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# Evaluation of potential strategies to SLow Ash Mortality (SLAM) caused by emerald ash borer (*Agrilus planipennis*): SLAM in an urban forest

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Emerald ash borer, *Agrilus planipennis* Fairmaire (Coleoptera: Buprestidae), an invasive pest native to Asia, has killed millions of ash (*Fraxinus* spp.) trees in North America since it was first discovered there in 2002. As of autumn 2011, *A. planipennis* has been detected in 15 US states and two Canadian provinces. A pilot project to slow the onset and progression of ash mortality, termed SLAM (SLow Ash Mortality), has been implemented in localized *A. planipennis* populations. Here we use spatially explicit simulations to evaluate the potential of a recently developed systemic insecticide to protect the ash resource in urban forests as a component of the SLAM approach. Over a 10-year horizon, simulations showed ash survival varied depending on: (i) how soon insecticide treatment began after the *A. planipennis* introduction; (ii) the proportion of trees treated; and (iii) the distribution of treated trees relative to the *A. planipennis* introduction point. Annual treatment of 20% of ash trees annually protected 99% of trees after 10 years, and the cumulative costs of treatment were substantially lower than costs of removing dead or severely declining ash trees.

**Keywords:** ash tree injection; economic costs; emamectin benzoate; *Fraxinus* spp.; invasive forest pest

## 1. Introduction

More than 450 species of non-native forest insects have become established in the USA and Canada and at least 14% of these species are considered invasive and damaging (Langor et al. 2009; Aukema et al. 2010). Invasive insects can alter ecological processes in forests and urban forests, reducing biodiversity, and regulations associated with these pests can be costly for plant-related industries, for example, nurseries and sawmills (Wilcove et al. 1998; Nowak et al. 2001; Aukema et al. 2010). Many non-native forest insects initially become established in urban areas following importation of infested wood as packing material, nursery plants, produce or other commodities (Work et al. 2005; Poland and McCullough 2006; Westfall et al. 2008). Despite increasing awareness of the impacts of invasive forest pests and regulatory efforts, non-native, phytophagous insects continue to be introduced into new habitats through international trade and travel (Work et al. 2005; Liebhold et al. 2006; McCullough et al. 2006).

Emerald ash borer (*Agrilus planipennis* Fairmaire) (Coleoptera: Buprestidae), a phloem-feeding beetle discovered in 2002 in Detroit, Michigan, and Windsor, Ontario (Cappaert et al. 2005), has become the most destructive forest insect ever to invade the USA. In its native region, Asia, *A. planipennis* is a secondary pest, colonizing severely stressed or dying ash (*Fraxinus* spp.) trees (Yu 1992). In North America, *A. planipennis* will preferentially colonize stressed ash trees (McCullough et al. 2009a, 2009b), but healthy trees growing under optimal conditions are also attacked and killed (Cappaert et al. 2005; Rebek et al. 2008; Limback et al. 2010). To date, *A. planipennis* populations have been found in at least 15 states in the USA and the Canadian provinces of Ontario and Quebec (EAB.info 2011). Tens of millions of ash trees, ranging from 2.5 cm to 1.5 m in diameter, have been killed in forest, riparian, and urban areas.

### 1.1. *Biology of* A. planipennis

Ash trees are injured by *A. planipennis* larvae feeding in serpentine galleries in the plant's cambium and phloem. Larval galleries also typically score the outer sapwood, which can be particularly damaging to ring-porous species such as ash. A low-density of *A. planipennis* larvae generally has little effect on the overall health of the tree, in part because ash trees are highly sectorial (Tanis et al. forthcoming 2012) and relatively efficient at vertical translocation of nutrients and water. As larval densities build, however, more tissue is damaged, translocation is disrupted, canopies thin, branches die and eventually the tree succumbs. In trees stressed by moderate to high densities of *A.*

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*planipennis*, girdling or other injuries, most larvae overwinter as prepupae in chambers approximately 1 cm deep in the sapwood or in the outer bark. Pupation occurs in spring and adult emergence begins in May or June (Cappaert et al. 2005). In relatively healthy trees, however, a high proportion of larvae overwinter as early instars, feed for a second summer, then overwinter as prepupae, completing a two-year life-cycle (Tluczek et al. 2011).

Adult *A. planipennis* beetles feed on the margins of ash leaves throughout their three- to six-week life-span, but cause negligible damage to trees. Maturation feeding on foliage occurs for approximately a week before mating begins and females feed for an additional two weeks before oviposition begins (Cappaert et al. 2005). Host preference of adult *A. planipennis* varies among North American ash species. For example, when green ash (*F. pennsylvanica*) and white ash (*F. americana*) co-occur, green ash is preferentially colonized and trees succumb sooner than those of white ash (Anulewicz et al. 2007, 2008; Limback 2010). Host-range tests are ongoing (Anulewicz et al. 2011), but preliminary results suggest that at least 16 North American ash species and several European ash species are potentially vulnerable to this invader.

### *1.2. Dispersal and detection of* A. planipennis

Adult *A. planipennis* beetles are agile fliers and, in laboratory tests, mature females were physiologically capable of flying at least 3 km (Taylor et al. 2007). Dispersal studies conducted by felling and de-barking of ash trees in areas with known introduction points have shown that most eggs are laid on ash trees within 100 m of the point where adult females emerged (Mercader et al. 2009; Siegert et al. 2010). A small fraction of larvae, however, has been observed in trees more than 700 m from the emergence point of the adults (Mercader et al. 2009; Siegert et al. 2010). This natural dispersal contributes to the spread of established populations, but factors that trigger dispersal and affect the proportion of females that engage in long-distance flights remain unknown.

Artificial dispersal of *A. planipennis* occurs when humans inadvertently move infested ash trees, logs or firewood into new areas. Beetles can emerge from ash logs or firewood for up to a year, and occasionally longer, after infested trees are cut (Petrice and Haack 2006, 2007). In numerous cases, localized outlier populations were traced back to shipments of infested ash nursery trees, sawn logs or firewood (Cappaert et al. 2005; Poland and McCullough 2006). Several of these outliers became established before *A. planipennis* was identified in 2002. Ash trees are common in forest, rural and urban areas across much of the northeastern USA, which makes it likely that emerged beetles will find a suitable host. Over time, the initially isolated outlier populations build and eventually coalesce with each other and the main invasion front, substantially increasing the spread rate (Shigesada and Kawaski 1997). Results from a large-scale dendrochronological study showed that ash trees were being killed by *A. planipennis* in the Detroit, Michigan, area as early as 1998 (Siegert et al. 2007). Given that *A. planipennis* populations require at least three to four years to build to densities high enough to cause mortality, there is little doubt that *A. planipennis* was established in southeast Michigan, at least by the early 1990s (Siegert et al. 2007).

Detection of newly established *A. planipennis* populations is exceedingly difficult because trees with low densities of *A. planipennis* exhibit few, if any, external symptoms. External evidence of infestation such as canopy decline, bark cracks and epicormic shoots become apparent only after populations have built to moderate or high densities. Small, D-shaped holes left by emerging adults or larger holes left by woodpeckers preying on overwintering larvae are often the first signs of infestation. Like North American *Agrilus* spp. (Anderson 1944; Haack and Benjamin 1982), *A planipennis* beetles often initially colonize branches in the upper canopy of all but the smallest trees (Cappaert et al. 2005; McCullough and Siegert 2007b), making it difficult to observe beetle exit holes or holes left by woodpeckers preying on larvae.

