THE SOCIAL-ECOLOGICAL CONSEQUENCES OF EMERALD ASH BORER IN GEORGIA

by

MITCHELL A. GREEN

(Under the Direction of Kamal J.K. Gandhi)

ABSTRACT

The emerald ash borer (EAB) is an invasive wood-boring beetle that is causing widespread mortality of North American ash trees (*Fraxinus* spp.). I sought to examine the consequences of EAB's invasion through social and ecological research in Georgia. Through an intercept survey with park users, I found that risk perceptions influenced support for both biological and chemical control as management tactics. Recreation preferences also influenced support for both methods. I also found that EAB-induced dieback was greatest in trees in larger DBH classes across sites in Northeast Georgia. The change in mortality and dieback from 2022 to 2023 was higher in sites with greater ash basal area, dominance, and importance value, and lower site diversity. Results from these studies may help inform EAB management and monitoring efforts in the Southeast United States.

INDEX WORDS: Biocontrol, Chemical control, *Fraxinus*, Human dimensions, Invasive species, Resource concentration, Recreation, Tree diversity

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW: EMERALD ASH BORER INVASION HISTORY, DYNAMICS, MANAGEMENT TACTICS, AND SOCIAL SCIENCE APPROACHES

1.1. Emerald Ash Borer Invasion in North America

1.1.1 Emerald Ash Borer: Invasion History and Impacts

The emerald ash borer (EAB), *Agrilus planipennis* Fairmare (Coleoptera: Buprestidae), is an invasive woodboring beetle originating from eastern Asia that has caused widespread death of ash trees (*Fraxinus* spp.) in North America (Cappaert et al., 2005; Poland & McCullough, 2006; Klooster et al., 2014; Burr & McCullough, 2014). EAB was first detected in North America in 2002 near Detroit, Michigan (Haack et al., 2002); but dendrochronological analyses suggest that it was already present and killing trees in the early 1990s (Siegert et al., 2014). Since its introduction, EAB has spread rapidly to 36 different states in the United States and five Canadian provinces (Emerald Ash Borer Information Network, 2023; USDA-APHIS, 2021). However, it is likely present in many more areas and avoiding detection.

In its native range, EAB is a secondary pest and attacks stressed or dying trees (Liu et al., 2003, 2007). However, North American ash that are evolutionarily naïve to EAB can be colonized and killed even when healthy (Gandhi & Herms, 2010a; Poland & McCullough, 2006). Tree death is caused by larval feeding on the phloem and cambium tissue (Poland & McCullough, 2006) (Fig. 1.1A). As larvae feed, they form distinctive serpentine galleries that cut off the transport of nutrients throughout the tree (Wang et al., 2010) (Fig. 1.1B). Over the course of 2-3 years, infection signs such as thinning canopies, epicormic sprouting, and bark splitting become visible on affected trees, which die shortly after (Fig. 1.1C). Since it was first identified, EAB has killed >99% of black (*Fraxinus nigra* Marshall), green (*Fraxinus pennsylvanica* Marshall), and white ash (*Fraxinus americana* L.) in areas near the invasion epicenter (Klooster et al., 2014; Morin et al., 2017). There are slight differences in susceptibility and preference of North American ash species. For example, blue ash (*Fraxinus quadrangulate* Michx.) appears to be more resistant than other ash species (Tanis & McCullough, 2012). However, all North American species are at risk relative to their Asian congeners, and EAB threatens to extirpate the genus *Fraxinus* on a continental scale (Herms & McCullough, 2014). EAB has also been shown to be able to complete development on white fringetree, *Chionanthus virginicus* L., a smaller understory flowering tree that is in the same family (Oleaceae) as ash species (Cipollini, 2015). Although EAB larvae tend to perform worse on white fringetree (Olson & Rieske, 2019), the ability of EAB to use a different host may help it persist in North America after the ash resource has been largely depleted.

When EAB was first discovered in Michigan, it was quickly realized that there could be wide-ranging devastation to the North American ash resource if it spread uncontrolled (Haack et al., 2002; Cappaert et al., 2005). Ash is distributed throughout the eastern United States and there are an estimated eight billion trees nationwide (Poland & McCullough, 2006). Ash makes up about 2.5% of the forest trees in the United States and it is a vital component of certain ecosystems (e.g., riparian corridors), which could be uniquely affected by EAB (MacFarlane & Meyer, 2005). It is expected that EAB-induced mortality is likely to alter successional trajectories as non-ash tree species respond to the influx of light due to canopy gap formation when overstory ash die and fall (Flower et al., 2013). There is some evidence that maples (*Acer* spp.) may benefit from the release of competition with dying ash in certain ecosystems (Costilow et al., 2017). However, the exact effects for many ecosystems are difficult to predict. Direct impacts on arthropod communities are also expected due to gap formation and the accumulation of coarse woody debris on the forest floor (Gandhi et al., 2014; Jennings et al., 2017; Perry & Herms, 2017; Ulyshen et al., 2011). It appears as though arthropod diversity initially might decrease, but these effects may diminish over time as canopy gaps close (Gandhi et al., 2014; Jennings et al., 2017). In addition, there are certainly ash specialist arthropods that will be affected by a decrease in their host resource (Gandhi & Herms, 2010b).

Ash is often planted in urban settings and is an important tree in many cities where EAB has invaded (MacFarlane & Meyer, 2005). Urban trees provide multiple beneficial services such as cooling, pollutant removal, control of stormwater runoff, as well as mental and physical benefits (Donovan et al., 2013; Lee & Maheswaran, 2011; Tzoulas et al., 2007). The major costs associated with urban ash decline due to EAB are removal and replacement (Sydnor et al., 2007). Removal costs of urban ash trees alone have been estimated at between $20-$60 billion (Cappaert et al., 2005; Kovacs et al., 2010). Such costs are often borne by individuals or local governments (Aukema et al., 2011).

In parts of North America, ash is also a valuable timber commodity and is often used for cabinet making, tool handles, and baseball bats (Poland & McCullough, 2006). Aukema et al. (2011) estimated the annual damage borne by forest landowners due to loss of timber to be $60 million. Loss of residential trees also decreases property values and, in the same study, annual residential property loss values were estimated at $380 million per year (Aukema et al., 2011). Black ash wood is also used in the basketmaking traditions of many Native American tribes, something that goes back numerous generations. Since black ash is particularly susceptible to EAB, this invasion threatens an important cultural tradition that escapes monetary valuation (Herms & McCullough, 2014). The unabated spread of EAB, therefore, has multiple significant ecological, economic, and cultural effects.

1.1.2. Emerald Ash Borer: Life History

EAB's life cycle consists of progression through four larval instars after egg hatch in the summer (Cappaert et al., 2005; Herms & McCullough, 2014). As noted above, larvae feed underneath the bark of the tree. If larvae complete feeding (i.e., progress through all four instars) during the first growing season, they will bore in the xylem and overwinter in pupal chambers as J-larvae prior to pupation in the spring (Cappaert et al., 2005). If they do not progress through all four larval instars however, they will overwinter in their feeding galleries, and continue feeding in the spring until development is complete (Siegert et al., 2010). Adults emerge in the spring to complete compensatory feeding, mate, and lay eggs (Cappaert et al., 2005). Recent work suggests that peak adult activity in North Carolina occurs around late April to mid-May, and EAB phenology is likely very similar in Georgia (Bohannon et al., 2022).

EAB's larval development varies based on two main factors: climate and host stress. Experiments in EAB's native range demonstrated that a one year life cycle is more common in southerly areas of China with warmer climates (Wang et al. 2010). Additional evidence for the importance of climate in larval development comes from numerous studies in the United States. Data from New York indicate that <25% of larvae overwinter in their larval feeding galleries (i.e., complete a two year life cycle), while the rest overwinter as J-larvae (Jones et al., 2020). Also, Gould et al. (2020) demonstrated that the percentage of EAB larvae overwintering as J-larvae corresponds strongly with an accumulation of growing degree days. Early in the EAB invasion, observations of a two-year life cycle being common in low-density outlier populations lent credence to the idea that host stress is also a factor in EAB voltinism (Cappaert et al., 2005). Tluczek et al. (2011) demonstrated that larvae develop faster and are more likely to complete a one-year life cycle instead of a two-year life cycle on girdled green ash relative to ungirdled controls. In either warmer areas, or highly colonized and stressed trees, EAB will therefore develop through a one-year life cycle whereas in colder areas, or less stressed trees, EAB larvae will develop through a two-year life cycle. The overwintering J-larvae stage appears to be crucial to adult development since even though individuals that develop over two years may finish feeding as a fourth instar as early as June of the second summer, they always emerge after the second winter (Jones et al., 2020).

1.1.3. Emerald Ash Borer: Management Tactics

In general, upon arrival of a new invasive species, the best management option is eradication, which is the complete removal of all individuals in the population (Simberloff, 2014). As an invasive species spreads, eradication becomes less likely and management strategies must subsequently shift to containment or mitigation strategies to reduce economic harm. Attempts to determine if EAB could be eradicated were undertaken immediately upon discovery (Cappaert et al., 2005). The earliest efforts included establishing quarantine zones to limit the movement of ash wood (Cappaert et al. 2005). However, it was discovered that many trees did not exhibit external symptoms until 3-4 years after initial infestation, and officials realized that outlier populations were forming much more frequently than originally thought (Poland & McCullough, 2006). As outlier populations continued to form, it was determined that

EAB was spreading more widely than could be contained through quarantines (GAO, 2006). It was also demonstrated that EAB spread near the invasion epicenter was not hindered by community composition or diversity (Smith et al., 2015). In addition, natural enemy surveys soon revealed that despite some parasitism by native larval parasitoids, overall attack rates of native natural enemies were too low to control EAB on their own (Duan et al., 2013). Once it was evident that eradication efforts would not contain the spread of EAB, the primary management strategy at the federal level became biological control (USDA-APHIS 2021, McCullough, 2019).

Classical biological control involves the importation and release of natural enemies from a pest's native range as a means of population control (Van Driesche, 1994). It is one of the two methods being undertaken to combat the EAB invasion in the United States by federal agencies, in addition to research on breeding resistant ash lineages (Merkle et al., 2023; USDA-APHIS 2020). Surveying of EAB's native range for natural enemies began in 2003 and led to the identification of three Hymenopteran parasitoids that showed promise for biocontrol (Liu et al. 2003). Two of the wasps are larval parasitoids, *Spathius agrili* Yang and *Tetrastichus planipennisi* Yang, and the third is an egg parasitoid, *Oobius agrili* Zhang and Huang (Yang et al. 2005; Zhang et al. 2005). These three parasitoids were approved for release in 2007 in the United States. A fourth parasitoid from the northern edge of EAB's range in Russia, *Spathius galinae* Belokobylskij, was later approved for release in 2015 (Belokobylskij et al., 2012; Duan et al., 2018).

All imported larval parasitoids can only parasitize 3^{rd} and 4^{th} instar larvae (Belokobylskij et al., 2012; Liu et al., 2007; Yang et al., 2010). In addition, they cannot parasitize J-shaped larvae (prepupae) in pupal chambers since they are too deep in the sapwood (Wang et al., 2010).

The timing of 3rd and 4th instar availability changes with latitude since it is based on development, which corresponds with heat accumulation. Parasitoids for biocontrol must be selected that have synchronous emergence times with suitable 3rd and 4th instars. In more southern latitudes, it may be more appropriate to use *S. agrili*, since it consistently emerges later than EAB adults, a pattern that is hypothesized to be more synchronous with such a one-year life-cycle (Gould et al., 2020; Ragozzino et al., 2020). This delay in parasitoid emergence allows enough time for EAB adults to mate and lay eggs and have their offspring progress to the 3rd and 4th instar in areas where a one-year life cycle is common. *Tetrastichus planipennisi* and *S. galinae*, are asynchronous with a one-year host life cycle, since they tend to emerge early in spring (Jones et al., 2020). However, in areas where a two-year life cycle is common (e.g., northern states), these parasitoids would have susceptible EAB larvae available since overwintering 3rd and 4th instar larvae will be present early in the spring. It is expected that *S. agrili* or the egg parasitoid *O. agrili* (which is not constrained by larval development times) would be the only parasitoids used for biocontrol in Georgia. These parasitoids are currently being released as a part of the first biocontrol releases in Georgia (Supplement A).

