteen Virginia municipal street tree populations. Ash abundance calculated from street tree inventories using i-Tree Streets assessment software.

Municipal Attribute	Pearson Correlation Coefficient	*P*-value (Ho: $\beta = 0$)
Date Founded (year)	0.1365	0.6418
Municipality Age (years)	-0.1365	0.6418
Tree City USA (years)	0.5670	0.0345*
Human Population	0.0765	0.7950
Human Population Density (#/mi^2)	0.0128	0.9654
Household Income ($)	0.0018	0.9953
Per capita income ($)	0.1301	0.6577
Urban Street Mileage	0.2327	0.4233
Land Area (mi^2)	0.1263	0.6670
Average Annual Precipitation (in)	-0.1886	0.5184
Average January Temperature (°F)	-0.5483	0.0424*
Average July Temperature (°F)	0.0138	0.9627
Elevation (ft)	0.1146	0.6964
Latitude	0.2957	0.3047
Longitude	-0.1633	0.5770
Forest Ash Relative Abundance (% stem count)	0.1177	0.6885

* Statistically significant: $\alpha \leq 0.05$

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Table 4.2.4: Gross annual benefits of native ash (*Fraxinus* spp.) street trees in fourteen Virginia municipalities based on analysis of inventory data using i-Tree Streets software. Resource units are shown in shaded lines and their monetary worth is shown in un-shaded lines. i-Tree does not compute a standard error for resource units or for values calculated from complete inventories.

	Aesthetic Benefits (n/a)		Stormwater Interception (m^3)		Energy Conservation (*GJ*)		Net CO_2 Sequestration (kg)		Net Air Pollution Red. (kg)	
Municipality	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000	$ × 1000	SE × 1000
Abingdon	2.4		3.8		0.8		0.2		-0.9	
resource units			1,445		50.9		12,698		-52	
Arlington	18.5		2.3		0.9		0.3		-0.3	
resource units			892		59.6		16,455		-5.7	
Charlottesville	12.0		5.6		1.6		0.5		-1.1	
resource units			2,131		100		27,962		-51.2	
Fredericksburg	14.4	6.4	4.5	2.0	1.4	0.6	0.4	0.2	-0.8	0.3
resource units			1,720		90.5		25,158		-31.7	
Harrisonburg	6.0	2.8	5.0	2.3	1.2	0.6	0.3	0.2	-1.1	-0.5
resource units			1,919		76.4		20,510		-59.2	
Leesburg	7.5		1.8		0.6		0.2		-0.3	
resource units			696		37.1		10,522		-13.1	
Lexington	1.6		0.9		0.2		0.1		-0.2	
resource units			349		12.2		4,215		-9.4	
Lynchburg	22.8	6.3	17.5	4.8	5.0	1.4	1.5	0.4	-3.3	-0.9
resource units			6,672		326		92,011		-149	
Martinsville	0.1	0.1	0.02	0.01	0.02	0.01	0.01	0.003	0.003	0.002
resource units			9.0		1.3		297		0.5	
Radford	6.6	2.2	3.8	1.3	1.3	0.4	0.4	0.1	-0.6	-0.2
resource units			1,456		86.9		25,402		-18	
Richmond	122.6	35.1	71.2	20.4	20.4	5.8	6.2	1.8	-13.4	-3.8
resource units			27,233		1,321		376,941		-601	
Roanoke	52.3	28.4	81.5	44.3	18.9	10.2	5.2	2.8	-18.8	-10.2
resource units			31,145		1,191		315,901		-1,003	
Williamsburg	1.3	1.2	0.4	0.4	0.1	0.1	0.04	0.03	-0.08	-0.07
resource units			164		8.6		2,484		-3.1	
Winchester	28.1	9.2	11.2	3.7	3.3	1.1	1.0	0.3	-2.1	-0.7
resource units			4,293		216		58,770		-92	

Table 4.2.5: Gross annual benefits (ecosystem services and aesthetic/real estate enhancements) provided by native ash (*Fraxinus* spp.) street trees in fourteen Virginia municipalities based on analysis of field inventory data using i-Tree Streets assessment software. Values calculated from complete inventories do not have a standard error.