Adult *A. planipennis* rely on host volatiles and visual cues to locate and identify ash trees but do not produce long-distance sex or attraction pheromones (Crook and Mastro 2010). Considerable effort has been devoted to the development of artificial traps and lures that emit host volatiles (McCullough and Poland 2009; Crook and Mastro 2010; Francese et al. 2010; Grant et al. 2010; Poland et al. 2011). These lures, however, must compete with volatiles emitted by ash trees, and are not nearly as effective as pheromone traps, such as those used for gypsy moth (*Lymantria dispar* L.) detection. Ash trees girdled in spring are highly attractive to ovipositing *A. planipennis* (McCullough et al. 2009a, 2009b; Tluczek et al. 2011) and grids of girdled trees have been used for the detection of *A. planipennis* infestations (Rauscher 2006; Hunt 2007; Poland and McCullough 2010; SLAMEAB.info 2011). Girdled trees need to be debarked in winter to determine whether *A. planipennis* larvae are present, which is a labor-intensive process, and suitable or accessible trees for girdling are not always available. To date, most *A. planipennis* outlier populations have been discovered at least three to four years after establishment, typically when trees begin to exhibit external signs of infestation.

Early identification of newly founded, low-density *A. planipennis* populations could provide foresters, municipal arborists, and property owners with time to develop and implement strategies to manage their ash resource. Since 2004, the United StatesDepartment of Agriculture (USDA) Animal and Plant Health

Inspection Service (APHIS) and cooperating states have allocated millions of dollars annually for *A planipennis* detection. Early programs relied on visual surveys to identify symptomatic trees, while subsequent efforts in Michigan, Ohio and Indiana employed grids of girdled ash trees (Rauscher 2006; Hunt 2007). Current efforts in the USA include deployment of thousands of artificial traps baited with host volatiles (Crook and Mastro 2010; USDA APHIS 2010), visual surveys of ash trees in high-risk sites (e.g., camping grounds), and outreach activities to increase public awareness of *A. planipennis*.

When a new *A. planipennis* population is detected, quarantines are imposed that prohibit or regulate transport of ash trees, logs, firewood and related materials, reducing the likelihood that infested ash material will be transported into new areas. Residents, municipal authorities and foresters, however, are left to cope with *A. planipennis* on their own. Costs of replacing or treating landscape ash trees in urban and suburban areas of the USA were projected to exceed US $10–20 billion between 2009 and 2019 (Kovacs et al. 2010). Widespread ash mortality in forested, rural and riparian areas is expected to have multiple cascading effects through ecosystems, potentially affecting numerous ecosystem services as well as the cultural traditions of several American Indian tribes in the USA and Canada (Cappaert et al. 2005; Poland and McCullough 2006). Without a different approach, this scenario will be repeated with every new *A. planipennis* infestation discovered.

## 2. SLAM – SLowing Ash Mortality

### 2.1. Goals and strategies of SLAM

Recent work has shown that slowing the growth and spread of *A. planipennis* outlier populations, particularly those located near urban areas and distant to the primary infestation, can save or delay the spending of millions of dollars in economic costs (Kovacs et al. 2011). A pilot project, referred to as SLAM (SL.ow A.sh M.ortality), was initiated in 2008 to develop, implement and evaluate an integrated strategy for localized, recently established *A. planipennis* outlier sites. The goal of the SLAM pilot project is to slow the onset and progression of ash mortality by slowing the growth of *A. planipennis* populations. Ideally, management activities will also help to slow the natural spread of *A. planipennis*. Given the difficulty of detecting and monitoring low density *A. planipennis* populations, however, and the serious ecological and economic impacts of ash mortality, the focus of SLAM is on the resource, that is, the ash trees within and around the infested area.

The SLAM pilot project was implemented in three areas of the Upper Peninsula of Michigan, (SLAMEAB.info 2011) where localized *A. planipennis* infestations were identified in 2007–2008. Personnel from two universities, two state and two federal agencies are involved in various aspects of the SLAM pilot project. All three project areas consist of heterogeneous landscapes that include small cities, rural areas, forests and swamps.

### 2.2. Management options in outlier sites

Management tools employed in the SLAM pilot project include girdled ash trees, a highly effective systemic insecticide, and harvest or removal of ash trees. In the pilot project, girdled ash trees play two roles. First, grids of girdled ash trees are debarked in autumn to assess *A. planipennis* distribution, as well as larval density and development rates. Artificial traps to monitor *A. planipennis* distribution supplement the grid of girdled trees in areas where ash trees are not present or accessible for girdling. While this intensive level of sampling is unlikely to be used in operational SLAM projects, results are substantially increasing our understanding of *A. planipennis* dynamics. Second, clusters of two to four girdled ash trees are established in selected areas to function as *A. planipennis* population "sinks" and reduce *A. planipennis* population growth. The stressed trees attract ovipositing female beetles, but the trees are destroyed in winter before the larvae can complete development.

A recently developed systemic insecticide, emamectin benzoate (sold as TREE-age®), is also employed in the SLAM pilot project. Unlike other insecticides which must be applied annually, this product provided at least two years of nearly 100% control of *A. planipennis* in large, replicated studies (Herms et al. 2009; Smitley et al. 2010; McCullough et al. 2011; D.G.M., unpublished data). The insecticide is injected into the base of the trunk of ash trees in spring, then translocated via the xylem to canopy branches and leaves, controlling both adult and larval *A. planipennis*.

A third management option is simply to harvest or fell ash trees to reduce the amount of phloem available to developing *A. planipennis* larvae in the SLAM project areas (McCullough and Siegert 2007a). Property owners may be able to sell merchandable white ash trees (e.g., $>25$ cm diameter) to timber buyers, or the ash trees can be used for firewood. Concerns have arisen as to whether removing a large proportion of ash trees in an area could increase spread rates of *A. planipennis*, either because beetles fly further to find suitable hosts or because of human transport of infested ash material. Outreach activities to inform residents and visitors to the area and regulatory efforts to reduce the risk of infested ash being transported into or out of the area are also incorporated into the SLAM pilot project.

Empirically-based models to project *A. planipennis* population growth, spread and ash mortality were developed to evaluate effects of activities in the SLAM pilot project areas (Mercader et al. 2011a, 2011b). Results from simulations have shown that treating ash trees with the emamectin benzoate insecticide is the single most effective option for slowing *A. planipennis* population growth and the progression of ash mortality. These simulations may, however, have underestimated the potential effectiveness of the systemic insecticide because they incorporated larval mortality on treated trees but did not account for mortality of adult beetles that consume leaves on treated trees. Insecticide treatment is also the only option that retains ash trees, including valuable landscape ash, in the project area. Establishing clusters of girdled trees is relatively inexpensive in forested areas and can slow *A. planipennis* population growth, but this option may not be practical in urban areas. Harvesting merchantable ash trees may provide landowners with some financial gain while limiting potential *A. planipennis* production (McCullough and Siegert 2007a). Sanitation to remove low vigour or hazardous ash trees can also be useful, especially in urban areas. Simply felling or removing ash trees to reduce the amount of ash phloem available for larvae, however, has the least effect on *A. planipennis* population growth and, if used exclusively, could increase spread rates (Mercader et al. 2011a, 2011b). Ideally, all management options should be integrated into a comprehensive SLAM strategy developed for individual outlier sites.

The outlier infestations in the ongoing SLAM pilot project in Michigan encompass a substantial amount of forested land, but implementing a SLAM project may be most efficient in developed, urban areas. Injecting landscape trees with emamectin benzoate – the option exerting the greatest constraint upon on *A. planipennis* population growth – is generally faster and less costly in developed areas where most trees are accessible to applicators than in forested or unmanaged landscapes where trees may be some distance from roads. In addition, municipalities often have up-to-date inventories of trees on public property, which simplifies cost estimates and logistical issues.