Spathius agrili is a gregarious idiobiont ectoparasitoid of 3rd and 4th instar EAB larvae (Yang et al., 2005). Females will oviposit anywhere between 1-18 eggs per larvae (Yang et al., 2010). They possess a long ovipositor that should allow females to parasitize EAB larvae feeding on trees of all sizes (Yang et al., 2010). In its native range, *S. agrili* can lead to high rates of parasitism, up to 60% in some areas (Yang et al., 2010). All these factors should make it a promising candidate for classical biocontrol. However, *S. agrili* has rarely ever established in the northern United States (Bauer et al., 2015). It is hypothesized that this is due to asynchronicity between the emergence time of *S. agrili* in June and the availability of suitable 3rd and 4th instar larvae in colder climates (Jones et al., 2020). It may have its best chance of success in the warmer climates of the southeastern United States, which are more similar to the native range of the imported population in Tianjian, China (Gould et al., 2020). EAB in both Tianjian and in the southeastern United States are more likely to exhibit a one year life cycle (Wang et al., 2007).

Oobius agrili is a solitary egg parasitoid first discovered in 2004 in EAB's native range (Zhang et al., 2005). As an egg parasitoid, it is not constrained by the same variations in larval availability as the other parasitoids and needs only be present when EAB adults are mating and ovipositing. It has been used widely and has been successful in terms of establishment (Duan et al., 2011). *Oobius agrili* was released initially in Michigan shortly after the parasitoids were approved for release and life table analysis indicated that it was having a measurable effect on EAB populations (Duan et al., 2014). Another study confirmed that parasitism rates of EAB eggs by *O. agrili* were 28% and 11% in release and control sites in Michigan, respectively (Abell et al., 2014). One *O. agrili* female can parasitize up to 80 EAB eggs in her lifetime, and evidence from its native range indicates that they can lead to parasitism rates of up to 32% of EAB eggs in a given location (Wang et al., 2016).

Chemical control is a highly effective EAB management option that could be considered by local communities or private landowners with valuable trees to protect (Herms et al., 2019). Emamectin benzoate applied as a trunk injection in particular has been found to be highly effective at decreasing larval densities relative to controls (Smitley et al., 2010, McCullough et al., 2011). In addition, it can provide up to 2-4 years of protection even when EAB population levels are high (Smitley et al., 2010). Other effective insecticides include imidacloprid and dinotefuran, which can both be applied as soil drenches (Herms et al., 2019). One study showed that application of imidacloprid can act in concert with biological control (Davidson & Rieske, 2016). In this study, the egg parasitoid *O. agrili* and the larval parasitoid *T. planipennisi* were recovered from areas that had received imidacloprid soil drenches, suggesting that enough larvae and eggs remained to be parasitized. However, this may only be important for areas where the more effective emamectin benzoate is not used or available and where an active biocontrol release program is ongoing. Studies show that chemical control to protect individual trees is more cost effective than removal and replacement, and retains the ecosystem services of those trees (Vannatta et al., 2012). However, chemical control methods are not feasible over large geographic areas. Also, it is unclear if the public will be accepting of chemical control as a treatment option due to general unease with pesticide usage (Estévez et al., 2015; Norgaard, 2007).

1.2. Importance of Site Attributes in Invasion Biology

Site attributes and tree species diversity may limit the spread and impacts of an invasive insect herbivore into new environments multiple ways. It is believed that diverse forests are more resistant to multiple biotic and abiotic disturbances such as insect invasions, small mammalian herbivory, or fire and windstorms (Jactel et al., 2017; Kennedy et al., 2002). Regarding insect herbivory, the positive effect of tree diversity is known as “associational resistance”. Evidence for associational resistance dates to an experiment designed to measure the effects of biodiversity on herbivory in an agricultural setting. Collards (*Brassica oleracea* L.) grown in pure stands consistently had higher herbivore loads that those grown mixed in with diverse meadow vegetation (Root, 1973). The author posited that this is because specialist herbivores are more likely to locate and immigrate to pure stands dominated susceptible hosts. This is termed the resource concentration hypothesis, and it provides an explanation for patterns of associational resistance.

Another proposed mechanism for associational resistance is the enemies hypothesis, which states that natural enemies are more efficient at controlling herbivores in mixed stands (Letourneau et al., 2009). In this case, it is believed that natural enemies are more abundant in more diverse stands and therefore provide greater top-down control. Host plant apparency via neighbor effects also might be an important driver of associational resistance (Castagneyrol et al., 2013, 2014; Fernandez-Conradi et al., 2018). In a manipulative study where both apparency and diversity were changed in a plantation setting, Castagneyrol et al. (2013) found that herbivory decreased when oaks (*Quercus* spp.) were surrounded by taller neighbors (and therefore, were less apparent). They concluded that in mixed stands consisting of trees with different growth rates, host apparency might be an important and oft overlooked covariate of tree diversity as a driver of associational resistance.

The predictions of the resource concentration hypothesis have been tested and in certain cases have held true (Conner et al., 2014; Guyot et al., 2015; Jactel et al., 2006). For example, it was found that defoliation by the chestnut gall wasp (*Dryocosmus kuriphilus* Yasumatsu) was lower in mixed stands containing four tree species versus pure stands (Guyot et al., 2015). In addition, the density of pine bast scale (*Matsucoccus feytaudi* Ducasse) was lower in mixed stands than pure stands (Jactel et al., 2006). Those authors also found that the predatory anthocorid bug (*Elatophilus nigricornis* Zetterstedt) showed the opposite pattern (it was more abundant in mixed stands). These results were taken as supportive of the enemies hypothesis. However, a tropical study in Mexico showed that there were no increases in predatory spiders across a diversity gradient (Abdala-Roberts et al., 2015). This study also showed that densities of specialists tended to follow the predictions of the resource concentration hypothesis, although this was not true for generalists.

Seemingly opposite effects, resource dilution, have also been observed, where some specialist herbivores achieve much greater loads in areas where their host plant is less frequent (Otway et al., 2005). In that study, loading of insect specialists decreased with increasing host plant mass. Although that study took place in an agricultural setting, similar results have been demonstrated in forest settings (Plath et al., 2012). Plath et al. (2012) found that while the specialist chrysomelid beetle, *Walterianella inscripta* Jacoby, showed increasing density and damage on monocultures of their host plant *Tabebuia rosea* DC, the specialist Crambid moth, *Eulepte gastralis* Gueneé, was more common and caused more damage in mixed stands. The authors attributed this to differences in ovipositing behaviors by females of each species. *Walterianella inscripta* likely prefers monocultures since it is highly mobile, and monocultures provide more access to multiple hosts whereas *E. gastralis* spend its entire lifecycle on one tree and ovipositing females might select for isolated trees that provide “competitor free space” from other herbivores. These results suggest that resource concentration effects might not be generalizable to all specialist species despite a recent meta-analysis that showed that many specialists follow a resource concentration distribution (Jactel et al., 2021). Whether or not patterns of EAB mortality and dieback can be explained by the resource dilution hypothesis, or the resource concentration hypothesis might help managers make predictions of spread in newly invaded environments.

Forest structure might also play a role in resistance to invasion. It is widely known that fragmentation and edge effects positively influence a given locations susceptibility to invasion (Mack et al., 2000). Multiple studies have shown this effect for plants (Yates et al., 2004). A meta-analysis also concluded that edge habitats favor invasive insects (Caitano et al., 2020), however this study only generated an effect size from seven papers. It is possible that as EAB spreads across the landscape, fragmented habitats are more likely to be initially colonized and/or experience more damage. Other aspects of forest structure, such as total site density, might also contribute to a type of "biological inertia" across the landscape (Holle et al., 2003).

It remains to be seen if associational resistance or other stand characteristics play a role in EAB's spread throughout the landscape. EAB has killed >99% of overstory ash trees near the invasion epicenter in Michigan since its introduction (Klooster et al., 2014). Even though blue ash appears to be less impacted by EAB (Tanis & McCullough, 2012) all native ash species are susceptible. In addition, although some native parasitoids will utilize EAB as a host, parasitism rates are not high enough to control EAB even in mixed stands (Bauer et al., 2008). It thus seems that the spread of EAB is not hindered by any site attributes, which Smith et al. (2015) demonstrated in sites near the invasion epicenter in Michigan. They found that distance from the epicenter of the invasion was correlated with ash mortality, however such a relationship did not exist with any other site level characteristics, including diversity and density. In addition, Knight et al. (2013) found evidence to support the resource dilution hypothesis when they found that sites with more suppressed or intermediate ash (as opposed to codominant or dominant) exhibited much more rapid dieback indicating that underlying tree vigor is another important factor.

It is possible that ash will not escape EAB's host finding abilities as it progresses throughout Georgia. However southern forests contain different assemblages of tree species than northern forests (Dyer, 2006). This might be critically important as some studies have suggested that specific species compositions might be more important to associational resistance than just diversity (Castagneyrol et al., 2014). In addition, unlike the Northeast where ash may represent 5% of forest trees, it only comprises about 1% of Georgia trees based on USDA Forest Service, Forest Inventory and Analysis (FIA) data (Brandeis et al., 2016). This lower concentration of ash may be low enough that stand attributes and diversity might influence spread and therefore, mortality patterns across the landscape. Hence, we cannot assume that the similar patterns of mortality and dieback as observed in the northern United States also occurs in the southeastern United States It is crucial to analyze invasive species spread and impacts as they encounter novel ecosystems where they might experience different climatic conditions, host concentration, and natural enemy complexes.

1.3. Social Science Approaches to Invasive Species

Increasingly, scientists across various conservation related disciplines are acknowledging the myriad of benefits that the application of the social sciences to conservation studies can have (Bennett, et al., 2017; Marzano et al., 2017). Despite this, there still exists a sizeable gap between the amount of published papers that look at invasion science through solely an ecological lens and papers that look at invasion science through a social-ecological or social lens (Vaz et al., 2017). This gap does not exist due to lack of desire; a survey of scientists in Argentina showed a sizable mismatch between how much scientists state that integrated social and natural science research should occur and the number of actual published studies that integrate social sciences (Anderson & Valenzuela, 2014). Some studies have used social science approaches to look at public perceptions of invasive species, but many are focused on invasive plants or vertebrates (Kapitza et al., 2019; Novoa et al., 2017). Promisingly, there is evidence from studies like these that the public supports invasive species management, but individual awareness of invasive species may dictate support (Höbart et al., 2020).

Outside of the invasive species literature, there has been some research on the importance of public perception of management tactics for native insect pests. For example, one study found variations in public perceptions of spruce beetle, *Dendroctonus rufipennis* Kirby (Coleoptera: Curculionidae) outbreaks in communities across Alaska depended upon time since outbreak (Flint, 2006). Another study found large support for management of forest pests in two different provinces in Canada (New Brunswick and Saskatchewan), but noted differences in preferred management tactic depended upon the location (Chang et al., 2009). In that study, chemical control was not preferred by residents of either province, but residents of New Brunswick were less supportive of chemical control than residents of Saskatchewan. One study that specifically asked about management of a non-native insect pest, the eucalyptus snout beetle, *Gonipterus scutellatus* Gyllenhal (Coleoptera: Curculionidae), found that individuals tend to be more supportive of biological control, since it is perceived as a more natural solution to pest management (Jetter & Paine, 2004). This was also found in a more recent study that looked at potential public responses to EAB management in Europe as they prepare for its increased spread from Russia (Marzano et al., 2020). It is difficult to generalize about the acceptance of management tactics for insect pests since they might vary by stakeholder group, level of awareness, temporal distance since outbreak, or locations (Japelj et al., 2019; Qin et al., 2015). However, as noted above, it does appear that biological control is viewed more favorably when weighed against chemical control since it is viewed as more natural.

