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Evaluation of the potential use of a systemic insecticide and girdled trees in area wide management of the emerald ash borer

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ABSTRACT

Emerald ash borer, *Agrilus planipennis* Fairmaire, has become the most destructive forest insect to invade North America. Unfortunately, tactics to manage *A. planipennis* are limited and difficult to evaluate, primarily because of the difficulty of detecting and delineating new infestations. Here we use data from a unique resource, the SL.ow A.sh M.ortality (SLAM) pilot project, to assess whether treating a small proportion of trees with a highly effective systemic insecticide or girdling ash (*Fraxinus* spp.) trees to serve as *A. planipennis* population sinks can result in discernable effects on *A. planipennis* population growth or ash mortality. Components of the SLAM pilot project included an extensive inventory of ash abundance across a heterogenous area encompassing >390 km^2, treatment of 587 ash trees with a highly effective systemic insecticide, and girdling 2658 ash trees from 2009 to 2012. Fixed radius plots were established to monitor the condition of >1000 untreated ash trees throughout the area from 2010 to 2012. While only a very small proportion of ash trees in the project area were either treated with insecticide or girdled, both tactics led to detectable reductions of *A. planipennis* densities and protected ash trees in areas surrounding the treatments. The number of trees treated with the systemic insecticide reduced larval abundance in subsequent years. In contrast, the area of phloem in the insecticide-treated trees had no discernable effect on *A. planipennis* population growth, indicating that the number of treated trees was more important than the size of treated trees. Significant interactions among girdled trees, larval density, and the local abundance of ash phloem indicate girdling trees has a positive, but complex potential as a management tactic.

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1. Introduction

The emerald ash borer, *Agrilus planipennis* Fairmaire, a phloem-feeding insect native to Asia, has become the most destructive forest insect to ever invade North America (Aukema et al., 2011; Herms and McCullough, 2014). Recent evidence has shown this pest became established in southeast Michigan by at least the early 1990s (Siegert et al., 2014) but it was not identified as the cause of ash (*Fraxinus* spp.) decline and mortality until 2002 (Cappaert et al., 2005). Ash mortality rates of >85% were recorded in plots in southeast Michigan and Ohio (Marshall et al., 2013; Burr and McCullough, 2014; Klooster et al., 2014, Knight et al., 2014) and to date, hundreds of millions of ash trees in the U.S. and eastern Canada have been killed by *A. planipennis*. In addition to natural dispersal, inadvertent human transport of infested ash trees, logs or firewood have spread *A. planipennis* long distances and infestations have been found in at least 24 U.S. states and two Canadian provinces (EAB.info, 2015). More than 8 billion ash (*Fraxinus* spp.) trees growing in forests plus millions of ash trees planted in landscapes are threatened by *A. planipennis* in the U.S. (Poland and McCullough, 2006). Economic costs of replacing or treating even half of the landscape ash trees likely to be affected by EAB in urban areas between 2009 to 2019 were projected to exceed 10 billion USD and if surrounding suburbs were included, expected costs doubled (Kovacs et al., 2010). These economic cost

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projections do not include lost ecological services such as stormwater capture in urban areas or effects of widespread ash mortality on nutrient cycling, biodiversity, and forest productivity (Gandhi and Herms, 2010; Burr and McCullough, 2014; Klooster et al., 2014; Flower et al., 2014). Moreover, loss of urban ash in U.S. cities such as Detroit, Michigan was linked to increased human mortality associated with cardiovascular and lower-respiratory-tract illness (Donovan et al., 2013).

Current and potential impacts of *A. planipennis* have elicited strong interest in development of practical and effective management options, particularly in areas with relatively new infestations. Eradication of *A. planipennis* was originally considered following the initial identification of this phloem-boring insect in southeast Michigan, but was abandoned as the extent of the EAB footprint became apparent (Herms and McCullough, 2014). Attention then turned to options for containing or minimally, slowing the spread of EAB populations. Successful containment or management of invasive forest pests, however, is generally dependent upon the timely detection and delineation of newly established infestations (Myers et al., 2000; Liebhold and Tobin, 2008). For example, strategies to slow the spread of gypsy moth (*Lymantria dispar* L) are based on grids of highly effective pheromone traps to detect and delineate small, isolated populations soon after establishment (Sharov et al., 2002; Liebhold and Tobin, 2008). These infestations can then be targeted for mass trapping, mating disruption, microbial insecticide application, or other appropriate tactics (Suckling et al., 2012; Blackwood et al., 2012; Epanchin-Niell et al., 2012; Tobin et al., 2013) to limit population growth and spread.

Unfortunately, detecting and delineating low density *A. planipennis* populations remains challenging. Like its native congeners, *A. planipennis* does not appear to produce effective long range pheromones that could be used as attractants for detection traps, mating disruption or mass trapping. Visual surveys to identify infested trees are also problematic. Larvae in newly infested and relatively healthy trees often require two years to develop (Siegert et al., 2010; Tluczek et al., 2011) and trees exhibit few, if any, external symptoms of infestation until larval densities have increased to moderate or even high levels (Cappaert et al., 2005; Poland and McCullough, 2006; Anulewicz et al., 2007). Some municipalities have attempted to identify infested landscape ash trees by debarking two branches, often accessed with bucket trucks, to assess larval presence (Ryall et al., 2011). This survey method is rarely used in rural or forested areas, however, and the efficacy of the method compared to artificial traps or girdled trees is unknown. Most operational detection programs currently rely on artificial traps in specific shades of green or purple that are suspended in the canopy of ash trees and baited with host volatiles to attract adult beetles with visual and olfactory cues (e.g., Crook and Mastro, 2010; Poland and McCullough, 2014). Field studies, however, have consistently shown the baited canopy traps are not highly effective at low *A. planipennis* densities (McCullough et al., 2011a; Mercader et al., 2012, 2013; Poland and McCullough, 2014). Girdling ash trees in spring then debarking trees in fall or winter to assess larval presence remains the most effective detection method (Rauscher, 2006; Hunt, 2007; McCullough et al., 2011a; Mercader et al., 2013), but is labor-intensive and trees suitable for girdling may not be available. Conventional insecticides cannot be applied in forests or over large areas and while microbial insecticides for *A. planipennis* control continue to be studied (Lyons et al., 2012), they have not been used operationally because of problems with persistence, distribution and efficacy (Herms and McCullough, 2014). Federal agencies in the U.S. have expended considerable efforts in classical biological control with Asian parasitoids (USDA, 2007; Duan et al., 2012, 2014) and native parasitoids and woodpecker predation can account for substantial local mortality of *A. planipennis* larvae (Lindell et al., 2008; Cappaert and McCullough, 2009; Duan et al., 2014; Flower et al., 2014). To date, however, there is no clear evidence that introduced or native natural enemies can regulate *A. planipennis* populations or alter rates of ash mortality in North America.

Given these difficulties and the impacts of unchecked *A. planipennis* infestations, a management approach focused on slowing the progression of widespread ash mortality at the local level by reducing the growth rate of *A. planipennis* populations was proposed (Poland and McCullough, 2010; McCullough and Mercader, 2012). This approach, termed SL.ow A.sh M.ortality, or SLAM, could also potentially slow ash mortality rates at a regional level by reducing *A. planipennis* spread from localized infestations. Kovacs et al. (2011) showed that containing localized *A. planipennis* infestations near urban areas, particularly those distant to the primary infestation, could save or delay millions of USD in economic costs. Simulation models based on empirical data from numerous field studies indicated two tactics, application of a highly effective systemic insecticide and the use of girdled trees as population sinks, were most likely to affect *A. planipennis* population growth (Mercader et al., 2011a, 2011b; McCullough and Mercader, 2012; Kovacs et al., 2014).