	Total Annual Benefits		Annual Benefits Per Tree		Annual Benefits Per Capita	
	$	SE	$	SE	$	SE
Abingdon	6,292	–	149.81	–	0.79	–
Arlington	21,694	–	78.89	–	0.10	–
Charlottesville	18,528	–	163.96	–	0.44	–
Fredericksburg	20,009	8,840	77.86	34.40	0.86	038
Harrisonburg	11,472	5,340	173.82	80.91	0.25	0.12
Leesburg	9,733	–	143.13	–	0.24	–
Lexington	2,657	–	166.07	–	0.39	–
Lynchburg	43,449	12,038	88.67	24.57	0.59	0.16
Martinsville	123	81	7.24	4.78	0.01	0.01
Radford	11,605	3,868	72.53	24.17	0.72	0.24
Richmond	207,046	59,281	146.12	41.84	1.01	0.29
Roanoke	139,078	75,596	125.07	67.98	1.47	0.80
Williamsburg	1,779	1,659	222.38	207.34	0.14	0.13
Winchester	41,547	13,582	80.36	26.27	1.58	0.52

4.3 Potential Impacts of Native Ash Loss on Street Tree Population

By analyzing the native ash trees located in the street tree population, potential environmental and economic impacts of EAB could be quantified. It is estimated that these fourteen municipalities could lose a total of 4,558 native ash trees should there be an EAB outbreak and no intervention performed. Losing these trees means losing a range of functional benefits and their associated monetary values for the municipalities.

On average, each locality would expend an additional 181 gigajoules of energy (valued at $3,900) annually in the absence of native ash street trees for each municipality (see Table 4.2.5). Monetary loss for annual energy conservation ranges from $20 in Martinsville to $20,400 in Richmond. In total, loss of native ash street trees to EAB would result in an additional 2,500 gigajoules of annual energy consumption (valued at about $56,000).

Should native ash street trees be suddenly lost to EAB outbreak, these municipalities will also face managing an additional 80,000 cubic meters of non-intercepted rainfall annually (valued at about $209,000) (Table 4.2.4). This means each municipality will need to prepare for, on average, over 5,700 cubic meters of stormwater each year. Roanoke could face the largest increase in runoff (over 31,000 cubic meters) and Martinsville the least (nine cubic meters). The economic impact from loss of stormwater mitigation benefits ranges from $20 per year in Martinsville to over $71,200 in Richmond.

The loss of native ash street trees would result in reduced accumulations of carbon and diminished aesthetic benefits to real estate value. In total, native ash street trees sequester nearly 990,000 kilograms of carbon each year. This equates to an average of 70,000 kilograms per municipality (valued at $1,100) per municipality, or over $16,000 in total. Richmond would lose the highest amount of carbon sequestration (376,000 kilograms) and Martinsville would lose the least amount (297 kilograms). On average, each municipality could lose up to $21,000 in aesthetic benefits. Richmond has the greatest estimated impact of aesthetic damage, valued at over $122,000. Martinsville has the least estimated aesthetic damage, valued near $100. In total, aesthetic benefits could be reduced by about $296,000 across all municipalities.

Summed across all functional benefit types, each native ash street tree in the studied localities provides about $121 in gross annual benefits on average (Table 4.2.5). Should EAB extirpate all native ashes, these localities stand to lose over $535,000 in gross annual benefits, or roughly $38,200 per locality (Table 4.2.5). Richmond faces the largest loss at over $207,000 in annual benefits while Martinsville will lose just under $125 per year in benefits. On average, each municipality could faces losses close to $38,200 in annual benefits.

In addition to negatively impacting functional benefits of street trees, EAB would also deplete native ash trees as structural assets in the municipal infrastructure. One critical aspect of this infrastructure is the carbon stored in native ash street trees. Nearly 17 million kilograms of stored carbon, valued at over $276,000, could be lost from these ash trees should they be removed and the wood waste not utilized in wood products or by similar carbon-securing means. Potential losses of stored carbon range from about 750 kilograms in Martinsville (valued at $120) to over 7.1 million kilograms in Roanoke (valued at about $117,000). In addition, the replacement of these lost street trees would be vital to restoring green infrastructure and its associated functional benefits. Replacement values of native ash trees, calculated from Streets, range from about $4,600 in Martinsville to over $9 million in Richmond. On average, each municipality would need to replace around $1.2 million in street trees, totaling across all municipalities nearly $17.7 million dollars.