There are few studies that have looked explicitly at public perceptions of EAB biocontrol. However, there have been studies that have looked at perceptions of a large range of management tactics. One study found that certain communities around Chicago also saw chemical control as unacceptable, and they were more concerned with the high costs associated with it (Marzano et al., 2020). Schlueter & Schneider (2016) inquired about multiple management tactics to reduce EAB spread and found that park users in Minnesota preferred wood regulations, sanitation cutting, and progressive thinning over chemical treatment, complete harvest, and doing nothing. However, they importantly noted that visitor confidence in all management techniques was low. In addition, although wood regulations are present throughout the invaded range of EAB, they can be hampered by low levels of compliance (Daigle et al., 2019) and other studies have found that tree removal programs can be controversial when implemented without proper communication (Mackenzie & Larson, 2010; Porth et al., 2015). Progressive thinning also might not be an effective management tactic since EAB can locate and kill trees in even low density stands (Knight et al., 2013).

Acceptance of any management tactic to control non-native species depends in part on individual's risk perceptions of that tactic and other options proposed. Risk perceptions include measures of both risk severity (e.g., how serious a problem is) and sensitivity (e.g., how likely a risk is to occur) (Haimes, 2009; Hanisch-Kirkbride et al., 2013; Sjöberg, 2000). Risk perception analysis has proved to be an important tool in determining support for biosecurity measures and conservation management in general (Estévez et al., 2015; McFarlane, 2005). McFarlane & Witson (2008) showed that perceptions of ecological risk influenced support for controlling the bark beetle pest, Mountain Pine Beetle (MPB), *Dendroctonus ponderosae* Hopkins (Coleoptera: Curculionidae), in parks in Canada. Specifically, the more ecological risk due to MPB outbreak that participants perceived, the more supportive they were of control measures. In another study, risk perceptions were shown to influence support for biosecurity in the herpetological trade

(Pienaar et al., 2022). Structural Equation Modeling (SEM) in this study demonstrated that sensitivity to risk was positively correlated with support for biosecurity measures. Risk perceptions also play a crucial role in the acceptance of pest management techniques (Marzano et al., 2020; Qin, 2015; Steele & Pienaar, 2021). Utilization of risk perception analysis thus shows broad utility across ecological disciplines and issues, and can be an important tool in the human dimensions of invasive species management.

An important component of the risk perception literature that has emerged involves examining how emotional responses influence individual's evaluations of risk (Slovic et al., 2007). Defining this as the "affect heuristic", Slovic et al. (2007) demonstrated that across a variety of disciplines, individuals tend to use non-logical cognitions in decision making and that often, these are at odds with their determination of risk. For example, the more positive associations with a decision, the less likely an individual is to judge that decision as risky. In the invasion science literature, the affect heuristic might lead to individuals viewing charismatic invaders as posing less "risk". Indeed, this has been shown in multiple papers. Steele & Pienaar (2021) surveyed individuals in Florida through an online questionnaire and found that individuals tended to oppose management of charismatic non-native species (e.g., the chestnut-fronted macaw, *Ara severus* L.) while tending to support management of non-charismatic non-native species (e.g., the Asian swamp eel, *Monopterus albus* Zuiew). Another survey in Scotland found that individuals also tended to oppose control of non-native birds (Bremner & Park, 2007). Interestingly, they also found that respondents tended to oppose management of non-native rhododendron (*Rhododendron ponticum* L.) but they supported management of giant hogweed (*Heracleum mantegazzianum* Sommier & Levier) and Japanese knotweed (*Fallopia japonica*

Houtt.). These studies indicate that even across different taxa, individuals may still make judgements based upon the attractiveness of the species at question.

Awareness of management programs and invasive species might also influence support for management (Bremner & Park, 2007; Novoa et al., 2017; Steele & Pienaar, 2021). For example, individuals who had previously heard of invasive species management programs tended to be more supportive of management in Scotland (Bremner & Park, 2007). A review of the literature also demonstrated that individuals who were aware of the negative impacts of invasive species tended to be more supportive of management (Novoa et al., 2017). However, individual awareness might not be associated with actual ecological impacts, as a study of invasive aquatic species has demonstrated (Gozlan et al., 2013). It is important then to consider awareness of ecological impacts of invasive species when evaluating support for management since less known species might lead to lack of support for management.

1.4. Thesis Goals and Objectives

This thesis had two specific goals that will examine the natural and social sciences of EAB's invasion in Georgia as follows:

1. Our first goal was to determine if park users in Georgia are supportive of the two major EAB control measures, biological control, and chemical control. We had three specific objectives for this project. Our first objective (O1) was to determine if ecological risk perceptions (of both EAB and control methods) will predict park users' support for management. Our second objective (O2) was to determine if awareness or knowledge of EAB and invasive species in general will predict park users' support for management. Finally, our third objective (O3) was to examine whether attitudes towards trees and

insects determine park users' support for EAB management. All these objectives were evaluated through the implementation of an in-person survey designed to measure those constructs across parks in Georgia that have an active EAB infestation.

2. Our second goal was to determine if any site characteristics in terms of structure and diversity are associated with the impacts of EAB across the Piedmont region of Georgia. We accomplished this goal through three specific objectives. Our first objective (O1) was to evaluate if there are any spatial trends to EAB's spread in this region. Spatial trends of EAB impacts could be reflective of radial spread from a discrete epicenter of introduction (presumably Atlanta) to the state. Our second objective (O2) was to determine if any site attributes are correlated with EAB impacts. We accomplished this objective by quantifying EAB impacts in 2022 and 2023 across sites in the northeastern areas in Georgia and completing correlation analysis with site attributes that describe structure and diversity. Finally, our third objective (O3) was to determine if any stand attributes or diversity metrics are correlated with temporal progression of EAB impacts. To accomplish this, we evaluated the change in EAB impacts from 2022 to 2023 and determined if such changes in EAB impacts were correlated with the same site attributes from O2.

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Figure Legend

Figure 1.1 (A) EAB larva in feeding gallery. (B) Distinctive S-shaped galleries caused by EAB feeding shown on a dead tree in Gwinnett County, Georgia. (C) An ash tree in Clarke County, Georgia exhibiting moderate levels of dieback due to EAB.

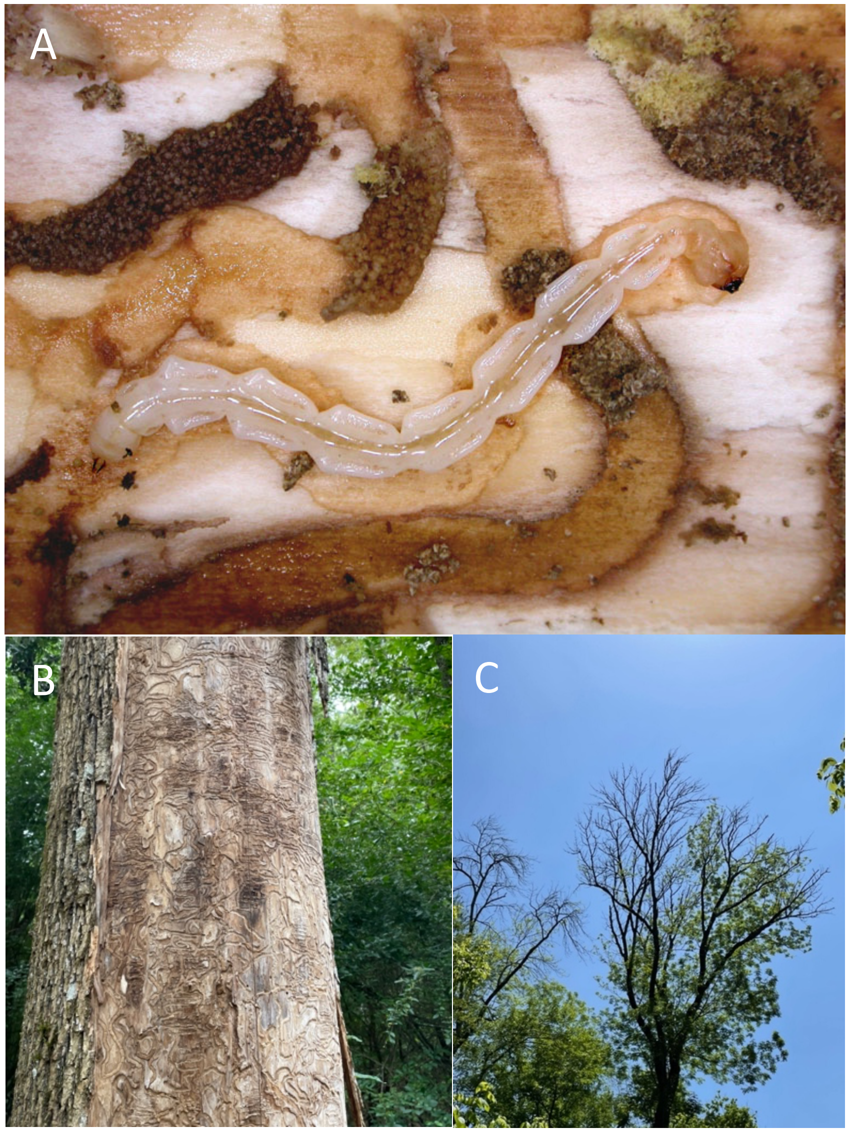

Figure 1.1

CHAPTER 2

EXAMINING PARK USERS' RISK PERCEPTIONS OF EMERALD ASH BORER (*AGRILUS PLANIPENNIS*) MANAGEMENT OPTIONS IN GEORGIA[1]

[1]Green, M.A., Barnes, B.F., Gandhi, K.J.K., and Pienaar, E.F. To be submitted to *Biological Invasions*.

Abstract

Emerald ash borer (*Agrilus planipennis*) (EAB) is a woodboring beetle that is considered one of the most damaging invasive forest insects in North America. It is causing near complete mortality of native ash (*Fraxinus* spp.) trees, leading to widespread ecological and economic effects. Management options include both biological control (biocontrol) through the importation and release of parasitoid wasps from EAB's native range, and chemical control, with systemic insecticides. While both strategies are used, there is a lack of knowledge on the social acceptability of these methods to relevant stakeholder groups. We determined levels of support for both management methods amongst park users in Northeast Georgia in areas recently infested with EAB. We then evaluated if support is associated with awareness, knowledge, attitudes towards ash trees and insects, risk perceptions of EAB and control methods, and demographics. We conducted in-person surveys with park users in four parks in northeastern Georgia in the spring of 2023. We received 174 usable surveys and completed ordinal logistic regression with our response variables as support for either biocontrol or chemical control, and our explanatory variables as awareness, knowledge, attitudes, risk perceptions, and demographics. In general, park users were more supportive of biocontrol than chemical control and perceived greater risk from chemical control. Perceived risk negatively influenced level of support for both biocontrol and chemical control. We also found that recreation preferences influenced support, whereby birdwatchers were less likely to support chemical control and individuals who like to sit and enjoy nature were more likely to support biocontrol. Individuals who expressed the highest levels of risk sensitivity to EAB induced ecological damage were more likely to support biocontrol, but not chemical control. Finally, more positive attitudes towards ash trees increased support for chemical control. Our results suggest that managing agencies seeking to control EAB through

either method may want to consider how individuals recreate in their parks and potentially adjust their messaging to reduce the perceived risk of both control methods as they are used in local parks.

Keywords

Biocontrol, Chemical control, Human dimensions, Invasive species, Recreation

2.1. Introduction

The invasion by non-native species into novel ecosystems is a global problem which can lead to wide-ranging ecological and cultural impacts (Mack et al., 2000; Simberloff, 2014). Invasive species are a major driver of loss of native biodiversity and ecosystem services (Boyd et al., 2013), and may indirectly impact human health (Donovan et al., 2013; Jones, 2017). Invasive species in the United States were estimated to cost $26 billion annually in economic damage during the 2010s (Crystal-Ornelas et al., 2021). Invasive insects are a particular concern as they cause ecological impacts (Kenis et al., 2009) and economic damage if they cause harm to valuable sectors such as forestry (Aukema et al., 2011). When a novel invasive insect arrives in an area, the first response option is eradication. However, eradication is not often possible and for established invasive insects and the response strategy must shift to management and mitigation (Simberloff et al., 2013).