Obviously, treating all of the ash trees in a given area with emamectin benzoate in alternate years would provide nearly 100% control of *A. planipennis* over time. There will be few situations, however, where resources are adequate to permit all ash trees in an infested area to be treated at two-year, or even three-year, intervals. Municipal foresters and *A. planipennis* program managers must, therefore, determine how to best allocate available funds and labor.

## 3. Simulations of insecticide applications in urban areas

### *3.1. Modeling approach*

We used a modified version of a spatially explicit, coupled-lattice simulation model (Mercader et al. 2011a, 2011b) to simulate the localized growth and spread of *A. planipennis*. In brief, the model couples relevant population processes onto matrices that record the quantity of available ash phloem and the number of *A. planipennis* larvae that develop in one or two years. Population processes linking these matrices include: (1) the number of adult beetles emerging in a given year, (2) dispersal of adult beetles, (3) population growth, and (4) the loss of ash phloem due to feeding by developing larvae. Variables were identified and parameterized using data collected from multiple field sites (Mercader et al. 2011a).

Emergence of adult *A. planipennis* beetles in a given year reflects the number of larvae that develop in 1 or 2 years from eggs laid in the previous 1 and 2 years, respectively. We assumed a 50:50 sex ratio for beetles, consistent with many observations at varying *A planipennis* densities (Cappaert et al. 2005; D.G.M., unpublished data). We further assumed that all females would be mated, and each female could potentially lay up to 20 eggs that would successfully develop into adults (i.e., approximately a 10-fold growth rate). Adult females in our simulations engage in a series of 10 dispersal trials, where each female will lay two eggs on each ash tree encountered. Females may encounter the same tree multiple times. Movement of females among trees was based on the distribution of larvae observed in two isolated infestations in Michigan derived from a single generation of *A. planipennis* (Mercader et al. 2009).

We modified the simulation model by altering the dispersal mechanism to allow for movement of individual females among trees during the same year. This enabled us to simulate adult mortality caused by beetles feeding on foliage of trees treated with the emamectin benzoate insecticide. In addition, by simulating our environment in terms of individual trees rather than patches of trees, we were able to simulate the decline of individual trees and costs of treatment. In previous simulations, we used patches of trees rather than individual trees and this coarser scale allowed environments to be greater than the maximum *A. planipennis* dispersal distance observed in simulations (Mercader et al. 2011a, 2011b). To maintain the spatial resolution required, however, environments were smaller than the potential maximum dispersal of adult beetles over the 10-year period. To simplify interpretation of results, we assumed that the environments used in our simulations were isolated from other environments containing ash trees and that adult beetles did not exit from our environment. Although it is likely that beetles may exit the environment despite the absence of ash trees, the proportion which would do so remains unknown.

### *3.2. Environments used in simulations*

Environments used in these simulations represent a neighbourhood consisting of 320 identical suburban blocks, with blocks arranged in a 20 × 16 grid. Blocks were separated by streets, each 10 m wide, and each block contained a total of ten lots arranged as in Figure 1.

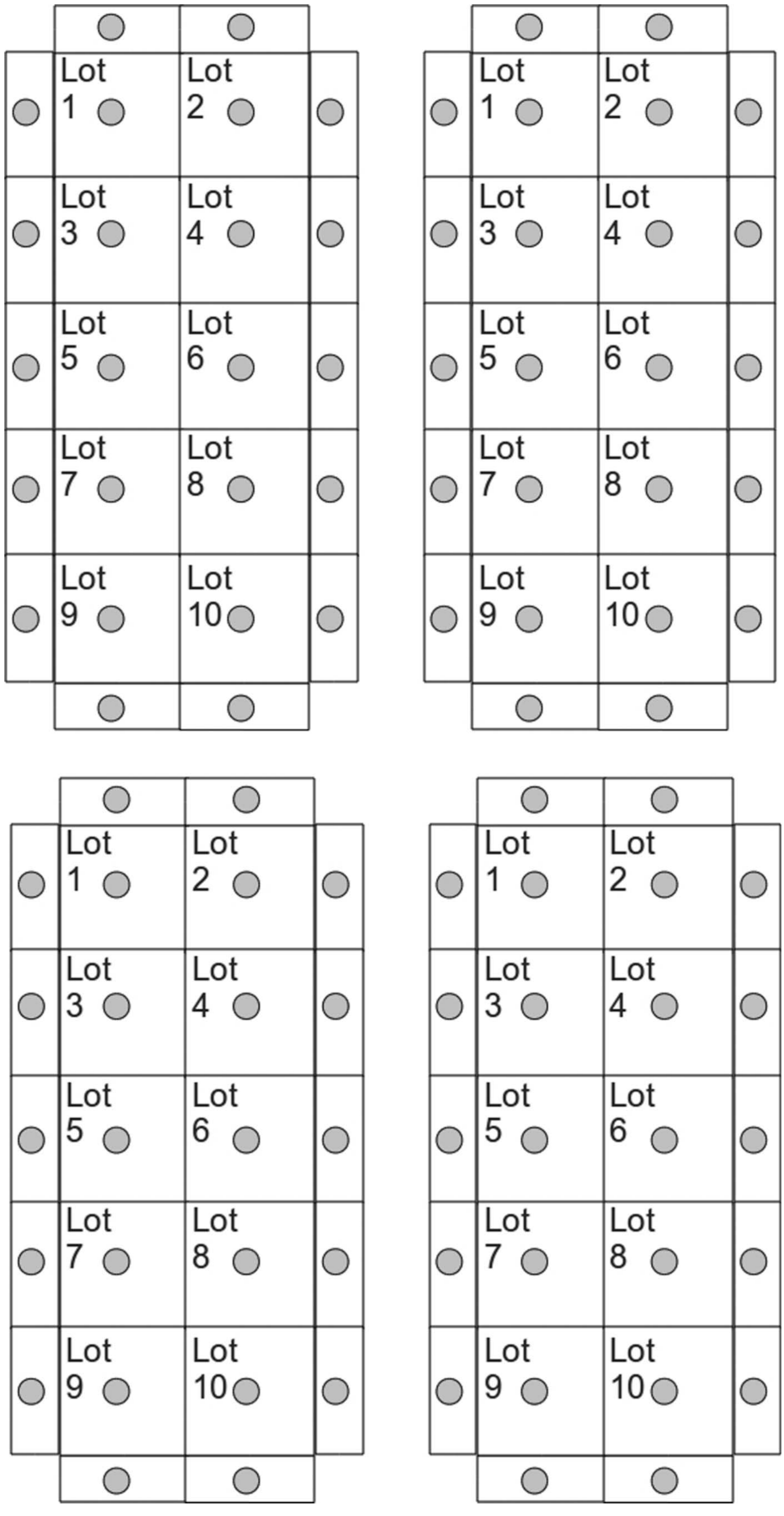


Figure 1. Example of the organization of four of the 320 blocks in the hypothetical environments used for simulations. Grey circles (24 per block) represent locations where trees could occur in residential lots in each block. On average, there were approximately seven ash trees per block and a total of 2314 ash trees within the entire environment used in our simulations.

Each lot consisted of a 20 × 20 m area representing a house and backyard (e.g., private property) and a 20 × 10 m area representing the front yard and boulevard (e.g., municipal property). A maximum of two trees, one in the backyard and one in the front lawn or boulevard area, could occur on each lot. Whether an ash tree was present in either potential location for each lot was determined by a single Bernoulli trial with a probability of 0.3, resulting in a total of 2314 ash trees. This environment would be similar to many municipalities where ash comprises up to 20–30% of the urban forest canopy (MacFarlane and Meyer 2005; Saint Paul, MN EAB 2011) and ensured that we had adequate data for our simulations. The surface area of ash phloem available for each tree was subsequently determined from a right-skewed distribution with a median diameter at breast height (DBH; measured 1.3 m above ground) of 40 cm (16 in) and an interquartile range of 28–51 cm (11–20 in), with a maximum DBH of 76 cm (30 in) and minimum of 15 cm (6 in) (Figure 2). Trees in our simulations were assumed to grow or repair damage caused by *A. planipennis* larvae at an annual rate equivalent to a 1% increase in surface area. A total of 200 environments were constructed and all simulations were run in the same 200 environments. Results represent mean values derived from 200 simulations conducted for each scenario.