4.4 Potential Costs of Removing and Replacing Native Ash Trees

Under a worst-case scenario, municipalities would undertake no interventions to control an EAB outbreak and all native ash street trees would eventually succumb to the pest. Once dead, these trees could become hazardous to vehicular and pedestrian traffic. Utility lines could also be damaged by falling dead debris, causing power outages and risking public safety. These potential consequences would compel municipalities to remove dead trees or risk substantial liabilities. Localities might also be compelled to proactively remove native ash street trees in order to slow the spread of EAB or to get a head start on reforestation with non-ash species. As such, a worst-case scenario might see the complete removal of native ash street trees from localities within a single year following outbreak. In the fourteen studied localities, it was estimated that there are a total of 4,558 native ash street trees. Using itemized tree removal costs from Virginia Department of Transportation contractors for fiscal year 2011, it was estimated that the total cost of removing all native ash trees in the studied localities would be nearly $1.75 million (Table 4.4.1), averaging $125,000 for each municipality.

Roanoke would potentially incur the largest expense (over $705,000) and Martinsville the least expense (nearly $790). Given that the structural replacement value of native ash street trees in these studied localities was estimated at $17.7 million (see Table 4.2.2), the total mitigation costs for native ash street tree removal and replacement (assuming replanting at a density that would instantly replace lost canopy cover) would therefore be $19.45 million. Added to this cost of mitigating structural losses would be the value of carbon dioxide stored in native ash trees. Assuming that removed trees were incinerated or chipped (thus resulting in oxidation of carbon stored in their wood), the lost carbon value would be about $277,000 (see Table 4.2.2). This would bring total losses in structural value as well as mitigation costs to $19.72 million. In addition to these costs related to the structural assets, the studied localities would also be facing a near-term loss in annual functional benefits totaling about $535,000 (see Table 4.2.5). As a result, the studied localities – should they incur the cost of removing and replacing all native ash trees within a single year – would experience a gross financial impact of about $20.26 million due to invasion by EAB.

Table 4.4.1: Estimated removal cost of native ash (*Fraxinus* spp.) street trees in fourteen Virginia municipalities. Removal costs are based on statewide average contractor fee charged per tree by diameter class to Virginia Department of Transportation in fiscal year 2011.

	Tree Removal Cost by Trunk Diameter Class ($)						
Municipality	0–15 cm[a]	15–30 cm	30–46 cm	46–61 cm	61–76 cm	> 76 cm[b]	Total
Abingdon	784	98	222	352	3,570	23,358	28,384
Arlington	9,604	4,704	3,996	2,816	1,190	4,122	26,432
Charlottesville	490	3,234	5,328	7,744	7,735	15,114	39,645
Fredericksburg	9,016	2,058	0	10,912	5,950	13,740	41,676
Harrisonburg	0	2,450	1,776	2,816	0	34,350	41,392
Leesburg	490	3,626	3,108	352	595	6,870	15,041
Lexington	98	490	444	352	1,785	4,122	7,291
Lynchburg	4,900	20,776	9,990	15,840	26,775	60,456	138,737
Martinsville	784	0	0	0	0	0	784
Radford	2,058	4,900	9,324	2,816	10,115	0	29,213
Richmond	6,321	25,284	85,914	81,664	122,570	248,694	570,447
Roanoke	15,876	17,836	8,880	7,040	71,995	583,950	705,577
Williamsburg	0	0	0	2,816	0	0	2,816
Winchester	14,602	9,702	8,880	14,080	5,950	41,220	94,434

[a]Because Virginia Department of Transportation does not contract per-tree removals under 15 cm, the cost used for this class was half of the per-tree cost of the 15–30 cm class.

[b]Median contractor fee for all 15-cm diameter classes over 76 cm was used to estimate per-tree removal cost of this class.

CHAPTER 5 – DISCUSSION

5.1 Implications of Native Ash Loss from Emerald Ash Borer

Exotic pests are appearing more rapidly in areas of the United States and becoming a nuisance, both in the rural and urban environment. Aukema et al. (2010) estimates that nearly 2.5 non-indigenous pest are brought into the United States each year. As these non-native pests encroach on native flora, ecosystem relationships are disturbed, benefits are lost, and municipal governments have had to produce additional funding for managing these infestations. Since its introduction, EAB has spread to 17 states and has eliminated over 50 million native ash trees from rural and urban forests (Kovacs et al 2010).