One of the costliest invasive insects is the emerald ash borer, *Agrilus planipennis* Fairmare (Coleoptera: Buprestidae; EAB) (Aukema et al., 2011). EAB is an invasive wood-boring beetle that has led to the widespread mortality of North America's ash resource since it was first detected in the United States near Detroit in 2002 (Cappaert et al., 2005; Haack, 2006; Herms & McCullough, 2014). Mortality occurs as larvae feed beneath the bark, which disrupts the flow of nutrients and water and can lead to individual tree mortality in anywhere from two to five years (Klooster et al., 2014; Poland & McCullough, 2006). The loss of ash trees across North America has multiple ecological, economic, and cultural effects, necessitating both social and ecological approaches to fully understand the breadth of impacts from EAB (Aukema et al., 2011; Donovan et al., 2013; Herms & McCullough, 2014; Sydnor et al., 2007). EAB is currently present in 36 states and five Canadian provinces, with a newly detected population on the west coast in Oregon (USDA-APHIS, 2022). It is predicted that EAB will continue to spread in novel habitats ultimately infesting 16 North American ash species in their range thus, resulting in major economic and ecological damage to local forests and biodiversity (Aukema et al., 2011; Gandhi & Herms, 2010a, b; Klooster et al., 2014; Kovacs et al., 2010).

At a federal level, the management focus is currently on biological control using four hymenopteran parasitoids (*Spathius agrili* Yang, *S. galinae* Belokobylskij, *Tetrastichus planipennisi* Yang, *Oobius agrili* Zhang and Huang) imported from China (USDA-APHIS, 2021). Three of the wasps, *S. agrili*, *T. planipennisi*, and *O. agrili*, were approved for release in 2007, with the fourth (*S. galinae*) approved for release in 2015 (Duan et al., 2018). Since approval, these parasitoids have been released across 31 states and three Canadian provinces to protect North American ash trees in forested areas (Duan et al., 2023). Chemical control efforts using either imidacloprid or emamactin benzoate are also highly effective (Bick et al., 2018; Smitley et al., 2010). Emamectin benzoate and imidacloprid are both applied directly to target trees by either trunk injection, soil drench, or soil injection. These methods lead to the translocation of the chemical throughout the tree where it kills vulnerable EAB life stages (Herms et al., 2019; Mota-Sanchez et al., 2009). Chemical control is currently seen as an effective strategy for organizations seeking to protect a few high value trees or as part of an integrated pest management program (McCullough, 2019; Sadof et al., 2021).

Managing EAB induced ash mortality will be crucial for natural resource professionals as EAB makes its way into new areas of the country, such as the Southeast. EAB was first detected in the Southeast in 2010 in Tennessee (Emerald Ash Borer Information Network, 2023). In addition to its impacts in southeastern forests, EAB will also affect residential parks and urban ecosystems (Lovett et al., 2016). Urban and residential areas receive direct benefits from healthy park trees, including cooling, pollutant removal, and health and wellness benefits (Chiesura, 2004; Donovan et al., 2013; Tzoulas et al., 2007). EAB is also able to disperse down linear greenways (Jones et al., 2019), such as those often found in cities. Residential parks are important hotspots of biodiversity in urban and peri-urban environments (Blood et al., 2016; Nielsen et al., 2014), and diverse ecosystems are more resilient and may better protect ecosystem services (Boyd et al., 2013; Cardinale et al., 2012).

EAB's movement into the Southeast and the vulnerability of ash trees in urban and residential parks underscores the importance of examining if park users are supportive of EAB management (e.g., chemical control or biocontrol) in this area. Opposition to management may lead to conflicts that might delay mitigation efforts (Crowley et al., 2017). Also, if managers are not willing to manage an invasive pest, but stakeholders express concern, stakeholder frustration may precipitate action (Gozlan et al., 2013). Research on stakeholders' perceptions of invasive species and management options is necessary to anticipate and plan for possible conflicts prior to management activities (Bennett et al., 2017; Estévez et al., 2015; Marzano et al., 2017; Vaz et al., 2017).

Multiple factors may influence the perception of management by stakeholders, including risk perceptions of both the invader and management options (McDaniels et al., 1995). Risk perceptions are often defined as including components that capture the severity and probability of risk from a natural hazard (Gore et al., 2009, Haimes, 2009). Risk susceptibility is the likelihood of a risk occurring, for example, the likelihood that chemical control leads to off negative, off-target ecosystem effects. Risk sensitivity captures level of concern of a risk. In the case EAB management, an individual might express great concern for chemical or biological control having unintended consequences, even if they perceive a low probability of that occurring. Risk perceptions can be affected by emotional reactions or an individuals' values, where values are fundamental and stable individual cognitions that underpin attitudes and behavior (Fulton et al., 1996; Slimak & Dietz, 2006; Slovic et al., 2007). Risk perceptions are an important determinant of an individual's support for native pest and invasive species management (McFarlane & Witson, 2008; Steele & Pienaar, 2021) and insect pests in particular are perceived by the public as a risk to forest biodiversity (McFarlane, 2005). Support for biosecurity measures in the wildlife trade are also related to values and individuals' risk susceptibility and sensitivity (Pienaar et al., 2022).

Risk perceptions of chemical control reduce support for chemical pesticides as a management tactic (Höbart et al., 2020; McFarlane et al., 2006). This is true even in scenarios where the public in general supports pest management (McFarlane et al., 2006). Chemical control may be seen by stakeholders as less natural, and therefore less preferred than options such as biocontrol (Chang et al., 2009; Fuller et al., 2016; Jetter & Paine, 2004). However, stakeholders may still oppose biocontrol in certain social and cultural situations, such as those where the invasive species is valued by the public (Warner & Kinslow, 2013).

Awareness of an invasive species may also be an important determinant of support for management efforts (Bremner & Park, 2007). Awareness of invasive species is generally low, and the public tends to perceive less risk from invasives than professionals (Gozlan et al., 2013; Prinbeck et al., 2011). Invasive insects may be less visible and less charismatic than other invasive species such as bird species (Jarić et al., 2020). Multiple studies have shown that the public tends to oppose general management of more charismatic invasive species (i.e., flowering plants and birds), and is more knowledgeable about terrestrial vertebrate invasive species (Bremner & Park, 2007; Gozlan et al., 2013; Novoa et al., 2017; Steele & Pienaar, 2021).

Accordingly, we predict that knowledge and awareness may positively influence support for management. Attitudes towards management interventions and invasive species also influence management support, but management options such as poisoning may not be supported even when there is desire to control the target species, such as invasive brown rats (*Rattus norvegicus* Berkenhout) (Bremner & Park, 2007). In cases like this, if stakeholders perceive more risk from the control measures than the target invasive species, they might oppose management.

A few studies on acceptability of EAB management have demonstrated how the public responds management efforts. For example, recreationists in Minnesota were most supportive of management options such as sanitation cutting (removal of infested ash), wood regulations (preventing long-range movement of potentially infested ash wood), and progressive thinning (Schlueter & Schneider, 2016). However, sanitation cutting, and wood regulations have been in place since around EAB first arrived in the United States and have not eradicated it or slowed its spread. Also, progressive thinning is possibly a less optimal silvicultural option as EAB can still cause complete mortality in stands regardless of density (Smith et al., 2015). A tree removal program ahead of the EAB invasion in Ontario was also widely unpopular with affected residents (Mackenzie & Larson, 2010). It is important to ask stakeholders about both biological control with parasitoid wasps and chemical control with systemic insecticides specifically however, since those are the prevalent management options that could be used in residential parks.

Park users' support for management may be influenced by pertinent demographic information as well. It is possible that proximity to parks, frequency of use, and recreational activities engaged in may increase or decrease management support (Gobster, 2011). For example, we predict that park users may be less supportive of a control measure that they perceive as personally riskier if they live closer to the park where the intervention will occur, visit the park more frequently, or think that an intervention will affect them uniquely because of their recreation preferences.

We aimed to fill knowledge gaps by analyzing the level of support from county park users for both biocontrol and chemical control of EAB via in-person surveys in Georgia. Georgia is a state on the leading edge of the EAB invasion, with EAB present in the state since 2013 (GFC, 2022). As EAB continues to spread in its range and cause greater ash mortality, we have an opportunity to gauge how the public responds to this threat. We predicted that park users in Georgia will be more supportive of biocontrol than chemical management efforts and that this will be due to increased risk perceptions from chemical control. In addition, we predicted that increased risk perceptions of either control method would negatively influence support. We also predicted that higher levels of knowledge and awareness of EAB would increase support for either control method, as would positive attitudes towards ash trees. Finally, we predicted that positive attitudes towards insects would increase support for biocontrol. Results from this study may inform best outreach and communication efforts for EAB and help land managers in decision-making and message framing, as this devastating pest continues to invade new habitats in North America.

2.2. Methods

2.2.1. Survey Design

The University of Georgia Institutional Review Board reviewed our survey and determined it to be exempt (Project Number 00006160). Our project was approved on October 22nd, 2022. A pilot survey was pre-tested in December 2022 and January 2023 in Clarke County, Georgia. We pre-tested our survey with 25 participants who provided us with feedback regarding the clarity of questions. We also conducted pre-tests with two expert faculty members from the D.B. Warnell School of Forestry and Natural Resources at the University of Georgia. Our pilot survey was modified according to these pre-test results and a final draft survey was completed by January 2023.

Our final survey instrument consisted of five sections and was designed to take approximately eight minutes to complete. To determine respondents' stated support for either chemical control with systemic insecticides or biological control with parasitoid wasps, we asked them to "indicate their level of support" for each control method, which we evaluated on a five-point Likert-type scale: strongly oppose (1), oppose (2), neutral (3), support (4), strongly support (5).

To capture attitudes towards park trees, we asked participants how important (Not at all important = 1, Slightly important = 2, Moderately important = 3, Important = 4, Very important = 5) the following characteristics were to them: 1) that the park contains trees that provide shade; 2) That the park contains a variety of trees with different leaf shapes and colors; 3) That the park contains trees that flower; 4) That the park contains trees of different shapes and heights. We also asked participants if they had heard of ash trees to evaluate awareness of ash trees (No = -1, Unsure = 0, Yes = 1). We then asked respondents about their ability to rate trees on a scale of 1 to 10 where 1 = I cannot identify any tree species, and 10 = I am extremely confident in my ability to identify tree species. For individuals who had heard of ash and entered a 7/10 or higher for their ability to identify trees, we evaluated their knowledge of ash trees by asking them to select the picture of an ash out of four images, where the other trees shown were a ginkgo tree (*Gingko biloba* L.), a tulip poplar (*Liriodendron tulipifera* L.), and a sweetgum (*Liquidambar styraciflua* L.). Finally, to capture attitudes towards ash trees, we showed individuals a picture of an ash tree and asked how much they agreed with the following statement: “I would like it if my street had trees like this” (Strongly disagree = -2, Disagree = -1, Neither agree nor disagree = 0, Agree = 1, Strongly agree = 2).

We showed participants pictures of the following insects: butterflies, bees, ants, flies, wasps, mosquitoes, and beetles and asked how much they liked or disliked those insect groups (Strongly dislike = -2, Dislike = -1, Neutral = 0, Like = 1, Strongly like = 2) to capture attitudes towards insects. Awareness of invasive species was measured by asking participants if they had heard of the term invasive species before taking this survey (No = -1, I’m not sure = 0, Yes = 1). After presenting participants with a definition, we captured invasive species knowledge by asking if prior to taking this survey they were aware that non-native species were considered invasive if they cause: 1) harm to the environment; 2) economic damage; and 3) harm to people and pets (No = -1, I’m not sure = 0, Yes = 1). We also assessed awareness of EAB (No = -1, I’m not sure = 0, Yes = 1) through a question asking participants if they had heard of EAB prior to taking this survey. Individuals who indicated yes were then asked how confident they were that they could identify an emerald ash borer to capture knowledge of EAB (1 = Not at all confident, 2 = Slightly confident, 3 = Moderately confident, 4 = Confident, 5 = Very confident).