Effective protection of individual landscape trees with systemic insecticides applied via trunk injection has advanced considerably since the discovery of *A. planipennis* in North America (Herms and McCullough, 2014; Herms et al., 2014). A systemic product with the active ingredient emamectin benzoate registered in 2010 and sold in the U.S. as TREE-äge® (ArborJet Inc., Woburn, MA) consistently provided at least two and up to three years of nearly complete protection in field studies (Smitley et al., 2010; McCullough et al., 2011b; Herms et al., 2014). This product is injected into the base of the trunk in spring, then translocated in xylem to canopy branches and leaves (Mota-Sanchez et al., 2009; Tanis et al., 2012). Results from laboratory bioassays and extensive field studies have shown *A. planipennis* beetles do not avoid trees treated with TREE-äge® or distinguish between treated and untreated trees (McCullough et al., 2011b). Adult beetles typically die after only one or two bites of a leaf from a tree treated with this product and few, if any, live larvae were recorded when treated ash trees were debarked one to two years post-injection (McCullough et al., 2011b).

Whether applications of a highly effective systemic insecticide, such as TREE-äge®, could affect *A. planipennis* population growth, however, remained to be determined. Results from simulations showed treating trees with this insecticide could slow progression of ash decline and mortality in a local area over time, but effects varied, depending on assumptions about the number and distribution of treated trees (Mercader et al., 2011a; McCullough and Mercader, 2012). When treated trees were assumed to affect only *A. planipennis* larval mortality, simulations suggested a relatively high proportion of trees would need to be treated to significantly reduce ash mortality rates (Mercader et al., 2011a). However, adult *A. planipennis* must feed on ash foliage throughout their 3–6 wk life span and die quickly if they feed on leaves of trees treated with TREE-äge® (McCullough et al., 2011b; Herms et al., 2014). Adjusting models to incorporate adult beetle mortality, along with the multi-year efficacy of the TREE-äge® treatment, yielded significant protection for local ash trees (Mercader et al., 2011a; McCullough and Mercader, 2012).

Girdling ash trees in spring or early summer involves removing a band of outer bark and phloem from around the circumference of the trunk, exposing the sapwood. This causes trees to slowly decline over the course of the season, altering volatile profiles (Rodriguez-Saona et al., 2006) and possibly visual cues associated with hyperspectral signatures of stressed trees (Bartels et al., 2008). Adult *A. planipennis* are attracted to and females preferentially oviposit on ash trees stressed by girdling (Yu, 1992;

McCullough et al., 2009a,b). While larvae on newly infested, healthy ash trees often require two years to complete development, most larvae feeding on girdled trees will develop in a single year, emerging as adults the next summer (Siegert et al., 2010; Tluczek et al., 2011). Girdled ash trees that are debarked or collected and destroyed (e.g., chipped or burned) following oviposition but before larvae develop can be used effectively as *A. planipennis* population "sinks" or as an "attract and kill" strategy, potentially reducing population growth. In addition, overall movement of a local population toward areas containing girdled trees was observed in forested sites where *A. planipennis* densities were very low (Siegert et al., 2009). Simulations suggested, however, this effect could potentially increase colonization of non-girdled trees near girdled trees if no further treatments were applied (Mercader et al., 2011a). Empirical studies and simulations have also shown differential *A. planipennis* attraction to stressed trees is reduced as density increases and a higher proportion of trees become stressed by larval feeding (Mercader et al., 2011b). Consequently, the attractiveness of girdled trees relative to non-girdled trees is likely to diminish over time. Girdled trees, therefore, present a potentially strong, but nuanced, potential for the management of *A. planipennis* at the local level.

Here we used data from the SL.ow A.sh M.ortality (SLAM) pilot project to evaluate whether treating trees with the TREE-äge® systemic insecticide and establishing girdled "sink" trees reduced *A. planipennis* population growth or the progression of ash mortality. The SLAM pilot project, a collaborative effort involving personnel from universities, and federal, state, and county agencies, focused on developing an integrated management approach for an *A. planipennis* infestation in the Upper Peninsula of Michigan (Poland and McCullough, 2010). The pilot project included extensive surveys of ash and *A. planipennis* across a heterogenous area that encompassed portions of a national forest, private forests, residential and rural areas, and small municipalities. Other components of the pilot project included public outreach campaigns to build awareness of EAB and the SLAM pilot project and regulatory efforts to reduce the risk of human transport of potentially infested ash material (Poland and McCullough, 2010; Herms and McCullough, 2014). Effects of the systemic insecticide applications and girdled ash trees were evaluated over multiple years and under varying *A. planipennis* densities across the relatively large project areas.

2. Methods

2.1. Sites

The SLAM pilot project was centered on an isolated *A. planipennis* infestation identified near Moran, (46°02′46″N84°50′35″W), Mackinac Co., Michigan, USA in September 2007, when state regulatory personnel debarked a tree previously girdled as part of an ongoing detection survey (Poland and McCullough, 2010). Cross-sections from this tree and surrounding ash that were felled and debarked by regulatory personnel indicated the *A. planipennis* infestation was likely established by 2003 (DGM, unpubl. data). Project boundaries were established at least 0.8 km (0.5 miles) beyond the furthest tree known to be infested. A few weeks later in 2007, another girdled tree was found to be infested in Straits Park near St. Ignace, (45°51′10″N84°43′10″W), Mackinac Co., Michigan, approximately 12–15 km southeast of the Moran infestation (Poland and McCullough, 2010; Fig. 1).

As part of the SLAM pilot project, systematic surveys of *Fraxinus* spp. were conducted at both the Moran and St. Ignace sites and supplemented with US Dept. of Agriculture Forest Service Forest Inventory and Analysis (FIA) data and stand maps, and inventories of trees in Straits Parks and on municipal land in St. Ignace (Poland and McCullough, 2010). A lattice of 200 × 200 m cells was overlaid over the Moran and St. Ignace project areas. Ash survey data were used to identify ash distribution and estimate the area of ash phloem per cell following McCullough and Siegert (2007). Fig. 1 illustrates the distribution and abundance of ash phloem across the project areas.

2.2. Girdled trees

Each year from 2008 to 2011, survey crews established systematic grids of girdled ash detection trees and additional clusters of girdled ash trees to act as population sinks. Ash trees selected for girdling were growing in full sun (i.e., roadside trees) or along edges of wooded areas, to take advantage of the known preference of adult *A. planipennis* for sunny conditions (Yu, 1992; McCullough et al., 2009a,b). Girdled trees were consistently 10–15 cm dbh, to minimize the risk of wind-breakage but small enough to enable trees to be efficiently debarked in autumn. Trees were girdled in June by removing a 10–15 cm wide band of outer bark and phloem from around the circumference of the trunk with drawknives, approximately 1 to 1.3 m aboveground (Mercader et al., 2013). All girdled trees were carefully debarked by removing the outer bark on the trunk and any branches >3.0 cm in diameter with drawknives in autumn and if *A. planipennis* larvae (or galleries) were present, larval counts were recorded for each infested tree. Quality control procedures were implemented annually to ensure accuracy (Mercader et al., 2013). Density of detection trees ranged from one to 16 per km^2 in the area around the origin of both infestations and one per 2.6 km^2 in the outer grid cells of the project area (Mercader et al., 2013). Results of *A. planipennis* surveys were evaluated annually by collaborators to establish boundaries and density of detection trees for the subsequent year as described in Mercader et al. (2013). Survey crews girdled 385 trees, 603 trees, 773 trees, and 855 trees in 2008, 2009, 2010 and 2011, respectively (Fig. 2).