Given its current location in Fairfax County and outside of the city of Winchester, EAB has the potential to spread throughout the state of Virginia. Kovacs et al. (2010) suggests that EAB could be widespread in Virginia by 2015 and in all counties of Virginia by 2019. This suggestion takes into consideration both natural flight patterns and human interaction. Table 4.2.3 also gives insight as to factors that may cause native ash composition in municipalities. Although these analyses were performed on a small portion of the state, and thus no statewide inferences can be made, native ash street trees either naturally occur or are planted in areas that are Tree City USA stewards which have a highly urbanized road system that have a colder average January temperature (see Table 4.2.3). These factors along with Kovacs formulations could show a trend as EAB spreads through the Commonwealth of Virginia.

Relative ash abundance in street tree populations in Virginia seems to be low. On average of the fourteen municipalities in the study, native ash comprises approximately 2% ranging from 0.1%

to 5.8%. This follows with evidence from Forest Inventory Analysis (FIA) data provided by the U. S. Forest Service that relative ash composition is low. This evidence could stem from the random plot generator of FIA data and spatial infrastructure of urban areas (e.g. random plots being centered on roadways or buildings). Comparatively, municipalities in Virginia should not be impacted like Midwestern portions of the United States. Some Midwestern cities and states will need to remove and replace tens of thousands of native ash street trees which can comprise a high percentage of their respective street tree population. Relative ash abundance in street tree populations in mid-western portions of the United States is as high as 36% of the total street tree population (Ball et al 2007).

Virginia's native ash component is not as significant as cities in the Midwest. Sydnor (2007) estimates Ohio's urban forests could lose $7 billion in ecosystem service benefits and removal costs from EAB invasion due to the proportionally higher amounts of native ash trees located in the Midwestern portion of the United States. Some factors that could influence the difference between this relative ash abundance in Virginia and the Midwest are the climactic variations, i.e. milder winters in Virginia, more precipitation during the growing season, and lower likelihood of major storm/wind events, parent soil material, and the greater diversity of tree species available for planting. Many urban tree species are selected because they are able to resist urban stress factors, thrive in depleted or water inundated soil conditions, and have the ability to grow to appreciable size and show positive aesthetic characteristics. Native ash trees are capable of surviving in harsh climate conditions, can grow quickly, withstand urban stressors, produce beautiful winter characteristics, and thrive in adverse soil conditions.

These fourteen street tree populations combine to potentially lose 4,558 native ash street trees. Native ash that will be removed in these municipalities will tally a loss of over $545,000 in annual benefits Richmond losing the majority of this figure. Stormwater management will be a key issue. Richmond has a larger street tree population than Roanoke, yet Roanoke will lose 4,000 extra cubic meters of stormwater abatement benefits; stormwater abatement being the most lost annual resource for these two municipalities.

Stormwater management is also difficult to manipulate. Once the infrastructure has been created, it is complicated to upgrade or remove. If stormwater loads are underestimated, or runoff has increased from urbanization, structures can be inundated and could be compromised. This management system is not easy to retrofit and accurate measurements of potential loads are required. Street trees could act as a buffer for peak flow and help infiltration into the soil and should be taken into consideration for stormwater management for future projects.

The most affected aspect of the municipality, in terms of the constituent of the municipality, is the real estate contribution street trees provide. As these trees are removed, median home values should begin to decline. Consumers value trees, and other shrubbery as well, in landscapes, and will pay a higher home premium to attain this value (Heimlich et al. 2008). As EAB progresses through the state, home prices may see a sharper decline, especially considering the current economic recession.

Although Abingdon and Winchester may lose fewer trees, a higher percentage of the street tree population will need to be replaced. Winchester has the highest reported ash abundance in the studied municipalities. Removing 5.8% of the street tree population could lead to gaps in formal street lawn areas, stumps located near the roadside, and, overall, detracting from the aesthetic value of the municipality. Abingdon could lose 3.5% of its street tree population, some of which are large, mature trees. Nearly 85% of the native ash in Abingdon is larger than 60 centimeters in diameter. Larger diameter trees have the capacity to impose more damage, as failures begin, and will need more monies to remove and dispose of the waste properly.

Richmond and Roanoke will be the most affected municipalities by EAB. However, smaller municipalities will be affected to a lesser extent. Leesburg, Charlottesville, and Lexington are above the average for percent composition in terms of native ash abundance and will need to allow for management of this pest.