We modeled each tree explicitly, simulating the emamectin benzoate application on a tree-by-tree basis. We assumed the insecticide provided complete control of *A. planipennis* for a 2-year period, which is consistent with field trial results (Smitley et al. 2010; McCullough et al. 2011; D.G.M., unpublished data). If in the course of the dispersal trials an adult female beetle encountered a treated tree, we assumed she would feed on the foliage and die, and that no larvae would be produced on that tree.

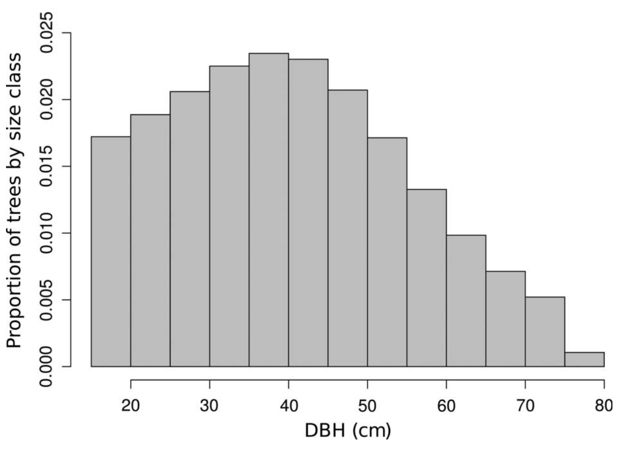


Figure 2. Frequency distribution of ash trees by DBH (diameter measured at breast height, 1.3 m above ground) in the simulated environment.

### 3.3. *Scenarios explored*

*Scenarios with no insecticide:* We assumed 400 *A. planipennis* adults emerged from ash firewood piled in a lot in the center of the environment for all scenarios. In the baseline scenario, we assumed no trees were treated with insecticide or removed. In the next scenario, we assumed the same conditions except that trees were removed once 60% of the phloem in a tree was consumed by *A. planipennis* larvae. At this point, canopy decline, dieback and external symptoms of *A. planipennis* infestation would likely be evident (Anulewicz et al. 2007). Declining landscape trees are unattractive and can pose a hazard to people, property and vehicles. Subsequent scenarios followed these same initial conditions, except that the emamectin benzoate insecticide was employed.

*Scenarios with insecticide applications:* The first set of scenarios incorporating the use of the insecticide considered the application of insecticide to either 10%, 20%, 30%, 40%, or 50% of all ash trees present at the start of the simulation on a yearly basis. In any given year, trees to be treated with the insecticide were selected at random from the trees that had not been treated in two or more years. We assumed that the *A. planipennis* infestation was either detected with artificial traps in summer when *A. planipennis* adults are active or in the autumn when detection trees were sampled. Therefore, the earliest the insecticide could be applied would be the spring following the initial detection. In our simulations, insecticide applications began either the spring following the initial infestation (e.g., one year post-introduction) or four years after the initial introduction.

In the second set of scenarios, we compared results of targeting ash trees near the introduction point for insecticide treatment versus randomly selecting trees for treatment. In these scenarios, the insecticide was applied to 10% of all ash trees, but applications targeted: (1) ash trees growing in the block where the infestation originated; (2) ash trees within a 1-block radius of the origin of the infestation (9 blocks total); and (3) ash trees within a 2-block radius of the origin of the infestation (25 blocks total). Less than 10% of the ash trees occurred in the targeted areas in these three scenarios. Thus, all ash trees in the targeted areas were treated, then randomly selected trees in the remainder of the environment were treated until the total number of treated trees equalled 10% of all ash trees. On average 9.7%, 7.2%, and 2.2% of ash trees beyond the targeted areas were also treated when the targeted area consisted of (1) the block where the infestation originated, (2) a 1-block radius and (3) a 2-block radius around the origin of the infestation, respectively. Because insecticide treatments remained effective for a 2-year period, in the years when trees within the targeted areas did not require treatment, 10%

of trees beyond the targeted area were treated. As with the first set of scenarios, we present results for simulations in which insecticide applications began either 1 year or 4 years after the *A. planipennis* introduction.

In a third scenario, we assumed privately owned ash trees in the backyards of the lots would not be treated with the insecticide. For this scenario, we again assumed that 10% of all ash trees would be treated, but only trees growing in the front lawns or along the boulevard were eligible for insecticide treatment. For these simulations, we assumed the infestation was detected the year that *A. planipennis* was introduced and insecticide applications began the next year.

### 3.4. Costs of ash tree treatment or removal and replacement

We acquired cost estimates (US$) for ash tree removal and replacement developed in 2010 by arborists or urban foresters for six cities in the Midwestern USA with *A. planipennis* infestations. Estimates included labor, equipment, fuel, and administrative support for felling, hauling and disposal of infested trees and grinding of stumps. Replacement costs included the cost of procuring, planting and mulching young trees ($\leq$6 cm diameter). Estimated treatment costs included the price of the emamectin benzoate insecticide (applied at relatively low label rates), application equipment, labor, fuel and administrative support. Variation in cost estimates among municipalities largely reflects the size, number, distribution and condition of ash trees on municipal property and local labor costs. Cost data were acquired for municipal ash in three cities in Illinois and one city in each of Michigan, Minnesota and Wisconsin. Cost estimates were relatively similar for tree removal and replacement, ranging from $750 to $1172 per tree, with an average cost of $888 $\pm$ 54. Estimated costs for emamectin benzoate treatment were less variable, ranging from $3.03 to $3.62 per 2.5 cm of DBH. To be conservative, we used a cost of $818 per tree for removal and replacement and $3.62 per 2.5 cm DBH for treatment in our simulations. We calculated annual and cumulative costs of treating 0, 10, 20, 30, 40, or 50% of ash trees when infestations were detected either 1 year or 4 years after the *A. planipennis* introduction.

A final set of scenarios was run to contrast the effect of altering the propensity for dispersal by adult female *A. planipennis*. Dispersal in the model is assumed to be $e^{-a*\text{distance}}$, where $a$ is 0.037, based on dispersal estimates from a homogeneous site in Michigan (Mercader et al. 2009). To examine the influence of dispersal on cost estimates, we altered the parameter $a$ from 0.057 to 0.017 (e.g., lower to higher dispersal) in 0.01 intervals in the simulations.

## 4. Results from simulations

### 4.1. Number of surviving ash trees under different scenarios

When no insecticide was used and the *A. planipennis* population was allowed to increase and spread, the first ash trees to exhibit severe decline and require removal (because *A. planipennis* larvae consumed $>$60% of the phloem) appeared in Years 3 and 4 (Figure 3a). As the *A. planipennis* population continued to grow and expand, the rate at which ash trees were lost accelerated. By Yr 7, 58% of the trees were gone and 9 years after the initial *A. planipennis* introduction, 97% of the original 2314 ash trees were gone (Figure 3a). Removing trees as they declined ($>$60% phloem loss) only slightly reduced the overall number of trees lost in any given year compared to the baseline scenario (Figure 3a).