We included a brief explanation of the ecological risks associated with EAB and then asked participants “how concerned are you about the loss of ash trees from emerald ash borer in this park” to capture risk sensitivity to the ecological effects of EAB (1 = Not at all concerned, 2 = Somewhat concerned, 3 = Moderately concerned, 4 = Concerned, 5 = Very concerned). We then explained both biocontrol and chemical control and captured risk susceptibility to both control methods by asking participants what risk they thought (1 = No risk, 2 = Low risk, 3 = Moderate risk, 4 = High risk) that 1) wasps used for biocontrol will harm the environment; 2) wasps used for biocontrol will harm humans (stinging, allergies, etc.); 3) wasps used for biocontrol will harm pets (stinging, allergies, etc.); 4) applying chemicals in or around trees will harm the environment; 5) applying chemicals in or around trees will harm humans; and 6) applying chemicals in or around trees will harm pets (ingestion, contact, etc.).

We used similar scenarios to measure risk sensitivity to biocontrol and chemical control by asking respondents how concerned they were (1 = Not at all concerned, 2 = Somewhat concerned, 3 = Moderately concerned, 4 = Concerned, 5 = Very concerned) that 1) that wasps used for biocontrol will attack native species in the park; 2) wasps used for biocontrol will harm human health; 3) wasps used for biocontrol will harm pets; 4) chemical control will have unintended environmental consequences; 5) chemical control will harm human health; and 6) chemical control will harm pets. We finally asked respondents for demographic information including gender, age, distance that they live from the park, and number of times they visit the park. Finally, we asked respondents about recreation preferences, allowing them to select up to three of the following activities, walking/hiking, running/jogging, cycling, birdwatching, picnics/Barbeques, and sitting and enjoying being outside. These variables were binary coded such that 1 = that they do participate in the activity and 0 = they do not participate in the activity.

2.2.2. Survey Methods

We conducted surveys in four parks across Clarke and Gwinnett Counties, Georgia (Supplement B). EAB was first identified in both Clarke and Gwinnett Counties in 2015, two years after EAB was first found in the state of Georgia. Sandy Creek Nature Center was used as a study site in Clarke County and McDaniel Farm Park, George Pierce Park, and Yellow River Park were our study sites in Gwinnett County. All these parks contained ash as a forest component although they differed in the amount of ash (74-181 trees per ha), the size of ash trees (mean diameter at breast height ranged from 15.8 $\pm$ 0.39 to 45.1 $\pm$ 0.40 cm), and the level of ash mortality due to EAB (mortality ranged from 18.2 to 57.1%) (Chapter 3). In addition, the parks varied in total size from 54.2 (McDaniel Farm Park) to 279.6 (Yellow River Park) ha.

We administered surveys on both weekdays and weekends starting on January 23rd, 2023 and ending on April 12th, 2023. We varied the time of day that we administered surveys at each park. We expected that many park users would tend to visit these parks at similar times of day so failing to vary the times that we administered surveys could exclude potential respondents. While conducting surveys, we stationed ourselves in areas of high pedestrian traffic close to the entrances of the parks. We asked all adults (individuals who appeared over 18) that we encountered if they would like to participate in our survey. Participants completed surveys on iPads using Qualtrics Offline Survey app. We also created QR-coded postcards that were passed out to individuals at parks in the last three weeks of data collection to capture responses from individuals who might be interested in the survey but did not have the time to stop and take it on an iPad. The mean completion time was approximately eight minutes and 15 seconds across both methods. There was no compensation for agreeing to participate and participants were free to withdraw at any time.

2.2.3. Statistical Analyses

All analyses were conducted in R (v4.2.3; R Core Team, 2023). We calculated Cronbach's alpha (Cronbach, 1951) to determine whether survey items could be combined to generate social-psychological constructs (e.g., risk sensitivity, risk susceptibility), using the psych package (Revelle 2023). We considered values of Cronbach's alpha ≥ 0.7 to be adequate measures of the internal consistency of our measured constructs.

We then used exploratory factor analysis using the factanal function in the native R stats package to generate factor loadings for each question within a construct (Spearman, 1904; Watkins, 2018). Factors with eigenvalues>1 were retained. To generate a weighted score for our construct variables we took the sum of each response multiplied by its factor loading from exploratory factor analysis and then divided by the number of items within the construct. We did this for the following variables: attitudes towards park trees, attitudes towards insects, risk sensitivity to biocontrol, risk sensitivity to chemical control, risk susceptibility to biocontrol, and risk susceptibility to chemical control.

In addition to the above constructs, we generated an invasive species knowledge score based on three questions that asked respondents if they knew about the effects of invasive species. After confirming that the responses to these questions were correlated (Pearson correlation coefficients ranged from 0.624-0.715), we took the sum of the effects coded responses to each question (where 1 = yes, 0 = not sure, and -1 = no) divided by three to generate an invasive species knowledge score.

To evaluate whether respondents were more supportive of biocontrol or chemical control we conducted a Wilcoxon rank sum test. We were also interested in whether or not respondents perceived greater risk from biocontrol or chemical control so we compared the mean values of risk sensitivity and risk susceptibility scores using t-tests. Finally, we conducted ordinal logistic regression for each dependent variable (support for biocontrol and chemical control separately) using the MASS package (Venables & Ripley, 2002). To determine which terms should be retained we completed model selection using stepwise backwards regression and retained final models with the lowest AIC values.

2.3. Results

2.3.1 Respondent Characteristics

In total, we received 178 surveys from 412 approached individuals. Four surveys were incomplete, leaving us with a final data set of 174 surveys and an overall response rate of 42%. One-hundred and fifty-eight of our responses were completed on iPads and 16 were taken from QR codes. Out of all respondents, 81 identified as male (4.6%), 89 as female (51.1%), and 4 (2.3%) preferred not to say (Supplement C). The median age range for response was 45-54 years (Supplement C). On average our respondents visited the parks at which they were surveyed at five times in the past month and lived within ten minutes driving distance of the park (Supplement C). The three most common recreation preferences that respondents indicated they participate in when they use the parks were walking/hiking (158; 46.3%), sitting and enjoying nature (55; 16.1%), and running/jogging (53; 15.5%) (Supplement C). We found that most individuals had heard of ash trees (59.2%), yet on average did not rate their ability to identify trees highly (mean = 4.4 out of 10, where a 10 indicated they could identify all trees and 1 indicated they cannot identify any trees). Out of the individuals who rated their ability to identify trees high ($\geq$ 7 out of 10, n = 24) only 50% (n =12) correctly selected an ash tree on a question asking them to pick an ash out of 4 possible options. Most park users had heard of invasive species (86.2%), yet only 23 (13.2%) individuals indicated they had heard of EAB prior to taking the survey. Most respondents also indicated they were aware that non-native species were considered invasive if they caused harm to the environment (n= 138; 79.3%), economic damage (n = 126; 72.4%), and harm to people and pets (n = 122; 70.1%) (Supplement D).

2.3.2 Attitudes Towards Park Trees

Most respondents (n = 104; 59.8%) stated that having shade-providing trees in the park was very important to them. Fewer respondents indicated that having trees with different leaf shapes and colors in the park was very important (n = 80, 46.0%). Additionally, 42% of respondents stated that the park containing trees that flower was very important (n = 73); and 43.1% stated that the park containing trees that are different shapes and heights was very important (n = 75). Exploratory factor analysis demonstrated that these questions loaded onto a single factor that measured attitudes towards trees (eigenvalue = 2.78; Cronbach's alpha = 0.85) (median = 3.23; 3.13 $\pm$ 0.05; range = 0.77-3.86; Supplement E).

2.3.3 Attitudes Towards Insects

The median response for the attitudes towards butterflies question was "strongly like". The median response for the attitudes towards bees question was "like", whereas it was "neutral" for both ants and beetles. The median responses for the attitudes towards flies question and the attitude towards wasps questions were "dislike" and the median response for the attitudes towards mosquitoes question was "strongly dislike."

Our attitudes towards insect questions would not load onto a single factor during exploratory factor analyses unless the attitudes towards butterflies questions was excluded. We removed the butterfly question from the rest of the attitudes towards insects questions and generated an attitude towards insects score using only the six questions that loaded onto a single factor (eigenvalue = 3.33, Cronbach's alpha = 0.83) (median = -0.23; -0.20 $\pm$ 0.04; range = -1.36-1.36; Supplement F).

2.3.3 Perceptions of Control Methods

Respondents indicated low levels of risk susceptibility from biocontrol using parasitoid wasps. The median responses for risk susceptibility questions asking what risk respondents thought there was that "wasps used for biocontrol will harm the environment", "harm humans", and "harm pets" were all "low risk." Exploratory factor analysis revealed that these responses separated onto a single factor, which we termed "risk susceptibility to biocontrol" (eigenvalue = 2.35; Cronbach's alpha = 0.86) (median = 1.65; 1.80 $\pm$ 0.05; range = 0.82-3.30; Supplement G). Risk susceptibility to chemical control was slightly higher. The median responses for risk susceptibility questions asking what risk respondents thought there was that applying chemicals would "harm the environment", "harm humans", and "harm pets" were all "moderate risk." Exploratory factor analysis also revealed that these questions loaded onto a single factor, which we termed "risk susceptibility to chemical control" (eigenvalue = 2.59; Cronbach's alpha = 0.92) (median = 2.67; 2.55 $\pm$ 0.06; range = 0.89-3.57; Supplement G).

The median responses for questions asking how concerned respondents were that wasps used for biocontrol would "attack native species in the park", "harm human health", and "harm pets" were all "slightly concerned." Exploratory factor analysis revealed that these responses loaded onto a single factor, which we termed "risk sensitivity to biocontrol" (eigenvalue = 2.48; Cronbach's alpha = 0.96) (median = 1.96; 2.07 $\pm$ 0.07; range = 0.86-4.31; Supplement G). The median responses for questions asking how concerned respondents were that chemical control would "have unintended environmental impacts", "harm human health", and "harm pets" were all “moderately concerned.” Exploratory factor analysis revealed that these responses separated onto a single factor, which we termed “risk sensitivity to chemical control” (eigenvalue = 2.65, Cronbach’s alpha = 0.91) (median = 3.01; 3.01 $\pm$ 0.08; range = 0.91-4.53; Supplement G).

In addition, respondents perceived greater risks from chemical control compared to biocontrol. The mean risk susceptibility to chemical control score was significantly higher than the mean risk susceptibility to biocontrol (Fig 2.2 A) ($t = -9.795$, $df = 332.37$, $p < 0.001$). Similarly, the mean sensitivity to chemical control score was significantly higher than the mean sensitivity to biocontrol (Fig 2.2 B) ($t = -8.513$, $df = 342.98$, $p < 0.001$).

2.3.4 Support for Control Methods

Respondents indicated significantly higher levels of support for biocontrol compared to chemical control based on a Wilcoxon rank sum test (Fig. 2.3) ($W = 22{,}562$, $p < 0.001$). The median score for the question asking respondents to indicate their level of support for biocontrol question was a 1, which in our survey corresponded with the response “support”. The median score for the question asking respondents to indicate their level of support for chemical control was 0, which corresponded with the response “neutral” (Table 2.1).