2.3. Insecticide treatments

Private contractors treated 229 trees in 2009 and 358 trees in 2010, ranging from 8.6 cm dbh to large multi-trunk trees with the phloem equivalent of trees up to144.8 cm dbh, with the systemic insecticide TREE-äge® a product with the active ingredient emamectin benzoate (4% a.i., Arborjet, Inc., Woburn, Massachusetts). Trees were injected at relatively low rates (0.2 g a.i. per 2.5 cm dbh) in June with the ArborJet Quik-Jet® and Tree IV® devices following label directions (Arborjet Inc., Woburn, MA, USA). Initially, insecticide-treated trees were to be concentrated in and around areas known to be infested by *A. planipennis.* However, unanticipated restrictions on insecticide treatments on National Forest land arose in spring 2009. At about the same time, the area designated as critical habitat for the Federally-listed endangered Hine's emerald dragonfly was substantially expanded. These restrictions limited treatment of trees across a significant portion of the project area. As a result, most insecticide-treated trees were concentrated in a few areas of private land or were growing on rights-of-way along roads (Fig. 3).

2.4. Treatment effects on local A. planipennis abundance

We evaluated effects of treatments, i.e., insecticide-treated trees and girdled trees, on the local abundance of *A. planipennis* recorded when girdled trees were debarked. Our primary interest was to determine whether treatments could produce a detectable reduction of *A. planipennis* at the population level. Therefore, effects on local abundance of *A. planipennis* would be expected in the years following the implementation of the girdling and

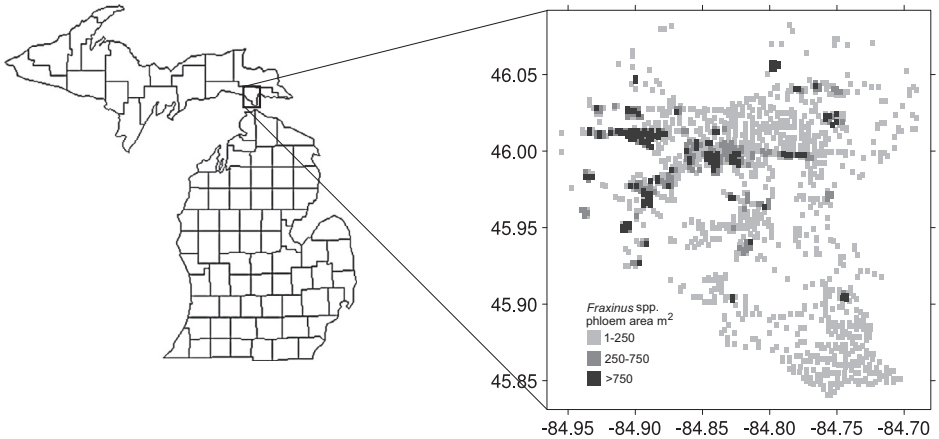

Fig. 1. Estimated area (m^2) and distribution of ash (*Fraxinus* spp.) phloem in 200 × 200 m grid cells across the SLAM pilot project area.

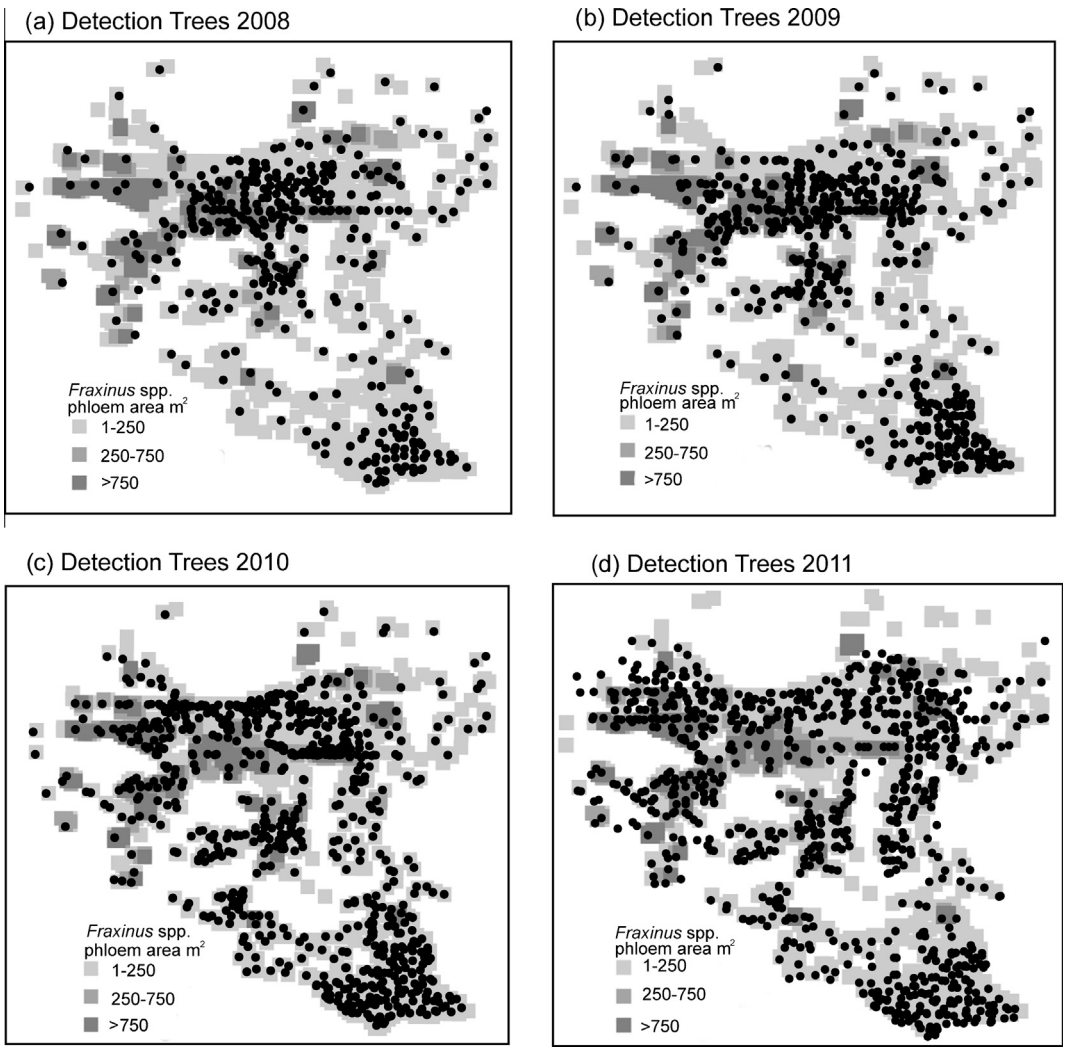

Fig. 2. Distribution of girdled ash (*Fraxinus* spp.) detection and sink trees across the SLAM pilot project area overlaid on ash phloem area (m^2).