Similarly, when 10% of trees were randomly selected for insecticide treatment, the number of live trees remaining in Yr 10 was comparable, regardless of whether declining trees were removed or not (Figure 3a). Comparisons of subsequent scenarios with and without the removal of declining trees were also

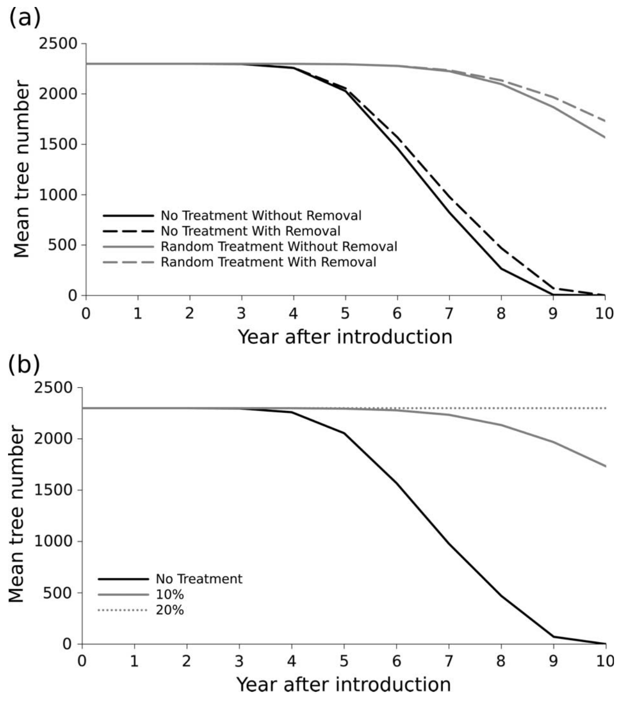


Figure 3. Mean number of live trees remaining over a 10-year period for 200 replicate simulations run with an initial introduction of 400 *A. planipennis* adults. (a) Simulations represent scenarios that differ in two conditions in a factorial design. The first scenario was whether or not a systemic insecticide was applied annually to 10% of randomly selected trees. The second scenario was whether or not trees were removed after *A. planipennis* larvae consumed greater than 60% of the phloem. (b) Simulations represent scenarios in which systemic insecticides were applied annually to 0 (No Treatment), 10%, or 20% of randomly selected trees.

qualitatively identical. To facilitate interpretation of our results, therefore, we present here only results from simulations in which declining trees were removed.

Treating at least 20% of the ash trees annually, instead of 10%, substantially increased the number of trees that survived over the 10-year period (Figure 3(b)). In our simulations, when 20% of the ash trees in the area each year were selected at random and treated with emamectin benzoate, 99.5% of the ash trees remained alive in Yr 10 (Figure 3(b)). Similarly, when 30, 40 or 50% of the trees were randomly selected and treated annually, $<0.5\%$ of trees required removal by the end of the period. In contrast, random selection and treatment of only 10% of the trees annually resulted in the loss of 25% of the ash trees by Yr 10.

When insecticide treatment began 1 year after the *A. planipennis* introduction and only 10% of the ash trees were treated, targeting ash trees growing in the block where the infestation originated (i.e., the block with the hypothetical firewood pile) protected more trees than randomly selecting trees for treatment (Figure 4a). Expanding the targeted area to encompass trees in a 1- or 2-block radius around the origin of the infestation protected even more trees (Figure 4a). Targeting trees within a 2-block radius of the block where *A. planipennis* was originally introduced meant that all ash trees within a 340 × 640 m area were treated in alternate years. Therefore, in any given year, 78% of the ash trees treated with the insecticide would occur within this area. Under this set of conditions, the *A. planipennis* population would be concentrated in trees near the firewood pile. A relatively high proportion of *A. planipennis* would be killed by the insecticide and only a few trees beyond the targeted area would become infested over the 10-year period. Over time, the benefit of focused treatment would, however, likely decline. At least a few surviving beetles would disperse each year and populations would build in more distant trees which were unlikely to be treated. This pattern starts to appear in Years 9 and 10; trees begin to be lost if insecticide treatments continue to be focused on trees within a 1-block radius around the introduction point (Figure 4a).

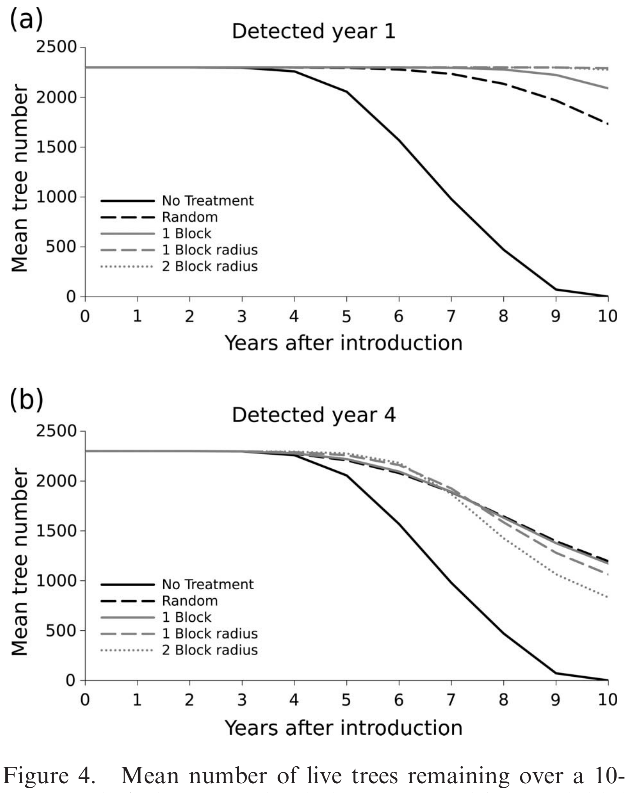


Figure 4. Mean number of live trees remaining over a 10-year period in 200 simulations run with an initial introduction of 400 *A. planipennis* adults. Simulations represent scenarios in which systemic insecticides were not applied (No Treatment) or applied annually to 10% of trees (a) 1 year or (b) 4 years following the *A. planipennis* introduction. Insecticide treatments were either applied to randomly selected trees, or targeted trees in the block where the *A. planipennis* introduction occurred, or trees within a one- or two-block radius of the *A. planipennis* introduction. The same number of trees were treated with insecticide in all scenarios.

When 10% of the ash trees were treated but insecticide applications did not begin until 4 years after the initial *A. planipennis* introduction, targeting treatments yielded different patterns (Figure 4(b)). Randomly selecting trees for treatment and targeting trees within the block where the infestation originated generated very similar results. At the end of the 10-year period, 52% of the ash trees remained when trees were randomly selected, while 51% of the trees remained when treatment was targeted at trees within the block where the infestation originated (Figure 4(b)). This occurred because only 0.3% of all ash trees in the environment were growing within the block where the infestation originated. The other trees that were treated (9.7% of all trees) were randomly selected from the other blocks. In contrast, when trees in either a 1- or 2-block radius around the block where the infestation originated were targeted for treatment, only 46% and 36% of the trees, respectively, remained by Yr 10 (Figure 4(b)). In this situation, some *A. planipennis* beetles would have dispersed annually in the four years before treatment began, infesting trees beyond the area encompassed by the 1- or 2-block radii. Because fewer trees growing beyond the 1- or 2-block radii could be treated, the insecticide had less effect on the *A. planipennis* populations building up in those trees. When the backyard trees, which accounted for 42% of the ash in the environment, were excluded from treatment, the overall effectiveness of the program was slightly reduced (Figure 5), even though the same number of trees were treated under both scenarios. If insecticide treatments were applied the year after the initial introduction, randomly selecting 10% of all trees, including those in backyards, protected 75% of ash trees over the 10-year period. When backyard trees were excluded from treatment, 70% of the ash trees remained in Yr 10 (Figure 5).