2.3.5 Ordinal Logistic Regression Analysis of Support for Control Methods

We found that respondents who expressed higher risk sensitivity to biocontrol ($\beta_{est} = -0.433$, $t = -2.382$, $p = 0.017$) and were more susceptible to risk from biocontrol ($\beta_{est} = -0.899$, $t = -3.128$, $p = 0.002$) were less likely to support biocontrol (Table 2.2). Individuals who use the parks to sit and enjoy nature were more likely to support biocontrol ($\beta_{est} = 0.893$, $t = 2.755$, $p = 0.006$). Other terms included in our final model included attitudes towards ash, gender, and preference for birdwatching, yet they were not significant (range of p-values = 0.074-0.300). Respondents who answered that they were less than "very concerned" about EAB were all significantly less likely to support biocontrol (concerned about EAB $\beta_{est} = -0.767$; moderately concerned about EAB $\beta_{est} = -1.153$; somewhat concerned about EAB $\beta_{est} = -1.446$; not at all concerned about EAB $\beta_{est} = -3.479$).

We found that individuals who were more sensitive to risk from chemical control ($\beta_{est} = -0.512$, $t = -2.510$, $p = 0.012$) and expressed more susceptibility to risk from chemical control ($\beta_{est} = -0.500$, $t = -4.985$, $p < 0.001$) were less likely to support chemical control as an intervention (Table 2.2). Birdwatchers were also less likely to be supportive of chemical control ($\beta_{est} = -0.899$, $t = -2.500$, $p = 0.012$). Finally, individuals who expressed more positive attitudes towards ash trees were more likely to support chemical control ($\beta_{est} = 0.340$, $t = 2.003$, $p = 0.045$).

2.4. Discussion

Our results have important implications for natural resource managers who may want to understand stakeholder perception of EAB management. For example, park users were more supportive of biocontrol than chemical control and this appeared to be due to increased risk sensitivity and susceptibility to chemical control. Contrary to our predictions, we found that neither knowledge nor awareness were significant predictors of support for either control method. We also found that positive attitudes towards insects were not significant in either control model. Attitudes towards ash trees positively influenced support for chemical control, but not biocontrol. Risk sensitivity to EAB was also significant, such that individuals who were less than "very concerned" about EAB were less likely to support biocontrol. Finally, recreation preferences appear to be important indicators of support. Individuals who indicated that they birdwatch when they visit the parks were less likely to support chemical control and individuals who indicate that they visit the parks to sit and enjoy nature were more likely to support biocontrol.

Other studies have reported similar results to ours, where individuals perceive greater risk from chemical control than biocontrol. Chang et al. (2009) indicated that chemical control ranked last in a list of preferred control options for two Canadian pests, spruce budworm (*Choristoneura fumiferana* Clemens) and forest tent caterpillar (*Malacosoma disstria* Hübner) in different provinces. In addition, chemical control was one of three control options rated as unacceptable for EAB management in state parks in Minnesota, whereas biocontrol was seen as much more acceptable (Schlueter & Schneider, 2016).

Our models suggest these results are due to the perceived risk of these two control methods. Risk perceptions are known to play a role in individual support for environmental interventions across various environmental fields, from invasive weed management (Norgaard, 2007) to management of chronic wasting disease (Harper et al., 2015; Vaske et al., 2018). Thus, it is unsurprising that perceptions of risk from EAB management play an important role in determining levels of support. Others have shown that the public, as compared to conservation professionals, tends to be more concerned about low-probability but high-risk events (Slimak & Dietz, 2006). It is likely that conservation professionals would view both biocontrol and chemical control through systemic insecticides as lower risk than the public since the parasitoid wasps used for EAB biocontrol and systemic insecticides are known to have few major non-target effects (Duan et al., 2018; Hahn et al., 2011; Herms et al., 2019). We did not test for differences between park users' and managers' perceptions of risk, but we suggest that managers may not presume that park users will perceive the same levels of risk as they do when it comes to control methods. Managers seeking to control for EAB may then choose to adopt persuasive communication strategies aimed at reducing the perceived risk of control in parks where they seek to use either biocontrol or chemical control as an intervention (Bier, 2001).

Interestingly, both of our final models included terms for recreation activities. Park users with specific recreation preferences such as birdwatching and sitting and enjoying nature might be more attuned to the natural world in ways that influence how they feel about management interventions. Although we did not test for values, it is likely that the shared values of individuals within these recreational groups might influence their support for certain management interventions via their risk perceptions (Gore et al., 2009; Steg et al., 2014). Birdwatchers, for example, might perceive a greater risk from chemicals used to treat trees causing off-target effects in the food web that might affect bird diversity. Biospheric values involving respect for nature might mediate their risk perceptions and disposition to not intervene (de Groot & Steg, 2008). Individuals who sit and enjoy nature may want to protect ash trees for their own benefits, perhaps mediated by altruistic values. Also, they might perceive a risk of coming in direct contact with chemicals if they aren't applied properly, and thus, would rather have parasitoid wasps released in their parks to protect ash trees. Improper training and application have come up as a concern amongst the public in focus groups about EAB control (Marzano et al., 2020). Future studies on the human dimensions of invasive insect management may want to examine if recreation preferences interact with values as described above. If such relationships exist, they may help explain why certain groups vary in their support of management interventions.

Risk sensitivity to ecological damage from EAB was a significant term in our model for support for biocontrol but not for chemical control. Individuals who responded less than "very concerned" about EAB were significantly less likely to support biocontrol. Importantly, these results may suggest that increasing perceived risk from EAB may increase support for biological control. For example, if managers seek to use biocontrol, they may want to adopt messaging strategies that speak to park users about the ecological risks due to EAB. Again, persuasive communication strategies that highlight the cascade of ecological effects due to EAB induced ash mortality could be used (Bier, 2001).

Our results also suggested that attitudes towards at-risk species might influence support for intervention. Park users who agreed more with a statement indicating that they would like ash trees on their streets were more likely to support chemical control. It is possible that this question captured individuals who have more positive attitudes towards trees in general, rather than ash trees *per se*. Since many people (41.9%) had not heard of ash, the fact that attitudes towards ash as street trees positively influenced support for chemical control may demonstrate that individuals responded positively to the aesthetic characteristics of the picture of the ash tree that they were shown (e.g., shape, color, etc.) Respondents also might recognize the importance of trees such as ash in providing things like shade, since many individuals indicated that trees that provide shade in their parks is very important. In this sense, it is possible that respondents recognized the importance of protecting greenspace trees in general from forest pests. Taken together, this suggests that messaging around protecting ash trees that provide shade or have characteristics that park users find appealing, combined with visual stimuli, may be an important component of EAB management communication.

Terms that were not included in our final models also provided information about what matters to park users regarding their levels of support for EAB management. For example, we hypothesized that positive attitudes towards insects could predispose individuals to supporting biocontrol. However, insect attitude scores were not included in our best fit models and did not influence support for either intervention. Interestingly, neither awareness of EAB nor knowledge of invasive species were included in our final models, although, awareness of EAB was quite low (only 13.2% of respondents had indicated that they had heard of EAB prior to taking the survey). Nonetheless, these results suggested that simply increasing park users' awareness of EAB or knowledge of invasive species might not increase their support for management, as has been demonstrated in other studies (Novoa et al., 2017). With our other results, the best messaging strategies would likely be those that combine risk messaging, positive visual stimuli, and speak to how doing nothing could affect recreation (for example, keeping ash trees alive as shade trees that would also protect bird habitat).

Although chemical control and biocontrol might be used in conjunction with one another as part of an integrated pest management strategy (McCullough, 2019), they generally operate at different spatial scales. Parks with a large, forested area containing ash would likely not opt for chemical control as it is logistically and financially challenging, whereas parks with only a few, high-value ash trees would not opt for biocontrol. Our results still showed that decision makers may need to be cognizant of how recreationists use their parks and adopt that into their messaging strategy. For example, in a case where a park manager seeks to protect a single, high value ash tree because it provides shade in an open area, they might want to speak towards how chemical control protects ash trees as bird habitat to avoid pushback from birdwatchers who are less likely to support chemical control.

As EAB continues to spread into new areas of the United States, agencies will be making decisions about ash resource management. The results of this survey may guide those responsible for making decisions to consider more refined and nuanced messaging about risk perceptions when undertaking EAB management through either chemical or biological control. A lack of appropriate message framing may lead to conflict between managers and park users (Crowley et al., 2017), which may hamper management efforts for high-impact invasive species and conservation efforts to protect and sustain our native forested ecosystems.

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Table 2.1 Survey respondents' support for two potential management options (biological control using parasitoid wasps and chemical control with systemic insecticides) for emerald ash borer in parks across Northeast Georgia (n = 174). Respondents answered the question "Please indicate your level of support for the following two control measures."

Control Method	**Median**	**Strongly Oppose**		**Oppose**		**Neutral**		**Support**		**Strongly Support**	
		Number	**%**	**Number**	**%**	**Number**	**%**	**Number**	**%**	**Number**	**%**
Biological control through the use of parasitoid wasps	Support	2	1.2	11	6.3	62	35.6	68	39.1	31	17.8
Chemical control through the use of systemic insecticides	Neutral	36	20.7	43	24.7	53	30.5	33	19.0	9	5.2

Table 2.2 Ordinal logistic regression models for respondents support for either biological control of emerald ash borer using parasitoid wasps or chemical control of emerald ash borer with systemic insecticides in Northeast Georgia.

Variables	Support Biocontrol	Support Chemical Control
Attitudes towards ash	0.265	0.340*
Risk sensitivity to control method	-0.433*	-0.512*
Risk susceptibility to control method	-0.899**	-1.500***
Concerned about EAB[a]	-0.767*	-
Gender[b]	0.428	-
Moderately concerned about EAB[a]	-1.153*	-
Not at all concerned about EAB[a]	-3.479***	-
Recreation preferences - Birdwatching	-0.669	-0.899*
Recreation preferences - Sitting and enjoying nature	0.893**	-
Somewhat concerned about EAB[a]	-1.446**	-
β_1	-8.139***	-7.214***
β_2	-5.996***	-5.407***
β_3	-3.171***	-3.355***
β_4	-0.807	-1.204*
Log Likelihood	-187.552	-208.626
AIC	403.104	435.253

$^{*}p<0.05$, $^{**}p<0.01$, $^{***}p<0.001$

[a] Binary coded variables in our regression, where the reference level = Extremely concerned about EAB.

[b] Coded as female = 1, Male/Prefer not to say = 0.\

Figure Legend

Figure 2.1. Conceptual diagram with possible ways in which independent variables might influence dependent variables (support for control) for emerald ash borer.

Figure 2.2 Differences in risk perception scores across all respondents through an intercept survey in county parks in Northeast Georgia. (A) Respondents expressed more susceptibility to risk from chemical control using systemic insecticides than biocontrol with parasitoid wasps for emerald ash borer management. **** indicate $p < 0.001$ (B) Survey takers also expressed more risk sensitivity to chemical control than biocontrol. ****indicate $p < 0.001$.

Figure 2.3 Comparison of the median support levels for both biocontrol with parasitoid wasps and chemical control using systemic insecticides for emerald borer management in county parks in Northeast Georgia, respondents expressed greater levels of support for biocontrol than chemical control. **** indicate $p < 0.001$ for Wilcoxon rank sum test ($W = 22{,}562$).