insecticide injections. This is particularly true for the deployment of girdled trees, given that (1) the detection tool and the treatment were the same and (2) girdled trees were expected to attract beetles to the vicinity, potentially producing the false appearance of an increase in population size the year trees were girdled. For these reasons, we analyzed effects of the 2009 treatments on larval data collected from trees debarked in autumn 2010, 2011, and 2012, effects of the 2010 treatments on larval data collected in autumn 2011 and 2012, and effects of 2011 treatments on larval data collected in 2012. In contrast to girdled trees, the insecticide-treated trees could potentially reduce population size the same year the treatment is applied due to adult beetle mortality. However,

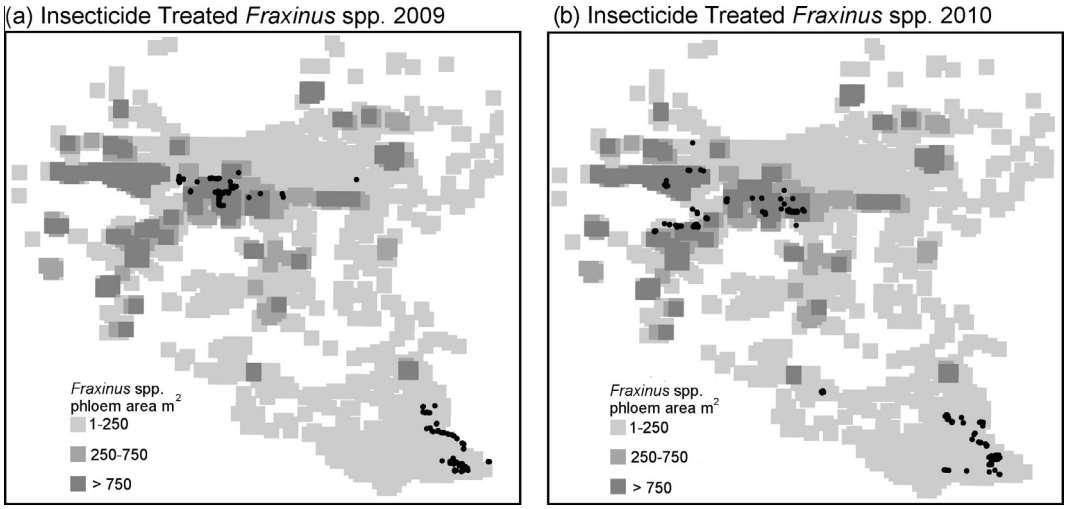

Fig. 3. Distribution of ash (*Fraxinus* spp.) trees treated with a systemic insecticide (TREE-age™) overlaid on ash phloem area (m^2).

detecting such an effect, which would likely be relatively subtle, would require a substantially greater density of insecticide-treated trees and detection trees, given the geographic scale of the project. We concentrated our analyses, therefore, on evaluating treatment effects on populations of *A. planipennis* in successive years.

We assumed treatment effects could be influenced by either the number or the size of treated trees, or both. Girdled trees were consistently 10–15 cm in dbh with a mean dbh of 11.3 cm and standard deviation of 1.9 cm, but insecticide-treated trees ranged from 8.6 to 144.8 cm dbh with a mean dbh of 29.6 cm and a standard deviation of 15.7 cm. We therefore incorporated two methods of defining treatments in our analyses. First, we simply considered the number of girdled or insecticide-treated trees within 800 m of each detection tree. This represented the number of toxic trees beetles could encounter or, in the case of girdled trees, the number of point sources acting as attractants and sinks. We selected an 800 m radius for the analyses based on results from previous studies. Laboratory studies with a flight mill have shown mature adult female *A. planipennis* could potentially disperse >2 km (Taylor et al., 2010). Intensive field studies, however, showed most eggs are laid within 100 m of the emergence point of the adult female and only a small proportion of females dispersed and laid eggs on trees 400–800 m away (Mercader et al., 2009; Siegert et al., 2010). Second, we used the total area of ash phloem in treated trees as a function of distance from each individual detection tree. Area of phloem within each tree that was either girdled or injected was calculated using dbh of the tree, following methods of McCullough and Siegert (2007). That value was then divided by the distance of treated trees to each detection tree. Results were summed for the sink and insecticide-treated trees separately, for each detection tree.

To determine whether the treatments significantly affected *A. planipennis* population size, a series of generalized linear mixed models (GLMM) were fit to the data using the glmmPQL function available from the MASS package (Venables and Ripley, 2002) for the R statistical package (R Development Core Team, 2014). We included local ash phloem density (ash phloem area within 200 m) and an estimate of local *A. planipennis* density as factors in our analyses, in addition to the treatment effects. Phloem area present within 200 m of each detection tree was estimated as a weighted average of the phloem area present in all grid cells and weighted by distance from a specific detection tree to each grid cell. A proxy for *A. planipennis* abundance was estimated for each site using the average of *A. planipennis* presence (1) or absence (0) in detection trees from the previous year that were growing within 2.0 km of each current-year detection tree. These estimates were again weighted by distance from a specific detection tree to each grid cell, as before. We used 2.0 km as a cutoff, because some portion of the female beetles could potentially fly this far or farther (Taylor et al., 2010).

Moran's I and semivariograms of the residuals were calculated to test for residual autocorrelation. When necessary, defined correlation structures were used in models to account for residual autocorrelation. Correlation between error terms was assumed to be a function of the distance between sampling points. Gaussian, spherical, and exponential correlation structures were attempted for the analyses as described in Dormann et al. (2007). Analyses of the 2009 treatment effects on the observed *A. planipennis* detections in 2010 for the Moran and St. Ignace sites indicated that residual autocorrelation remained an issue and those results should be interpreted with caution.

2.5. Treatment effects on ash condition

Survey crews established circular, fixed radius plots as time permitted during the 2010–2012 growing seasons to assess current and, ideally, long term condition of ash and other overstory species. Plots were established systematically, excluding areas where ash trees were not present. Plot size varied, depending on local ash density, ranging from 404 m^2 where ash was scarce down to 40 m^2 in areas with abundant ash. There were 71 plots with 784 ash trees, 111 plots with 1133 ash trees, and 103 plots with 1081 ash trees in 2010, 2011 and 2012, respectively. Variables recorded for ash trees followed methods used by the USDA Forest Service FIA program (Schomaker et al., 2007), including estimates of tree vigor, canopy dieback, uncompacted live crown ratio, and crown density. Crew members were trained by experienced forest health professionals and quality control procedures were applied to ensure accuracy and consistency.

Visual estimates of vigor were recorded as: 1 = healthy, few dead twigs; 2 = occasional dead branch, foliage density below normal; 3 = moderate dieback, several dead branches; 4 = up to 1/2 crown dead; 5 = over 1/2 crown dead and 6 = tree dead. For analysis, however, we combined ratings into three classes (1 + 2; 3 + 4; and 5 + 6) and applied the Cumulative Link Mixed Models (CLMM) function in the Ordinal package (Christensen, 2012) for the R statistical package (R Development Core Team, 2014). Treatments and estimates of *A. planipennis* density were included as fixed effects in these analyses, as described above (see analyses

of *A. planipennis* local abundance). We also included estimates of the presence and severity of "other" damage (i.e., damage not related to *A. planipennis*) in these analyses if the damage preceded *A. planipennis* colonization (USDA Forest Service, 2005; Schomaker et al., 2007). This variable effectively represents an estimate of tree health affected by factors other than *A. planipennis*. This variable was treated as a fixed effect and as trees in each plot were not independent, plot was fit as a random effect. Significance tests for fixed effects in the models were conducted by fitting models with and without the fixed effect of interest and conducting likelihood ratio tests.