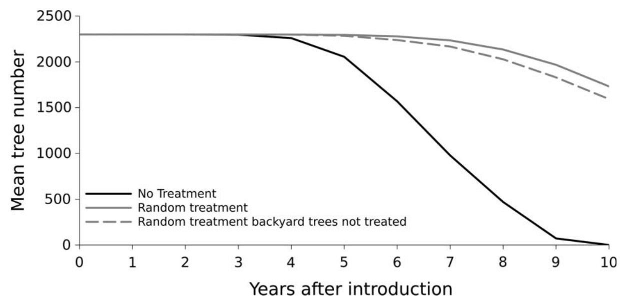


Figure 5. Mean number of live trees remaining over a 10-year period in 200 simulations run with an initial introduction of 400 *A. planipennis* adults. Simulations represented scenarios in which systemic insecticides were either not applied (No Treatment) or were applied annually to 10% of randomly selected trees. Trees treated with insecticide were either distributed across the entire environment or trees located in backyards were excluded from treatment.

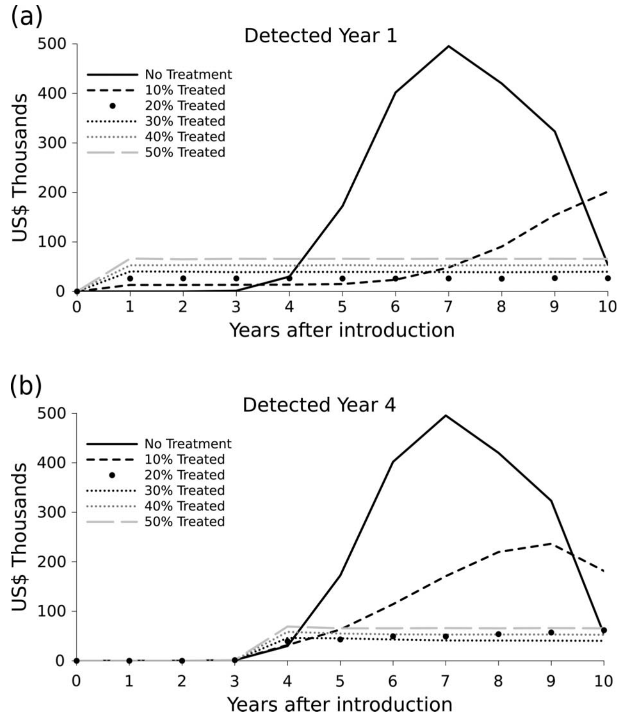


Figure 6. Mean cost per year of treating and removing trees in 200 simulations run over a 10-year period with an initial introduction of 400 *A. planipennis* adults. Simulations represent scenarios in which systemic insecticides were not applied (No Treatment) or were applied annually to 10%, 20%, 30%, 40% or 50% of randomly selected trees. Simulations represent the application of treatments beginning (a) 1 year or (b) 4 years after the *A. planipennis* introduction.

### 4.2. *Costs for ash tree removal or insecticide treatment*

Costs for removing and replacing declining ash trees were dramatically higher than costs associated with treating ash trees with the emamectin benzoate insecticide (Figures 6 and 7). When none of the trees was treated with the insecticide, by Yr 3, on average, only three trees were severely declining and required removal with a total cost of $2438. In Yr 4, removal costs increased to $30,168. Annual costs after Yr 4, however, ranged from $171,077 in Yr 5 to $481,197 in Yr 7, when 588 ash trees required removal. Cumulative costs of removing and replacing ash trees over the 10-year period were $1.9 million (Figure 7).

Costs of treating ash trees with the emamectin benzoate insecticide were substantially lower than costs of tree removal and replacement when no insecticide was used under all scenarios (Figures 6 and 7). This disparity in costs was evident even when 50% of the trees were treated annually (Figures 6 and 7). Given the 2-year efficacy of the insecticide, this would have protected 100% of the trees from *A. planipennis* injury. When 50% of the trees were treated annually, cumulative costs over the 10-year period were $661,160 if treatment began 1 year after *A. planipennis* was introduced (Figure 7(a)) and $468,168 if treatment began 4 years after the introduction (Figure 7(b)).

Treating a lower proportion of the trees, however, was optimal in terms of protecting an acceptably high proportion of the ash trees at the lowest overall cost over the 10-year period (Figures 6 and 7). When 20% of the trees were randomly selected for treatment and treatment began 1 year after the introduction, less than one ash tree on average (0.4 trees) required removal in any year and the cumulative costs over the 10-year period were $265,271 (Figure 7(a)). When 20% of the trees were randomly selected and treatments did not begin until Yr 4, on average a total of 216 ash trees required removal over the 10-year period. Annual costs to remove and replace the 15–127 trees that declined ranged from $12,242 to $130,238 per year (Figure 6(b)). Cumulative costs of removing declining trees while continuing to treat 20% of the remaining ash trees were $364,554 after 10 years (Figure 7(b)). Treating 30% or 40% of the trees annually yielded cumulative costs of $397,206 and $529,778, respectively, when treatments were initiated in Yr 1 (Figure 7(a)), and $304,524 and $381,369, respectively, when treatments were initiated in Yr 4 (Figure 7(b)).

Reducing the propensity for *A. planipennis* dispersal in our simulations resulted in a higher concentration of beetles in trees near the source of the infestation for a longer period of time. When no insecticide was used, the concentration of *A. planipennis* near the origin of the infestation caused fewer trees to decline and to require removal over the 10-year period, which reduced and delayed costs that would be incurred in this scenario (Figure 8). If 20% of trees were randomly selected and treated annually beginning in Yr 1, overall costs increased slightly because fewer trees were protected (Figure 8(a)). When only 10% of trees were treated annually beginning in Yr 1, overall costs were reduced, primarily because fewer trees

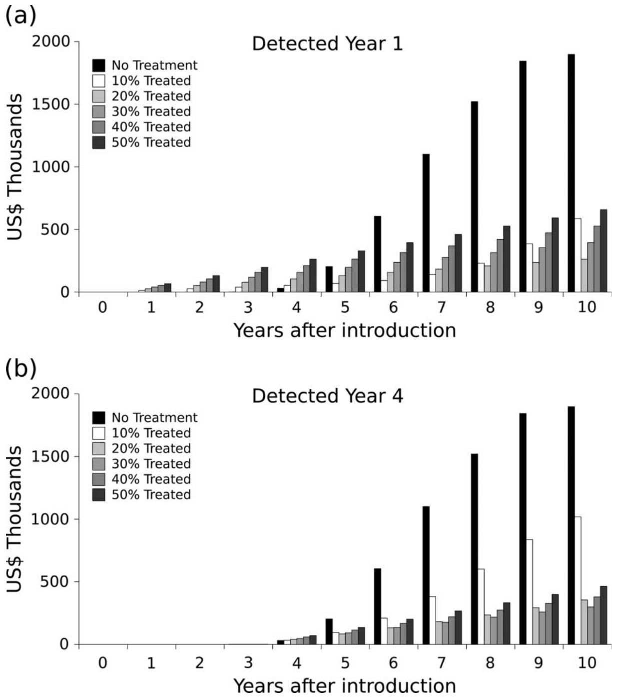


Figure 7. Mean cumulative cost of treating and removing ash trees over a 10-year period in 200 simulations run with an initial introduction of 400 *A. planipennis* adults. Simulations represent scenarios in which systemic insecticides were not applied (No Treatment) or were applied annually to 10%, 20%, 30%, 40% or 50% of randomly selected trees beginning (a) 1 year or (b) 4 years after the initial *A. planipennis* introduction.