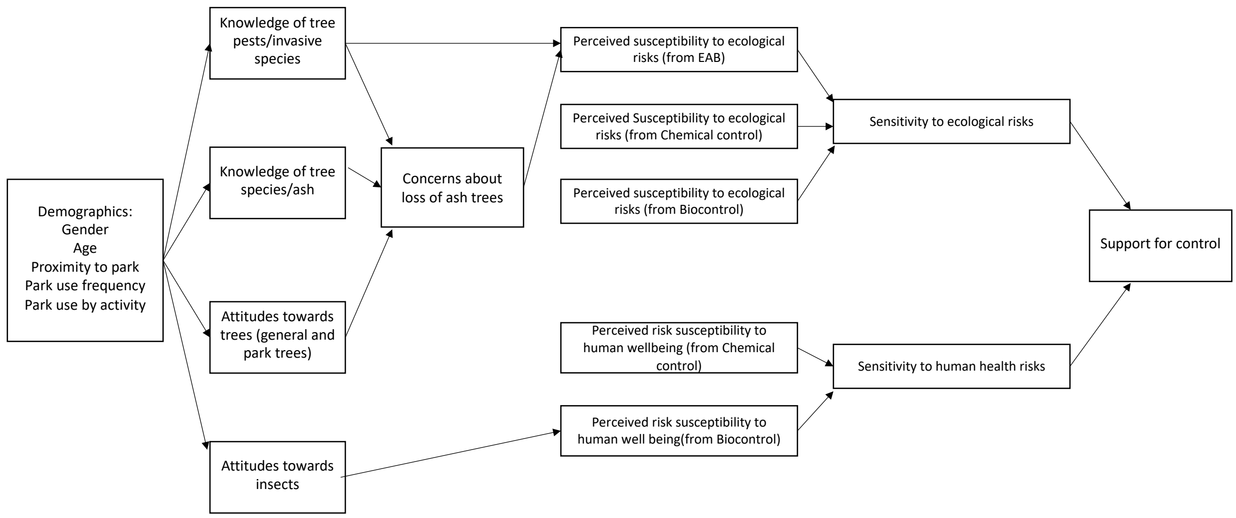
Demographics:
Gender
Age
Proximity to park
Park use frequency
Park use by activity
Knowledge of tree pests/invasive species
Knowledge of tree species/ash
Attitudes towards trees (general and park trees)
Attitudes towards insects
Concerns about loss of ash trees
Perceived susceptibility to ecological risks (from EAB)
Perceived Susceptibility to ecological risks (from Chemical control)
Perceived susceptibility to ecological risks (from Biocontrol)
Perceived risk susceptibility to human wellbeing (from Chemical control)
Perceived risk susceptibility to human well being(from Biocontrol)
Sensitivity to ecological risks
Sensitivity to human health risks
Support for control

Figure 2.1

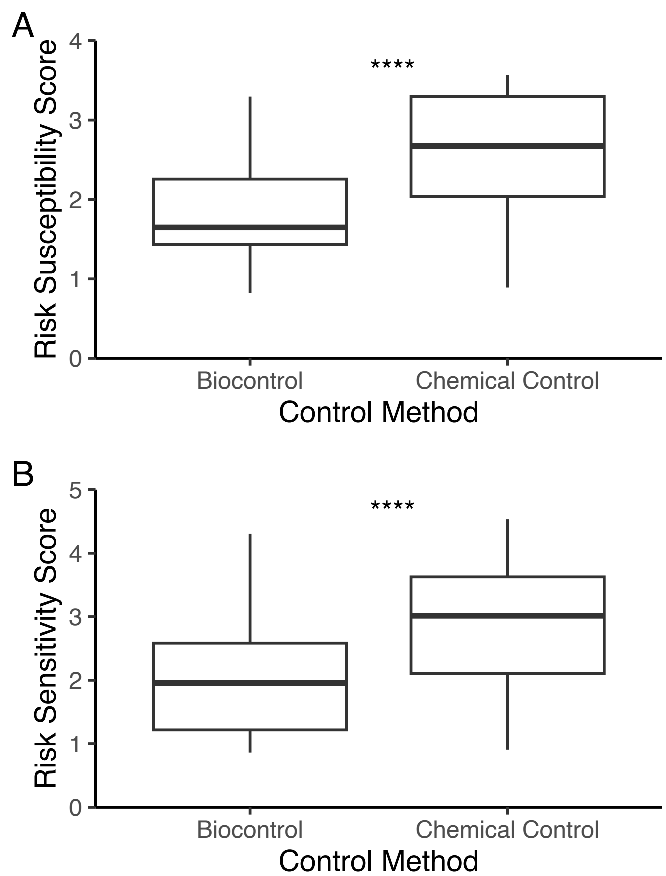
A
Risk Susceptibility Score
0
1
2
3
4
****
Biocontrol
Chemical Control
Control Method
B
Risk Sensitivity Score
0
1
2
3
4
5
****
Biocontrol
Chemical Control
Control Method

Figure 2.2

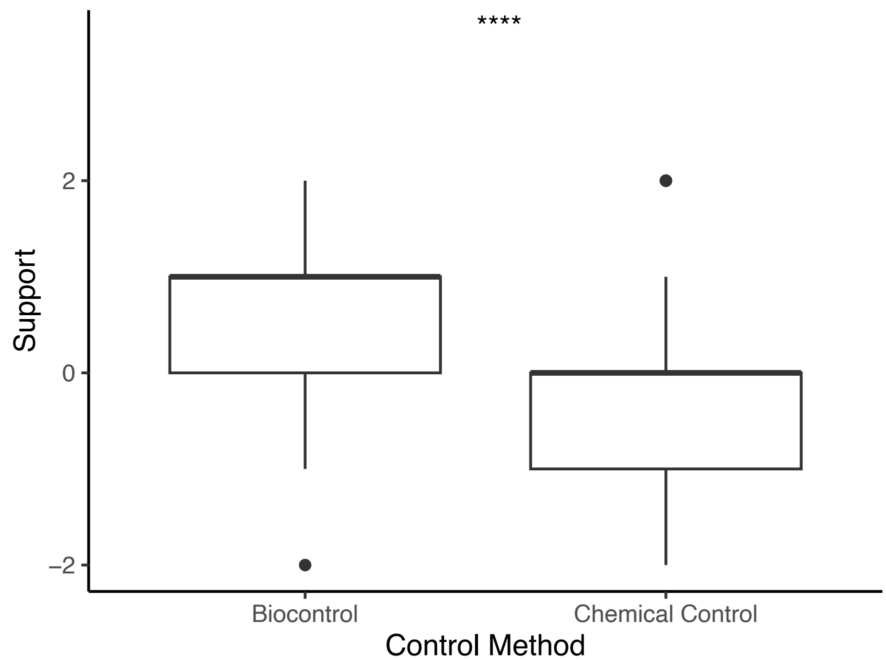

Figure 2.3

CHAPTER 3

EFFECTS OF SITE CHARACTERISTICS ON EMERALD ASH BORER (*AGRILUS PLANIPENNIS*)-INDUCED MORTALITY AND DIEBACK IN NORTHEAST GEORGIA[1]

[1]Green, M.A., Barnes, B.F., and Gandhi, K.J.K., To be submitted to *Environmental Entomology*.

Abstract

The emerald ash borer (*Agrilus planipennis*) (EAB) (Coleoptera: Buprestidae) is an invasive insect pest whose continued spread in the eastern United States has led to the near complete mortality of native ash (*Fraxinus* spp.) trees. EAB was first detected in the United States in Michigan, and accordingly, most research has taken place in the Midwest and Northeast states. However, forest dynamics and structure, particularly ash composition, change drastically in the Southeast. Therefore, we aim to fill knowledge gaps by determining factors associated with EAB-induced ash mortality and dieback in sites in Northeast Georgia. We established plots at 15 sites in the Piedmont region of Georgia to determine if EAB impacts (tree dieback rating, percent mortality, and percent trees with EAB signs) were correlated with site characteristics (e.g., tree density, basal area, and diversity) . In 2022 and 2023, ash trees in two of the largest DBH classes had greater dieback than trees in the smallest DBH classes. We did not find any site characteristics correlated with EAB dieback and mortality in 2022 or 2023. However, annual change in site dieback rating and percent mortality were positively correlated with ash density, basal area, dominance, and importance value. Similarly, annual change in site dieback rating was negatively correlated with tree richness, diversity, and evenness. The annual change in percent mortality across sites was also negatively correlated with evenness. These results suggest that over a year, EAB-induced mortality and dieback may proceed quicker at sites with an abundance of ash and lower overall tree diversity in the southeastern Piedmont region. Forest managers may choose to give priority to these sites for monitoring efforts as EAB continues to spread across the landscapes in southern regions.

Keywords

Forest pest, *Fraxinus*, Invasive species, Resource concentration hypothesis, Tree diversity

3.1. Introduction

Invasive species are a pervasive and chronic conservation issue impacting the biodiversity and function of native ecosystems worldwide (Simberloff et al., 2013). Some established invasive species can cause economic damage through costs associated with management and mitigation actions. Management of the worst species can cost upwards of millions of dollars in expenditures per year (Aukema et al., 2011) and lead to potentially irreversible ecological effects (Gandhi & Herms, 2010). For example, the invasive hemlock woolly adelgid, *Adelges tsugae* Annand (Hemiptera: Adelgidae), can lead to 90% hemlock mortality in affected stands (Lovett et al., 2016), causing impacts that can range from increased invasive plant abundance to changes in nutrient cycling (Eschtruth et al., 2006; Jenkins et al., 1999). The invasive spongy moth, *Lymantria dispar* L. (Lepidoptera: Erebidae), has been present in the United States since the 1880's and its episodic outbreaks lead to significant defoliation and mortality of susceptible hosts such as oaks (*Quercus* spp.) (Davidson et al., 1999). Estimates suggest that such high impact invasive insects accumulate in the United States at a rate of approximately one every two years (Aukema et al., 2010), necessitating the continued evaluation of invasive insect impacts on native forests.

One of the highest impact invasive forest insects to invade the United States in recent years is emerald ash borer (EAB), *Agrilus planipennis* Fairmaire (Coleoptera: Buprestidae), a woodboring beetle that has caused the death of hundreds of millions of ash trees (*Fraxinus* spp.) across the eastern and central United States since its introduction (Cappaert et al., 2005; Emerald Ash Borer Information Network, 2023a; Poland & McCullough, 2006). Every ash species in North America is considered susceptible to EAB, including black (*Fraxinus nigra* Marshall), green (*F. pennsylvanica* Marshall), and white (*F. americana* L.) ash (Cappaert et al., 2005). Blue ash (*Fraxinus quadrangulata* Michx.) appears to be more resistant than on North American ash species, although it is less widely distributed and EAB can still kill otherwise healthy blue ash trees (Tanis & McCullough, 2012). EAB was first detected near Detroit, Michigan in 2002, but it was likely present in that area in the mid 1990s (Haack et al., 2002; Siegert et al., 2014). It is now present in 36 states and five Canadian provinces, encompassing most of the eastern and central United States and Canada, and recently reaching Oregon in 2022, likely via movement of infested wood (USDA-APHIS, 2022). It is expected that as EAB continues to expand its distribution range, more ash trees will die across the landscapes (Herms & McCullough, 2014).

EAB kills ash through larval feeding on the phloem and outer xylem, cutting off the transport of nutrients throughout the tree (Poland & McCullough, 2006). Although EAB is a secondary pest in its native range throughout East Asia, it attacks and kills healthy trees in North America (Liu et al., 2003, 2007; Gandhi & Herms, 2010). It is possible that EAB induced ash mortality could lead to the extirpation of North American ash as canopy trees succumb to EAB (Herms & McCullough, 2014). This rapid change could result in altered successional trajectories, changes in nutrient cycling, possible co-extinctions of ash mutualists, and changes in overstory and understory plant composition (Engelken et al., 2020; Flower et al., 2013; Gandhi & Herms, 2010).

A critical question in invasion biology concerns whether certain ecosystems are more susceptible to invasion and subsequent effects such as host plant mortality. On a continental scale, the diversity of trees in North America may make it more susceptible to non-native phytophagous insects (Liebhold et al., 2013; Niemalä & Mattson, 1996). However, there may be associational resistance at smaller scales where increased plant diversity reduces ecosystem invasibility and host tree vulnerability to insect herbivores (Jactel et al., 2006, 2021; Kennedy et al., 2002). Multiple mechanisms have been proposed that would explain associational resistance patterns, including that natural enemies are more prevalent and effective in diverse forests (Letourneau et al., 2009), and/or reduced host tree apparency (either chemical or visual) in diverse forests can hamper phytophagous insect host searching (Castagneyrol et al., 2013; Kerr et al., 2017; Zhang & Schlyter, 2004).