Estimates of canopy dieback, uncompacted live crown ratio, and crown density were recorded as percentages (Schomaker et al., 2007). However, these variables were recorded as either 99% or 0% for a considerable number of trees and transformations to allow the use of a standard distribution could not be achieved without excluding a large portion of the data. We therefore treated these variables as binary data, where values of 60% or above were treated as 1 and those below 60% were treated as zero. Binary data were analyzed as generalized linear mixed models using a logit link and Laplace approximation in the glmer function in the lme4 package (Bates et al., 2014) for R. As in the CLMM analyses above, we included an estimate of "other" damage as a fixed effect and plot was fit as a random effect.

In all analyses, effects were considered significant at the 95% level, and marginally significant at the 90% level and are reported as such in the results.

3. Results

3.1. Treatment effects on local A. planipennis abundance

For simplicity, results of the analyses are presented by the year in which the detection trees were sampled (2010 and 2011). Overall, results showed that treating trees with the systemic insecticide or girdling trees to act as sinks significantly reduced local *A. planipennis* populations in the SLAM pilot project areas.

3.1.1. Effects of treatments on 2011 A. planipennis abundance

Surveys of *A. planipennis* in 2008–2010 indicated the two infestations were distinct and separate but they had clearly merged by 2011 (Fig. 4). The number of insecticide-treated trees in either 2009 or 2010 within 800 m of the 2011 detection trees significantly reduced the number of larvae present in those detection trees when they were debarked in autumn 2011 ($t_{841} = -2.604$ and $P = 0.009$). Coefficients from the generalized linear mixed model summarized in Table 1 can be interpreted as the expected reduction or increase in the log count of larvae for the effect in question conditional on the other effects included in the model. Therefore for every insecticide treated tree placed within 800 m of the detection tree the expected log count of larvae is reduced by 0.011. Although the size of this reduction is small (Table 1), it must be considered in terms of the scale of the project, i.e., the number of insecticide-treated trees within an area of approximately 2 km^2 (the 800 m radius around the detection tree).

Likewise, the number of sink or detection trees girdled in 2010 located within 800 m of the 2011 detection trees significantly reduced the number of larvae present in those trees ($t_{841} = -2.44$ and $P = 0.0148$). There was no evidence of interactions between insecticide-treated trees and our *A. planipennis* density proxy, Prior Detections, and our ash density proxy, Phloem Area, but there were for girdled trees. We also analyzed the detection tree data as a binomial variable, infested/not infested, which indicated qualitatively identical responses.

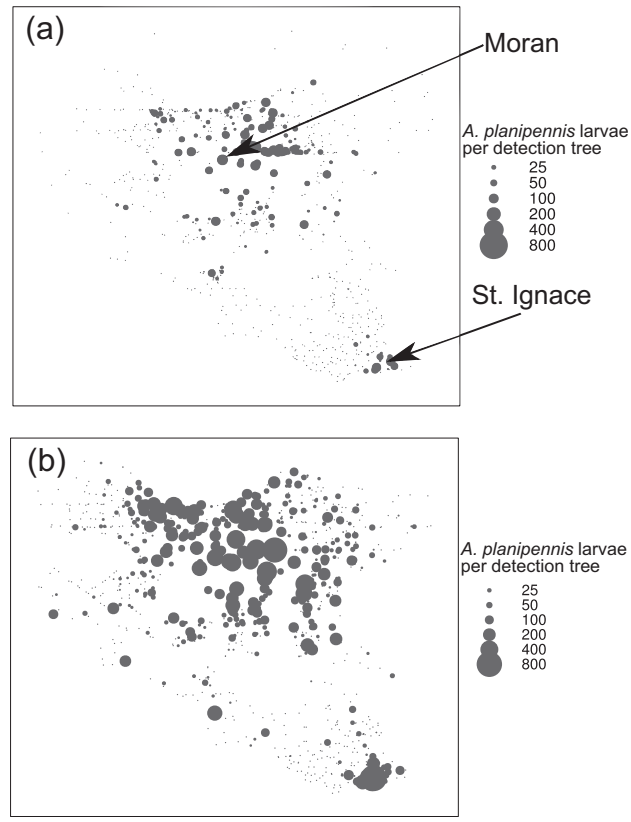

Fig. 4. Counts of *A. planipennis* larvae recorded in 2010 and 2011 from girdled ash (*Fraxinus* spp.) detection and sink trees in the SLAM pilot project area.

Table 1
Generalized linear mixed-effects model of effects of insecticide applications in 2009 and 2010, trees girdled in 2010, *A. planipennis* detections in 2010, and local phloem area on *A. planipennis* larval counts recorded from girdled detection trees debarked in autumn 2011 in the SLAM pilot project area.

Variable	Estimate	Std. error	*t*	*df*	*P*-value
Treated trees within 800 m					
Intercept	2.943	1.228	2.397	841	**0.017**
Insecticides (I)	−0.011	0.004	−2.604	841	**0.009**
Sinks (S)	−0.314	0.128	−2.442	841	**0.015**
Phloem area (P)	−0.025	0.009	−2.857	841	**0.004**
Prior detections (D)	0.335	2.237	0.150	841	0.881
P × D	0.051	0.240	3.406	841	**<0.001**
S × P	0.003	0.001	3.399	841	**<0.001**
S × D	0.817	0.240	3.487	841	**<0.001**
S × D × P	−0.007	0.002	−4.093	841	**<0.001**
Treated phloem area/distance					
Intercept	3.702	1.979	1.871	841	0.062
Insecticides (I)	0.025	0.02	1.247	841	0.213
Sinks (S)	−9.453	4.63	−2.042	841	**0.042**
Phloem area (P)	−0.036	0.015	−2.419	841	**0.016**
Prior detections (D)	0.884	3.743	0.236	841	0.814
P × D	0.059	0.027	2.197	841	**0.028**
S × P	0.096	0.035	2.753	841	**0.006**
S × D	15.891	7.954	1.998	841	**0.046**
S × D × P	−0.156	0.059	−2.643	841	**0.008**

Complex interactions between the effectiveness of detection/sink trees under varying ash and *A. planipennis* density proxies are illustrated in Fig. 5. Not surprisingly, in the absence of girdled sink/detection trees, as *A. planipennis* density in the surrounding area the previous year increases, the expected number of larvae in a given detection tree also increases. However, as ash phloem area per cell increases, the expected larval density, as a function