Relative Performance Index (RPI) estimates native ash species performed at the same level as a typical municipal street tree, meaning these trees are high in vitality and, essentially, performing well with other trees in the municipality. Given the reported "Importance Values", native ash were found to be in top 5 species, for RPI, for two municipalities. Abingdon and Roanoke both had native ash to be important in terms of leaf area, canopy cover, and stem count. Using Santamour's (1990) rule of 30-20-10 (no proportion of urban forest higher than 30% of one family, 20% of one genera, and 10% of one species), relative importance values can be viewed in the same concept. If one species comprises more than 10% of the importance of the entire population, issues with monocultures could become a factor. Each municipality from this study had at least one *Acer* genus in the top 5 of Importance Values. This could be a concern if Asian long-horned beetle spreads from the Northeast.

As these street trees are removed, municipalities will need to be cautious as to which species will replace native ash street trees. Street tree diversity is a key element to sustaining and managing the urban forest. If a street tree population contains low species richness, widespread damage can occur if an insect or a pathogen, infesting a solitary species or genus, is introduced. Using the urban forest model of species diversity (Santamour 1990), this study could be an opportunity to evaluate urban forests across the state and begin discussing diversity on a street tree level and in places where critical management is needed, i.e. hazardous, dying, dead, or structurally poor trees exist.

Overall, EAB will impact the state, but not to the extent that has occurred in the Midwest. However, as trees begin to die, removal and replacements costs will begin to be revealed. We estimate that well over $2 million will be needed to remove EAB-killed trees properly and an additional $17 million to replenish lost canopy cover. All total, over $20 million will need to be allocated in these studied localities for EAB management and response.

5.2 Consequences for Municipal Budget and Public Safety

Annual monetary benefits will be lost from native ash trees being removed from street tree populations across Virginia. Richmond and Roanoke stand to lose the most in both lost functional benefits and have the largest removal and structural replacement cost. Due diligence would also dictate the municipality to investigate native ash which are present as non-street trees in the urban forest.

Winchester and Abingdon are smaller municipalities, yet, have the highest relative ash abundance. Since these municipalities are small, in population, local government may become strained from removing, replacing, and handling the lost portion of their street tree population. Richmond and Roanoke will be dealing with a similar situation. These municipalities are large and have low relative ash abundance; however, the total number of trees that need to be removed, and replaced, will be more than a thousand street trees, per municipality. These municipalities will need to begin strategizing an approach to ensure the safety of the public.

As benefits are lost to EAB, homeowners and municipalities will see reductions in ecosystem service provision. Stormwater runoff will flow on hardscapes at a quicker pace and at higher volumes. This will cause more resources to be needed to channel or divert runoff into detention/retention ponds, facilities to filter and clean the water, or into an adjacent stream/creek/river. Roanoke and Richmond, who have a history of flooding issues, will have to bear the cost of additional water infiltrating their flood water management system during rain events. Energy conservation will decrease with more solar radiation interacting with hardscape surfaces. Summer cooling from canopy shading will be lost and more radiation will enter the road surface; degrading the hardscape at a quicker pace and causing the municipality to repave the road surface more often. The potential for carbon storage will be reduced because overall woody biomass will be subtracted. This could play a role in future carbon sequestration models for the municipalities.

5.3 Response and Recovery from Emerald Ash Borer

In the Midwest, a few communities are trying to be proactive in managing this insect. Many areas are removing this species from their urban forests to slow the spread of the insect (Heimlich et al 2008). At www.emeraldashborer.info, there are photographs that show volunteers stopping tourists at highway rest areas and handing out literature about EAB; even conducting searches for firewood in recreational vehicles and campers. In Virginia, a position has been created within the Department of Forestry that oversees all pertinent information about EAB and its connection to Virginia's forests. There are also signs being posted at National Forest entrances which describe EAB and why firewood needs to remain in the area from where it originated.

As native ash street trees begin to succumb to EAB, municipalities will be tasked with removing these trees and disposing waste properly. Standard USFS protocol, when dealing with an invasive insect infestation, is to quarantine the area so that wood products are not allowed to move outside the area; and dispose of the waste by burning or treating as necessary. In urban areas, handling of these dead trees may become hazardous, due to overhead powerlines, pedestrians, vehicles, and affluent landscapes.