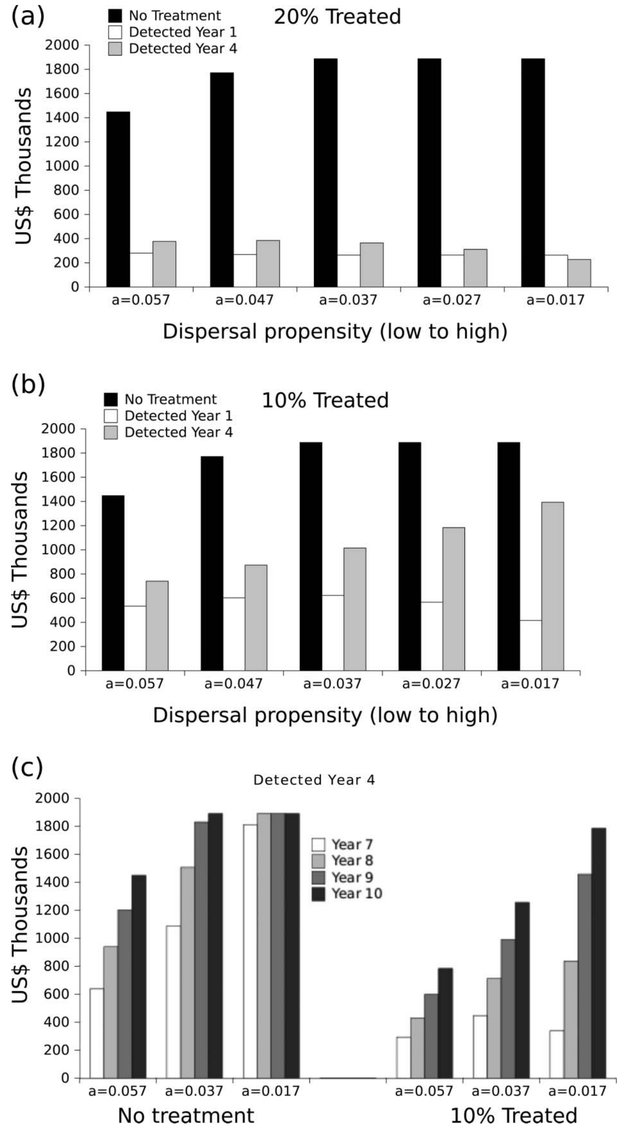


Figure 8. Simulations represent scenarios in which the propensity for adult *A. planipennis* dispersal varied in 200 simulations run with an initial introduction of 400 *A. planipennis* adults. Mean cumulative costs of treating and removing ash trees over a 10-year period are shown for scenarios in which 0 (No Treatment) or (a) 20% or (b) 10% of randomly selected trees were treated annually and treatments began either one or four years after the initial *A. planipennis* introduction, and (c) total costs are shown for Years 7 to10, when 0 (No Treatment) or 10% of randomly selected trees were treated annually and treatments began 4 years after the initial *A. planipennis* introduction.

became heavily infested and required removal (Figure 8(b)). If treatment did not begin until Yr 4, however, the cost effectiveness of treating only 10% of the trees was greatly diminished (Figure 8(b)).

As expected, increasing the propensity for *A. planipennis* dispersal in our simulations resulted in the loss of more trees, higher cumulative costs and costs incurred earlier in the 10-year horizon (Figure 8(c)). This scenario generated a more diffuse distribution of *A. planipennis* across the environment. Growth of the population was initially slow because larval densities in individual trees remained low and a relatively high proportion of larvae required two years for development. The pace of *A. planipennis* population growth increased in later years, however, as larval density built and individual trees became more stressed. This resulted in a high proportion of larvae developing in a single year, more heavily infested trees that required removal, and higher costs.

## 5. Discussion

Managing isolated *A. planipennis* outlier infestations to slow the progression of ash mortality and the rate at which new infestations become established can yield substantial economic benefits for urban and suburban areas on regional and national levels (Kovacs et al. 2010, 2011). Assessing the effectiveness, costs and benefits of managing *A. planipennis* at a local level, however, is vitally important for municipal foresters and property owners. Substantial benefits associated with mature trees in developed and residential settings are well accepted (Nowak and Crane 2002; Casey Trees 2010; Sander et al. 2010), but resources available to maintain the health of urban forests are generally limited. Ash trees comprise a substantial portion of the urban forest canopy in many US cities and suburbs

(Kovacs et al. 2010) and when a new *A. planipennis* infestation is detected, decisions about ash tree management must be made in a relatively short time-frame. Our simulations provide a basis for such decisions.

The results of our simulations are consistent with the progression of ash tree decline and mortality caused by *A. planipennis* observed in many sites in Michigan and surrounding states. Typically, there is little evidence of *A. planipennis* infestation for at least four years following establishment. As *A. planipennis* populations build, pockets of declining ash become apparent and the rate of mortality accelerates over the next few years. In our simulations, serious decline ($>60\%$ phloem loss) was not apparent until four years after the *A. planipennis* introduction. Ash decline and mortality then progressed rapidly and by Yr 9, if no insecticides were applied, all ash trees in our environment had declined and required removal.

The rate at which ash trees in a given area decline and succumb is obviously influenced by numerous factors, including the number of *A. planipennis* beetles introduced into the area. In our scenarios, the infestation was initiated by 400 beetles, consistent with the number of *A. planipennis* beetles estimated to have emerged from a pickup load of infested ash firewood at an outlier site (Mercader et al. 2009). The exponential growth rate of the *A. planipennis* population reflects the vulnerability of ash species (Anulewicz et al. 2007, 2008) and the general lack of effective natural enemies in North America (Bauer et al. 2005; Lindell et al. 2008; Duan et al. 2009). In addition, as host trees become stressed, the proportion of *A. planipennis* larvae that develop in a single year, rather than two years, increases (Mercader et al. 2011b; Tluczek et al. 2011), effectively decreasing generation time.

Removing heavily infested ash trees ($>60\%$ phloem loss) in our simulations eliminated some *A. planipennis* that would otherwise have contributed to reproduction, which slightly delayed the progression of ash decline or mortality. The number of *A. planipennis* beetles that can potentially develop on a tree is limited by the available phloem (McCullough and Siegert 2007a), however, and the ability of infested trees to produce beetles decreases as phloem is consumed. Thus, removing the declining trees did not dramatically affect *A. planipennis* population growth. A similar pattern occurred when we assumed backyard ash trees were not eligible for insecticide treatment. In this scenario, the untreated ash trees in backyards functioned as refuges for *A. planipennis* and would continue to produce beetles until they succumbed, but a greater proportion of trees in the front yards and along the boulevards were treated annually. The overall effect of excluding backyard trees on ash survival over the 10-year period was relatively low.

Even in a best-case scenario, trees are unlikely to be treated with emamectin benzoate until at least the year after the first *A. planipennis* beetles emerge and disperse from infested ash firewood or other infested material. Systemic insecticides, including emamectin benzoate, are ideally injected into ash trees in spring or early summer. This timing ensures the compound can be transported through the tree and into the canopy to control adult *A. planipennis* beetles feeding on ash leaves, as well as larvae (Herms et al. 2009, McCullough et al. 2011). Beetle activity typically peaks between mid-June and early August, depending on local temperatures and latitude (McCullough et al. 2009a, 2009b; Poland et al. 2011) and detection traps in operational programs are generally checked only in middle or late summer (USDA APHIS 2010). The chance that a new *A. planipennis* infestation will be detected and treatments initiated before any beetles disperse is, therefore, very unlikely. Early detection and treatment of new *A. planipennis* infestations contributes to the overall effectiveness of any management program. Under all scenarios considered in our simulations, initiating treatment the year after the *A. planipennis* introduction delayed the onset and progression of ash decline in the area, even when only 10% of the trees were treated. Given the difficulties associated with *A. planipennis* detection, however, infestations will more commonly be discovered only after *A. planipennis* density builds and at least a few trees become symptomatic. Simulations showed that when treatment began four years after the *A. planipennis* introduction, treating 20% of the trees was economically efficient and protected a high proportion of the trees.