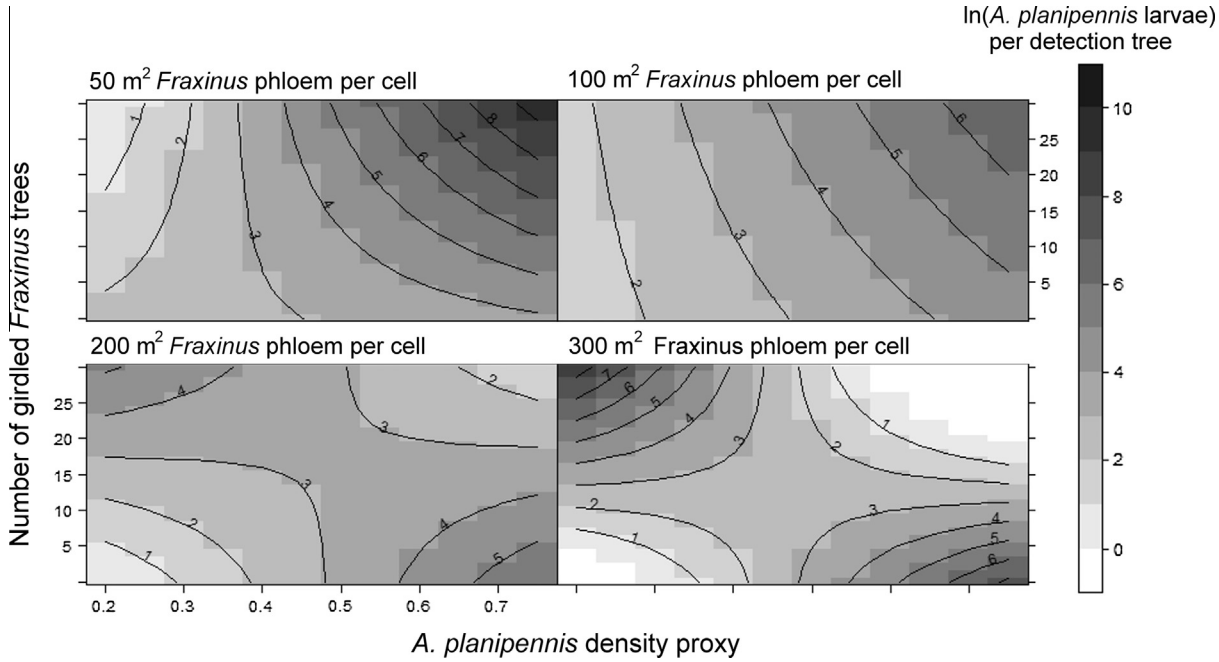

Fig. 5. Expected log number of larvae in randomly sampled girdled ash (*Fraxinus* spp.) detection trees in the SLAM pilot project area as a function of the number of girdled trees present within 800 m and the local *A. planipennis* larval density. Estimates were conducted for areas with ash phloem densities of 50, 100, 200, and 300 m^2 per 200 × 200 m cell. The larval density proxy for the area was estimated as the distance weighted mean of trees detected as positive (1) and negative (0) the previous year, ranging from 0 to 0.8 (0–80% of trees in the immediate area infested).

of local *A. planipennis* density in the previous year, increases more rapidly in areas with higher amounts of ash phloem (per cell) than in areas where phloem is scarce. This effect reflects the large numbers of *A. planipennis* that must be present to produce high *A. planipennis* densities in areas where ash phloem is abundant and the attraction of ovipositing females to areas with high ash phloem (Siegert et al., 2010). Of even more interest, the effect of detection/sink trees girdled the previous year on the expected larval density in the current year differs as a function of both ash phloem and local *A. planipennis* density. In other words, results indicate that when ash phloem is low, girdled trees will likely reduce the local *A. planipennis* population if they are placed in areas where the local *A. planipennis* density was low in the previous year. However, in areas where ash phloem is low but local *A. planipennis* densities were relatively high the previous year, girdled trees will likely increase the *A. planipennis* population in the immediate vicinity. In contrast, when ash phloem is high, the inverse is expected; girdled trees decrease the local *A. planipennis* population when *A. planipennis* densities in the surrounding area are high. These results highlight the duality of detection trees as both an attractant and population sink for *A. planipennis* and the heterogeneous distribution of *A. planipennis* within infested areas.

When we examined effects of phloem area in treated trees weighted by distance (Table 1), the phloem area of 2010 girdled trees significantly affected *A. planipennis* density in 2011, but the phloem area in trees treated with the insecticide in 2009 or 2010 did not. Nearly all girdled trees were 10–15 cm dbh, but the size of insecticide-treated trees varied considerably. Taken together, these results indicate the number of insecticide-treated trees had a greater effect on *A. planipennis* density than total phloem area treated with the insecticide. This result is logical as the likelihood that an ovipositing female would encounter a toxic tree should increase as a function of the number of treated trees.

When considering the number of trees treated with insecticide in 2009, a significant reduction in the number of larvae present in detection trees in 2011 ($t_{845} = -2.476$ $P = 0.014$) was also observed. This result was anticipated, given that field studies have consistently shown this insecticide remains highly effective for at least two years (McCullough et al., 2011b; Herms et al., 2014). In contrast, the number of detection/sink trees girdled in 2009 (within 800 m) did not significantly affect larval counts in the 2011 detection trees ($t_{845} = -0.513$ $P = 0.608$).

3.1.2. Effects of treatments on 2010 A. planipennis abundance

In 2010, the St. Ignace and Moran areas were considered to be two distinct infestations, given that most detection trees in between the two projects areas were uninfested (Fig. 4). Treatment intensity, i.e., the number of injected or girdled trees per km^2, was considerably greater in the St. Ignace area than in the Moran area (Fig. 3). Therefore, larval counts from the 2010 detection trees were analyzed separately for each area.

When the number of treated trees within 800 m of the 2010 detection trees was considered for the St. Ignace area, we observed no significant effects of either insecticide-treated trees ($t_{148} = -1.27$ and $P = 0.206$) or girdled trees ($t_{148} = -0.527$ and $P = 0.599$). However, when the phloem area in insecticide-treated trees in 2009 was considered, we did see a significant difference in the larval counts in the 2010 detection trees ($t_{148} = -3.648$ and $P < 0.001$). This reflects the relatively high number and concentration of trees injected with the insecticide in the comparatively more recently established *A. planipennis* population in the St. Ignace area, compared to the Moran area.

Density and distribution of *A. planipennis* in the Moran area were considerably greater than in the St. Ignace area (Fig. 4). Due to logistical constraints, larval counts in 2010 were truncated, which along with a high level of heterogeneity, led to poor model fits to the count data. Therefore, the data were analyzed as presence or absence of *A. planipennis* in the detection trees. Although less sensitive, this analysis indicated that insecticide-treated trees in 2009 had a marginally significant effect on the probability of a tree being detected as infested ($t_{557} = -1.81$ and $P = 0.071$) and the girdled detection/sink trees had a significant effect ($t_{557} = -2.43$ and $P = 0.015$).

3.2. Treatment effects on ash condition

A systematic network of plots established in the SLAM project area was intended primarily to monitor condition of overstory ash and other species over time. Potential effects of the insecticide applications or girdling on the condition of ash trees observed within 1–2 years, as is the case here, would presumably be relatively minor and most likely would be observed in trees previously stressed by *A. planipennis* or other factors. Therefore, small but detectable treatment effects should be viewed cautiously, but not disregarded.

3.2.1. Effects of treatments on 2012 ash condition

Overall, results from plots surveyed in the Moran-St. Ignace project area in 2012 provide evidence that the insecticide and girdling treatments afforded ash trees some protection from *A. planipennis* injury. Proximity to insecticide-treated trees was marginally related to estimates of ash vigor ($z = 0.016$; $P = 0.06$). When vigor was recorded, low values were indicative of healthy trees, while high values indicated trees were severely declining or dying. Thus, the negative relationship between the number of insecticide-treated trees within 800 m of the plots and vigor estimates indicates trees in plots near higher densities of insecticide-treated trees were more likely to be healthy (Table 2). Other variables, including crown density, canopy dieback, and uncompacted live crown ratio were not significantly affected by the number of injected trees within 800 m of the plot (Table 3).