As trees are being removed and disposed of properly, local government will need to begin reviewing tree species that will replace native ash in the public rights of way. Now, with the establishment of DED and EAB, urban forest managers need to realize that planning and planting a variety of tree species is critical in attempting to diversify and effectively manage urban forests. Species richness needs to be reviewed in many localities to discover where diversity may be lacking.

Tree species selection will play a vital role in the structure, function, and value of the urban forest. Streets where power lines reside, trees will need to be of smaller stature. Trees in medians will need to be chosen with no low lying limbs or canopies which touch the ground. Lastly, this could be a chance to use the urban forest management module of 30% of the urban forest not be in the same family, 20% not of the same genus, and 10% not of the same species.

5.4 Future Work and Conclusions

Assessing the structure, function, value, and management needs of urban forests are in high demand. Urban forest assessments are helpful by providing information on management issues and value on urban trees, which municipalities can use to try and acquire more funding. These inventories can be altered to suit the goals of the municipality, e.g. improper pruning performed, pest issues, hazard/health assessment, sidewalk damage from lifting roots, etc., and can provide a multitude of benefits to the managers of the urban forest.

As information gathering and storing technology improves, these inventories should become more readily available to smaller municipalities. Many urban forest assessments can be performed by pencil and paper, later loaded into a computer database; a simple inventory could be performed with pencil, paper, and a diameter tape or Biltmore stick.

As more assessments are performed, inventory programming should become efficient and more accurate in estimating structure, function, and value. As empirical models become more precise, emphasis can be placed on the statistical significance. These inventories should lead to better protocols for inventorying trees and streamline the process.

As this portion of the project is being finished, the second portion will begin. Several more inventories will be completed and municipalities will still be contacted to determine if additional street tree inventories exist. This next portion of the project will look more in detail on existing inventories within this study and check for errors in current datasets. The project will then be reassessed and a second graduate student will begin to decipher the inner workings of i-Tree Streets and determine if the statistical models are accurate in determining overall street tree populations from the random samples taken.

Anecdotal evidence suggests that native ash trees are low in relative abundance throughout the Commonwealth of Virginia. In the fourteen studied municipalities, relative native ash street tree composition ranges from 0.1% to 5.8%, averaging approximately 2% of the total street tree population. This constitutes about 4,500 street trees within the Commonwealth. These street trees provide ecosystem service benefits that allow for energy conservation, stormwater mitigation, carbon storage, carbon sequestration, and improve aesthetic and real estate value. These trees account for approximately $535,000 in annual functional benefits, $277,000 in structural benefits, and have a replacement cost of over $17 million; there could be a total loss of over $20 million to these select municipalities in Virginia.

This study also found through Importance Values and other data not shown, that municipalities with high abundance of *Acer* genera, Richmond, Roanoke, Fredericksburg, and Winchester, in their street tree populations will need to start preparing plans for Asian long-horned beetle (*Anolophora glabripennis* Motschulsky) spread. Also, through the use of pairwise correlation, a relationship was evident that native ash populations may be more abundant in municipalities that are Tree City USA with a highly urbanized transportation system that have a lower average January temperature.

As our economy expands and becomes more global, diligence is key to lessening the odds of incidental introductions of non-native pests. A lack of species diversity within the urban forest has shown, repeatedly, that proactive management and planting the right tree in the right place is paramount in actively managing the differing populations of urban trees. City planners and other professionals in the Green Industry could use this study to better understand the complex environment of the urban forest and the other tangible benefits that arise from trees in urban areas.

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APPENDIX A

Appendix A: Tree attributes and value ranges employed in sample street tree inventories conducted in six Virginia localities during 2008–2010.

Trunk Diameter Class	
0-3”	3-6”
6-12”	12-18”
18-24”	24-30”
30-36”	36-42”
>42”	
Structural Condition	
Dead/Dying - Extreme problems	Poor - Major problems
Fair - Minor problems	Good - No apparent problems
Functional Condition (Health)	
Dead/Dying - Extreme problems	Poor - Major problems
Fair - Minor problems	Good - No apparent problems
Prevailing Land Use	
Single family Residential	Multi-family Residential
Small Commercial	Industrial/Institutional/Large Commercial
Park/Vacant/Other	
Site Type	
Front Yard	Planting Strip
Cutout	Median
Backyard	Forest edge
Other Maintained Location	Other Un-maintained Location
Age Class	
Young	Immature
Mature	Geriatric
Critical Risk Assessment	
Yes	No
Critical Health Assessment	
Yes	No