Previous efforts to simulate effects of using insecticides for *A. planipennis* management considered only larval mortality (e.g., Mercader et al. 2011a) and the level of protection afforded by the insecticide was lower than that used here. Recent studies have shown mortality of *A. planipennis* beetles approaches 100% when beetles feed, even minimally, on leaves from trees treated with emamectin benzoate during the current or previous year (McCullough et al. 2011). Adult *A. planipennis* must feed on leaves daily and potentially have multiple opportunities to encounter a treated tree during their three to six week life-span. In the simulations presented here, we modeled beetle movement among individual trees within years. The actual extent to which *A. planipennis* adults move among trees is, however, unknown. For this reason, we used a conservative estimate of the number of trees each beetle could encounter in our simulations and limited egg-laying females to encounters with no more than 10 ash trees during her life-span. Altering the propensity for *A. planipennis* dispersal affected the number of surviving trees and costs incurred, which illustrates the importance of considering trivial movement on the potential efficiency of management strategies.

A natural inclination when a new *A. planipennis* infestation is discovered is to focus insecticide treatments on trees in the vicinity of positive traps or trees that are known to be infested or symptomatic. This strategy assumes, however, that the origin and extent of the *A. planipennis* population can be accurately determined. Targeting specific areas for treatment is risky in most cases, given the difficulty of detecting trees with low *A. planipennis* infestations and the very low likelihood of detecting the true origin or the center of mass of an infestation. In our simulations, targeting trees in the area known to be infested provided a high level of control initially, especially when treatments began in Yr 1. Annual beetle dispersal, however, results in an increasing, but unknown, number of trees outside the target area that become colonized. Treatment strategies were not adjusted over time in our simulations and *A. planipennis* populations were allowed to build on trees located outside the target area. Even with perfect knowledge of the origin of the infestation, the benefit of targeting treatments was substantially lower when treatments did not begin until Yr 4 compared with benefits accrued when treatment began in Yr 1. Furthermore, the benefit of targeting treatments decreased over time, even when treatments were applied in the year after the *A. planipennis* introduction. Randomly selecting trees for treatment, for example, ultimately protected more trees over the 10-year period than targeting trees within a 2-block radius from the origin of the infestation for treatment.

These results suggest that municipal foresters should be cautious about focusing treatment efforts only in areas where *A. planipennis* life stages or symptomatic trees have been observed. Obviously, municipal foresters may wish to identify and prioritize specific, high-value ash trees for treatment. Overall, however, when the location and year of origin of an *A. planipennis* infestation is unknown or questionable, our simulations indicate that randomly or systematically selecting trees for insecticide treatment will likely be an optimal strategy.

Results incorporating cost estimates clearly illustrate the economic benefits associated with treating trees with the emamectin benzoate insecticide compared to removing and replacing trees as they decline or die. Treating 20% of the ash trees each year protected substantially more trees at less than half the cumulative costs over the 10-year period compared with treating 10% of the trees. When we assumed treatment began in Yr 1, randomly selecting and treating 20% of the trees annually protected 97% of the ash trees over the 10 years. When only 10% of the ash trees were treated, approximately 60% of them remained in Yr 10, but declining trees would have required removal and replacement, substantially increasing costs. Cumulative costs associated with treating only 10% of the trees were more than double the cumulative costs for treating 20% of the trees, regardless of whether treatment began in Yr 1 or Yr 4. Treating only 10% of the ash trees with emamectin benzoate, however, still yielded lower cumulative costs than not using the insecticide at all. Costs of ash removal and replacement were approximately fourfold higher than in any of the scenarios that included insecticide treatment. Obviously, treatment costs will continue into the future, whereas costs of removal and replacement occur only once. The dramatic difference in cumulative costs incurred, however, means that 20% of the ash trees could be treated for many years before treatment costs would approach removal and replacement costs. When costs of ash tree removal, removal and replacement, or insecticide treatment were compared for ash trees on a Wisconsin campus over a 20-year horizon, VanNatta et al. (2010) similarly found treating all ash trees with insecticides was the most economically viable option.

Economic costs resulting from our simulations do not account for less tangible benefits and ecosystem services provided by large, mature trees in urban forests. In areas where *A. planipennis* has become established, ash trees decline and die over a relatively short period. Widespread mortality, especially when ash trees comprise a substantial component of the urban forest overstory, is likely to be associated with impacts such as increased stormwater run-off, higher cooling and heating expenses for property owners, and reduced property values (Anderson and Cordell 1988; Dwyer et al. 1992; McPherson et al. 1994). In Westland, Michigan, one of the first communities adversely affected by *A. planipennis*, removal of the 3000 municipal ash trees killed by *A. planipennis* led to a 33% increase in outdoor water consumption, which subsequently caused the regional water authority to levy a 10% surcharge on the city (T. Wilson, Westland Dept. of Public Works, pers. comm.). In addition, treating a known number of trees at a known cost enables municipal officials to incorporate the logistics and costs of managing *A. planipennis* into annual budgets without sacrificing maintenance activities that contribute to the overall health of the entire urban forest. Municipal foresters and arborists also have the option of combining treatment with the gradual replacement of ash trees (Krouse 2010; Poland and McCullough 2010). Over time, this strategy can diversify species composition, reducing the overall vulnerability of the urban forest to pest problems.

The emamectin benzoate insecticide was the focus of our simulations because it appears to be the key element for protecting ash trees, slowing *A. planipennis* population growth, and delaying the progression of ash mortality in localized outlier sites (Mercader et al. 2011b). An avermectin product, it has a generally favorable environmental profile and other avermectin products are used in veterinary medicine, aquaculture and for agricultural pests in California (Durkin 2010; Hahn et al. 2011). The emamectin benzoate formulation sold as TREE-age® is a recently developed product; full registration by the US Environmental Protection Agency was granted in 2010. In our simulations, we assumed a single injection of the emamectin benzoate insecticide would provide virtually complete *A. plannipennis* control for two years, but trees would require re-treatment in Yr 3. This level of efficacy is similar to that reported by McCullough et al. (2011), who found larval densities were $>99\%$ lower in treated trees than in untreated controls, even two years post-treatment. Data from recent studies suggest a single emamectin benzoate application may provide high levels of *A. planipennis* control for up to three years (Smitley et al. 2010, D.G.M., unpublished data), which would substantially reduce treatment costs.

Overall, our simulation results show that using the emamectin benzoate insecticide to protect ash trees represents an economically viable management option for urban forests. Ideally, the use of systemic insecticides will be integrated with other *A. planipennis* management tools like those used in SLAM projects. Sanitation cuts to remove diseased, injured or unhealthy ash trees, for example, should be incorporated into short-term and long-term management plans. Three species of parasitoids to *A. planipennis*, native to China, have been released in the USA and may eventually help to slow *A. planipennis* population growth (Bauer et al. 2008). These species, as well as native buprestid parasitoids that attack *A. planipennis* larvae (Cappaert and McCullough 2008; Duan et al. 2009), however, have so far been largely overwhelmed by high *A. planipennis* densities. Treating a portion of urban ash trees with emamectin benzoate may increase the likelihood that parasitoids can exert detectable effects on *A. planipennis* densities. Other natural enemies of *A. planipennis*, including woodpeckers that prey on late instar or prepupal larvae, are unlikely to be affected by emamectin benzoate applications (Hahn et al. 2011). Presumably, additional options for managing *A. planipennis* infestations in North America will eventually become available, decreasing the need to rely on emamectin benzoate or other insecticides. In the immediate future, however, strategies incorporating highly effective, systemic insecticides warrant consideration for slowing the rate of ash decline and mortality in urban forests.

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