The number of girdled trees within 800 m of the plots in 2010 and 2011 did not affect estimates of ash condition recorded during the growing season in 2012 (Table 3). Tree condition in 2012 did, however, appear to be positively affected by proximity to trees girdled in 2011. There were marginally significant ($P < 0.1$) beneficial effects on canopy dieback ($z = -1.684$; $P = 0.0923$), canopy transparency ($z = -1.889$; $P = 0.0588$), and crown density ($z = 1.712$; $P = 0.0869$). In contrast, the number of trees girdled in 2009 within 800 m of the plots had detrimental effects on trees evaluated in 2011, including vigor estimates ($z = 0.045$; $P = 0.0065$), canopy dieback ($z = 2.456$; $P = 0.014033$), transparency ($z = 2.530$; $P = 0.01141$), and uncompacted live crown ratio ($z = -2.331$; $P = 0.0198$). If girdled trees attracted beetles to a locality in 2009, progeny of those beetles would have been feeding in 2009–2011 and may have been responsible for the diminished health of trees reflected in the 2012 estimates. These results corroborate the complex effect of girdled trees noted earlier; girdled trees can act both as sinks to reduce population size, but may also attract mated female beetles that can oviposit on nearby trees.

3.2.2. Effects of treatments on 2011 ash condition

Proximity of ash trees to insecticide-treated trees in 2009 or 2010 again indicated a significant effect on estimated vigor ($z = 0.017$; $P = 0.044$), but other variables were not affected. Likewise, none of the ash condition variables recorded in 2011 were affected by proximity to the 2010 girdled trees (Table 4). Proximity to the 2009 girdled trees had a detrimental effect on vigor estimates ($z = 0.0577$; $P = 0.0122$).

4. Discussion

Detecting and accurately delineating new *A. planipennis* infestations in North America remains remarkably difficult (Crook and Mastro, 2010; Mercader et al., 2012) and barring unforeseen developments, will likely continue to challenge scientists, regulatory officials, arborists, and foresters. Most *A. planipennis* infestations are at least 4–6 years old before they are discovered (Herms and McCullough, 2014) and this situation seems unlikely to change. Managers, therefore, will nearly always lack the ability to clearly define *A. planipennis* distribution and density in established populations. A key element of an operational SLAM program, therefore, is the emphasis on protecting the ash resource rather than focusing effort and available funds on intensive *A. planipennis* surveys. The extensive grid of girdled detection trees employed in the SLAM pilot project provided a unique opportunity to monitor this

Table 2
Cumulative link mixed effect models of effects of insecticide applications in 2009 and 2010, trees girdled in 2009, 2010, and 2011, and *A. planipennis* detections in 2011 on ash (*Fraxinus* spp.) vigor estimated in summer 2012 in fixed radius plots in the SLAM pilot project area. Vigor ratings of 1 indicated trees were healthy while higher ratings were indicative of canopy decline or dieback.

	Vigor	
Variable	Estimate	*P*-value
Insecticides (I)	−0.029	**0.060**
Sinks 11 (S11)	−0.033	0.414
Sinks 10 (S10)	0.036	0.358
Sinks 09 (S09)	0.095	**0.001**
Other damage	0.322	**<0.001**
Prior detections (D)	1.820	**0.001**

Table 4
Cumulative link mixed effect models of effects of insecticide applications in 2009 and 2010, trees girdled in 2009 and 2010, and *A. planipennis* present in 2010 on visual estimations of ash (*Fraxinus* spp.) tree vigor estimated in summer 2012 in fixed radius plots in the SLAM pilot project area. Vigor ratings of 1 indicated trees were healthy while higher ratings were indicative of canopy decline or dieback.

	Vigor	
Variable	Estimate	*P*-value
Insecticides (I)	−0.034	**0.044**
Sinks 10 (S10)	−0.008	0.800
Sinks 09 (S09)	0.058	**0.012**
Other damage	0.424	**<0.001**
Prior detections (D)	0.866	0.199

Table 3
Generalized linear mixed effect models with effects of insecticide applications in 2009 and 2010, trees girdled in 2009, 2010, and 2011, and *A. planipennis* presence in 2011 on the visual ratings of canopy transparency, dieback, uncompacted live crown ratio and crown density of ash (*Fraxinus* spp.) trees surveyed in 2012 in fixed radius plots.

Variable	Transparency		Dieback		Crown ratio		Crown density	
	Estimate	*P*-value	Estimate	*P*-value	Estimate	*P*-value	Estimate	*P*-value
Intercept	−2.606	**<0.001**	−4.842	**<0.001**	0.343	0.215	−0.355	0.172
Insecticides (I)	−0.042	**0.039**	−0.026	0.235	0.009	0.385	0.003	0.786
Sinks 11 (S11)	−0.088	**0.059**	−0.095	**0.092**	0.028	0.319	0.046	0.087
Sinks 10 (S10)	0.034	0.409	0.040	0.339	−0.011	0.690	−0.038	0.189
Sinks 09 (S09)	0.075	**0.011**	0.066	**0.014**	−0.049	**0.02**	−0.039	**0.0962**
DBH	−0.101	**0.025**	0.001	0.864	−0.099	**<0.001**	0.184	**<0.001**
Other damage	−0.154	**0.025**	−0.194	**<0.001**	0.112	0.106	−0.447	**<0.001**
Prior detections (D)	1.839	**0.005**	2.968	**0.002**	0.269	0.483	−1.334	**<0.001**

infestation, but also enabled assessment of whether girdling or treating a small proportion of ash trees in the area could exert detectable effects on *A. planipennis* population growth or distribution.

Despite the very small proportion of girdled or insecticide-treated trees in the large, heterogenous project area, results indicate both tactics can reduce *A. planipennis* populations and slow the progression of ash injury and mortality. Specifically, we detected significant effects of the girdled trees and insecticide-treated trees on *A. planipennis* densities and variables related to ash condition in areas surrounding girdled or insecticide-treated trees.

The 587 trees (229 trees in 2009; 358 trees in 2010) treated with the TREE-äge® systemic insecticide represented much less than 1% of the ash trees or ash phloem available to *A. planipennis* across the SLAM project area, which encompassed more than 390 km^2 by 2011. Treating even this minute proportion of the ash trees in the area still yielded detectable reductions in subsequent *A. planipennis* larval counts in the Moran and St. Ignace areas. The TREE-äge® insecticide selected for this project has consistently provided highly effective protection from *A. planipennis* for two years after injection (Smitley et al., 2010; McCullough et al., 2011a; Herms et al., 2014) and is not known to repel or affect host selection of *A. planipennis* beetles. Intensive studies have shown leaves from trees treated with TREE-äge® remained toxic to adult *A. planipennis* for at least two years and few larval galleries were present when treated trees were debarked one and two seasons post-treatment (McCullough et al., 2011b). Other systemic insecticides, such as imidacloprid or azadirachtin products that are less persistent or that have different modes of action, can effectively protect individual ash trees if applied annually (Herms et al., 2014), but without further evaluation, we cannot assume that such products would yield results similar to those observed here.

The number of insecticide-treated trees within 800 m of untreated ash trees was also significantly and positively related to estimates of tree vigor recorded when untreated trees were surveyed in 2012. This result was somewhat unexpected, given the relatively short time frame between the onset of treatments and the evaluation of tree condition in the fixed radius plots. If insecticide treatments and monitoring had continued over time, these differences would presumably have become more pronounced as *A. planipennis* densities increased in areas where no treatments occurred. This result is notable in that it suggests the insecticide treatments protected not only the trees that were injected but also could potentially be used as part of area-wide management strategies for *A. planipennis*. While much of the SLAM pilot project area was forested, these results may hold even more promise for urban and suburban areas where ash trees would be easier to access and a higher proportion of the ash trees could potentially be treated at two or perhaps three year intervals. Results from simulations indicated that in an urban or residential setting, treating 20% of the ash annually with TREE-äge® or a similarly effective product could dramatically reduce ash mortality and economic costs over a ten-year period (McCullough and Mercader, 2012; Kovacs et al., 2014). The presence of detectable, area-wide effects of the insecticide treatments in the SLAM pilot project provides a basis for cautious optimism that foresters or urban foresters can successfully manage *A. planipennis* within typically limited budgets and logistical constraints.

The portion of ash trees/phloem treated with the systemic insecticide, relative to the total number of ash trees at the site, was considerably higher in the St. Ignace area than in the Moran area (Fig. 3). Moreover, in both St. Ignace and Moran, most insecticide-treated trees were concentrated in relatively small areas where applicators could readily access and inject trees. Clearly, the number and distribution of insecticide-treated trees was less than optimal than it would be in a research study designed to evaluate area-wide effects of the treatment. Similar logistical constraints, however, are likely to be encountered in nearly any operational project undertaken to slow *A. planipennis* population growth and damage. Nevertheless, the dearth and unequal distribution of insecticide-treated trees limited our ability to assess treatment effects under different initial *A. planipennis* densities. We did not detect a significant interaction between the number of insecticide-treated trees and our *A. planipennis* density proxy. We suspect this result reflects the concentrations of insecticide-treated trees where infestations were highest, particularly in the St. Ignace site, and does not necessarily indicate the insecticide treatment similarly affected *A. planipennis* populations regardless of *A. planipennis* densities.

Effects of insecticide-treated trees on the *A. planipennis* population in 2011 were significant when the number of treated trees was considered, but not when the area of treated ash phloem was considered. This is an important difference because it indicates the number of treated trees was of greater significance than the total quantity of insecticide product applied when considering the impact of insecticide treatments across a large area. This result is not entirely surprising as increasing the number of treated trees increases the likelihood that *A. planipennis* adults will encounter a toxic tree while feeding on foliage or laying eggs. However, at a smaller scale, the quantity of phloem treated with insecticide may be relevant. In 2010, when the St. Ignace population still appeared to be a distinct infestation, insecticide treatments had a significant effect on the *A. planipennis* population when the total phloem area treated was considered, but not when the number of trees treated was considered. Not only was the St. Ignace project area relatively small, most of the insecticide-treated trees in 2009 were concentrated in only a few areas, predominantly in the town of St. Ignace, where the treatment provided strong control of the local *A. planipennis* populations. Anecdotal evidence of a similar effect has been observed in field trials when treatment of a high proportion of ash trees protected untreated control trees (DGM, unpubl. data).

A significant reduction in the *A. planipennis* population attributable to the girdled detection/sink trees was also observed in Moran and St. Ignace, but our analysis revealed effects of girdled trees that were considerably more complex compared to the insecticide applications. Significant interactions with the ash density proxy and the *A. planipennis* density proxy were detected in the project area, indicating girdled trees had multifaceted effects on local *A. planipennis* populations. The girdled trees directly reduced the *A. planipennis* population by removing the larval progeny of female beetles that were attracted to and preferentially oviposited on the stressed trees. The detection/sink trees, however, can also attract beetles to adjacent or nearby ungirdled trees, effectively producing a spillover effect and increasing the larval density on those trees. These results highlight the importance of considering how girdled trees may influence the movement and host selection behavior of adult beetles (Mercader et al., 2012). Attracting ovipositing females to one or a few stressed trees can lead to a more general attraction of a local, low density *A. planipennis* population toward trees in the vicinity of the stressed trees (Siegert et al., 2009). Therefore, while girdled trees can function as a population sink for *A. planipennis*, attraction of beetles may result in more *A. planipennis* larvae in that area than the number eliminated by destruction of the girdled trees.

This effect may also be reflected in the impact of girdled detection/sink trees on ash surveyed in the fixed radius plots. Estimates of tree vigor were significantly ($P < 0.05$) or marginally significantly ($0.1 > P > 0.05$) higher when the number of trees girdled in the previous year was considered. In contrast, the number of trees girdled near the plots girdled two or three years prior to

surveys of ash condition yielded significant or marginally significant effects in the opposite direction. Simulations have previously indicated that the attraction of mature *A. planipennis* females to an area with girdled trees can potentially affect the outcome of management efforts (Mercader et al., 2011a,b). The empirical results presented here demonstrate the important role that attraction of *A. planipennis* females to an area containing stressed trees can play in determining results of *A. planipennis* management efforts.

Despite the potential risk associated with girdling trees to attract *A. planipennis* females, this may also be a key element in a thoughtfully designed, integrated pest management strategy. Girdled trees, for example, could potentially be used to direct an *A. planipennis* population away from high value trees. Of even more interest is the potential to combine girdled sink trees with insecticide-treated trees. Ideally, girdled trees and insecticide-treated trees could be coupled to provide a more effective "attract and kill" design. For example, "sinks" consisting of clusters of three or more girdled trees could be used to retain adult beetles and reduce dispersal. Those trees could be surrounded by or intermixed with trees injected with the TREE-äge® insecticide. Such combinations would prevent the spillover effect detected here and could perhaps increase the population reduction resulting from adult beetles encountering toxic trees. Alternatively, "sinks" could be established in sacrificial zones with low value ash trees while high value ash trees in nearby areas are protected with the systemic insecticide.

The presence of detectable effects despite the small number of treatments is remarkable, but also highlights the need for higher density of treatments to functionally manage *A. planipennis* populations. This is likely to be most economically feasible in urban and suburban areas (McCullough and Mercader, 2012; Kovacs et al., 2014). Potential spillover effects associated with girdled trees could be problematic if not anticipated and addressed as part of an overall management strategy. However, intermixing or coupling girdled trees and trees treated with highly effective systemic insecticides could potentially yield synergistic effects, by attracting beetles to areas where they are more likely to encounter a toxic tree. This effect could reduce the number of insecticide-treated trees needed and associated costs, and warrants further exploration as progress toward area-wide management of *A. planipennis* advances.

Area-wide programs that integrate multiple pest control tactics have been established for a number of prominent forest pests in North America. For example, integrated management programs for Southern pine beetle (*Dendroctonus frontalis* Zimmerman) combine various treatment options including augmentative biocontrol, pesticides, semiochemicals such as aggregation pheromones and utilization or felling and burning of infested host trees depending on site conditions, resources and management objectives (Coulson and Saarenmaa, 2011). Decades of research on gypsy moth (*Lymantria dispar* L.) have provided the foundation for successful Slow the Spread and suppression programs, which are implemented annually across municipal, county and even state boundaries (Sharov et al., 2002; Tobin et al., 2012). While current knowledge and management options for *A. planipennis* control are relatively limited, significant advances have been made since *A. planipennis* was first detected in North America (Herms and McCullough, 2014). Adopting an area-wide approach to *A. planipennis* management is likely to yield multiple benefits, and simulations suggest such an approach could substantially reduce costs (Kovacs et al., 2014). Overall results from the SLAM pilot project indicate that insecticide-treated trees and girdled trees are flexible and functional tools that offer considerable potential for the development of integrated area-wide management strategies of *A. planipennis*.

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