Similarly, site attributes like increased host plant density have been hypothesized to lead to increased population densities of insect herbivores, as they are more likely to migrate to and stay in areas of high host density (Root, 1973). By extension, it has also been hypothesized that this should lead to increased impacts due to herbivory (Carson et al., 2008; Long et al., 2003). These findings are not universal, as some research has shown increased herbivorous insect loads in less dense host patches, e.g. the resource dilution hypothesis (Otway et al., 2005). Greater insect loading on a per plant basis in less dense patches may lead to greater levels of damage on their host plants, contradictory to the predictions of the resource concentration hypothesis. The effects of density and diversity are not necessarily mutually exclusive. For example, high diversity sites may have lower host density for a single focal species, and therefore, resource concentration and associational resistance might be confounded. However, contrasting effects are still possible depending on whether site density effects or diversity effects are predominant factors affecting herbivorous species impacts and distributions (Damien et al., 2016).

There are limited studies testing the effects of site characteristics on the invasion process of EAB. Smith et al. (2015) demonstrated that site attributes and community composition do not influence susceptibility to EAB near the epicenter of invasion in Michigan. Another study in Ohio reported that at lower density sites, ash trees exhibited faster mortality over five years, which was taken as evidence for resource dilution effects (Knight et al., 2013). As EAB spreads into new areas of North America (e.g., the southeastern and western United States) with different climates and tree species compositions, it is possible that different patterns of mortality may be observed across the landscape. Mitigating the effects of EAB in new areas such as the Southeast requires an understanding of possible differential affects across diverse habitats.

In the southeastern United States, EAB was first detected in Tennessee in 2010 (Emerald Ash Borer Information Network, 2023b). EAB was first detected in Georgia in 2013 in Dekalb and Fulton counties (GFC, 2022) which are in the northcentral area of the state. The exact epicenter of its arrival in Georgia is not definitively known, and those two counties are highly urbanized, so it is possible that it was already present in other counties and avoided detection for several years as reported in other studies (Siegert et al., 2014). In contrast to some northern states, ash only comprises about 1% of forest trees in Georgia (from USDA Forest Service, Forest Inventory Analysis) (Brandeis et al., 2016) and co-occurs with different tree species (Dyer, 2006). Both differences may influence important aspects of EAB's invasion dynamics. Our goal was therefore, to assess the spatial and temporal invasion patterns of EAB in southeastern United States forests. We had three specific objectives in Northeast Georgia as follows: 1) to determine the spatial trends to EAB's spread in northeastern Georgia; 2) to assess if EAB induced mortality and dieback was correlated with site characteristics; and 3) to determine the temporal trends of EAB dieback and mortality across sites during 2022-2023. Results from this study may guide monitoring and management decisions as related to site characteristics in EAB-invaded sites in the southeastern region.

3.2. Methods

3.2.1. Site Selection

During 2022-2023, we sampled 15 sites in seven counties in northeastern Georgia where ash was present (Supplement I). Across all our sites, green ash was the most dominant species, reflective of the fact that most sites were low lying bottomlands (Fig. 3.1A). Other common overstory trees across the sites included sweetgum (*Liquidambar styraciflua* L.) and red maple (*Acer rubrum* L.). American hornbeam (*Carpinus caroliniana* Walter) and privet (*Ligustrum sinese* Lour.) were both common in the understory across all sites (Fig. 3.1B). All sites were located in the southern outer Piedmont ecoregion, where inceptisols and entisols are the common soil types with red clay subsoils (Griffith et al., 2001). These sites receive an average of 129.1 ± 0.9 cm precipitation per year, with an average annual temperature of 16.5 ± 0.06 ° C (PRISM Climate Group, 2023).

3.2.2. Site Sampling

Within each site, we established three 10-m radius plots along a transect (Supplement J). We placed our first plot intentionally in an area where ash was present because of its heterogenous distribution in our sites. We then placed our second and third plots >50 m away around the next area containing ash trees. Each plot center was geo-referenced and recorded (Supplement I).

Within each plot we measured and identified each tree ≥5 cm diameter at breast height (DBH). Each ash tree was also rated on a dieback scale of 1-5, where 1 is a healthy tree, 5 is a dead tree, and 2-4 represent progressive levels of dieback (Smith et al., 2015). We tagged all the ash trees within plots with a unique code, which allowed us to track individual trees from year to year. We recorded percent canopy openness within each plot by taking the mean of four measurements in each cardinal direction at the center of the plot with a hand-held densiometer. In 2023, we revisited tagged ash trees, rated them again on the 1-5 dieback scale.

3.2.3. Statistical Analyses

All data were analyzed in R (v4.2.3; R Core Team, 2023). We first determined whether there were any spatial trends in EAB induced mortality and dieback due to potential differences in initial arrival at the sites. We plotted each site on a map with a color scale corresponding to different measurements of EAB impact [percent site mortality (i.e., percent of ash trees in dieback category 5) and mean canopy dieback rating] using the sf package in R (Pebesma & Bivand, 2023) (Supplement K). Visual inspection of these maps showed no clear spatial trends in EAB impacts, so we did not consider site location as a variable in our analyses.

All data were first tested for normality prior to analyses. We rectified non-normal data using log or square root transformations, and, if transformations did not work, we instead used non-parametric tests. We either used Pearson's correlation when data met assumptions of normality or Spearman's rank test otherwise.

The unit of replication was the site level (N = 15). We calculated two metrics that quantified EAB dieback and mortality at each site in 2022-2023 as follows: site ash dieback rating; percent ash mortality. The following site level data were tested for correlation with EAB dieback and mortality metrics: ash basal area (m^2/ha), basal area of all trees (m^2/ha), ash density (stems/ha), tree density of all species (stems/ha), ash dominance (percent ash basal area), ash importance value (measured as the sum of relative ash dominance, relative ash density, and relative ash frequency), and canopy cover (percent cover). In addition, we calculated the following tree diversity metrics: species richness, Shannon-Weiner diversity index, and evenness, which were also correlated with three measurements of EAB dieback and mortality. Finally, we assessed the changes in dieback rating and percent mortality at each site during 2022-2023 and conducted correlation analyses with those and all site characteristics.

We also grouped ash trees across all sites into eight DBH classes ranging from 5.0-10.0 cm DBH and >40.1 cm DBH in 5.0 cm increments to determine if ash dieback rating in 2022 and 2023 varied amongst DBH classes. We performed a Kruskal-Wallis test of dieback rating as a function of DBH class. To determine which DBH classes were significantly different from one another we then performed a Dunn's test for multiple comparisons using the dunnTest function from the FSA package (Ogle et al., 2023).

3.3. Results

3.3.1. Site Descriptions

Overall, we measured 282 individual ash trees $\geq$5 cm DBH. Most of the ash in our plots were green ash (number of trees = 260) while white ash was relatively rare (22). The three most common non-ash overstory species were sweetgum (135) followed by red maple (122), and boxelder (127). Privet (123) and American hornbeam (120) were the most common understory species across all sites (Fig. 2.1B). In total, we identified 37 different tree species across our sites, with species richness' ranging from 5 to 16 species and Shannon-Weiner's Diversity Index ranging from 1.25 to 2.29.

Total site density ranged from 446 stems/ha to 1,571 stems/ha. Mean total site density at our sites was 840 $\pm$ 79 stems/ha and mean ash density was 200 $\pm$ 30 stems/ha. Although ash was present at all our sites, it was a minor component at some sites, e.g., 4.5% of total basal area, and a much larger component at others, e.g., 70.63% of total basal area. Sites where ash was dominant primarily had much larger trees, for example, the site with the high ash dominance had mean ash DBH of 45.16 $\pm$ 1.90 cm. This is opposite for sites where it was not dominant, as the site with the lowest ash dominance had mean ash DBH of 16.05 $\pm$ 1.15 cm. Our sites had a wide range of EAB induced mortality and damage. Some sites did not have any dead ash trees (i.e., rated as a 5 on the 1-5 dieback scale) in 2022, however, one site had >50% mortality. In 2023, all but one site had at least one dead tree as dieback progressed across the landscape. Mean dieback ratings were also variable from 1.2 $\pm$ 0.11 to 4 $\pm$ 0.49 in 2022 and 1.53 $\pm$ 0.274 to 4.14 $\pm$ 0.404 in 2023.

3.3.2. Site Attributes in 2022

We found that EAB dieback and mortality were not correlated with either total tree density (p-values ranged from 0.412-0.982) or total basal area (p-values ranged from 0.216-0.865). Similar results were found for site attributes such as ash density, basal area, importance value, and dominance (p-values ranged from 0.219-0.984). EAB dieback and mortality were also not correlated with any diversity metrics (Shannon-Weiner diversity index, evenness, and richness). However, percent EAB mortality was negatively correlated with tree canopy cover (Spearman's Rho = -0.551, p = 0.050) (Fig. 3.2).

Kruskal-Wallis test revealed that there were significant differences between DBH classes and mean dieback ratings (χ^2 = 20.99, df = 7, p = 0.004). According to a post-hoc Dunn's test, trees in the 30.1-35.0 cm DBH class had greater mean dieback ratings than those in the 5.0-10.0 cm (Z = 2.89, p = 0.026) and 10.1-15.0 cm (Z = -2.89, p = 0.035) DBH classes. In addition, trees in the 35.1-40.0 cm DBH class had greater mean dieback ratings than trees in the 5.0-10.0 cm

DBH class ($Z = 3.18$, $p = 0.02$), and in the 10.1-15.0 cm DBH class ($Z = -3.19$, $p = 0.04$). However, there were no differences between the other age classes (p-values ranged from 0.102 to 0.882) (Fig. 3.3A).

3.3.3 Site Attributes in 2023

We found that no correlations with ash dieback and mortality and measurements describing site structure in 2023 (p-values ranged from 0.082 to 0.815). Additionally, neither dieback nor mortality were correlated measurements of site composition in 2023 (p-vales ranged from 0.082 to 0.889).

Similar to 2022, when we grouped all of the ash trees across all our sites into DBH classes a Kruskal-Wallis test revealed differences between DBH classes and dieback ratings ($\chi^2 = 34.66$, df = 7, $p < 0.001$). A post-hoc Dunn's test revealed that trees in the three highest DBH classes (30.1-35 cm, 35.1-40 cm, >40.1 cm) all had greater mean dieback ratings than trees in the smallest DBH class (5-10 cm). Also, two of those DBH classes (30.1 cm-35 cm, and >40.1 cm) had greater mean dieback ratings than the 10.1-15 cm and 15.1-20 cm DBH classes (Fig. 3.3B).

3.3.4 Temporal changes in ash dieback and mortality

Correlation analyses with the changes in dieback and mortality and site attributes at each site from 2022 to 2023 revealed multiple relationships. The change in percent ash mortality over one year was positively correlated with ash basal area ($r = 0.840$, $p < 0.001$), dominance ($r = 0.841$, $p < 0.001$), and importance value ($r = 0.730$, $p = 0.002$) (Fig. 3.4A-C). The change in ash dieback rating was also positively correlated with ash basal area ($r = 0.846$, $p < 0.001$), dominance ($r = 0.819$, $p < 0.001$), and importance value ($r = 0.763$, $p < 0.001$) (Fig 3.5A-C).

In addition, there were negative correlations between site diversity and change in mortality and dieback from 2022 to 2023. The change in percent mortality (Spearman's Rho = -0.572, p = 0.047) was correlated with site evenness (Fig. 3.6), but not with Shannon-Weiner Diversity index (p = 0.08). The change in mean dieback rating was negatively correlated with Shannon-Weiner Diversity Index (r = -0.556, p = 0.031) and evenness (Spearman's Rho = -0.64, p = 0.010) (Fig 3.7A-B). Neither change in percent mortality nor dieback rating was also not correlated with species richness (p = 0.054-0.063).

3.4. Discussion

Our study revealed several important patterns related to EAB-induced mortality and dieback in our sites in Piedmont of Georgia as follows: 1) ash tree mortality and dieback was not correlated with any site attributes such as ash or total basal area, density, or diversity when analyzed in any given year (2022 or 2023); 2) ash trees in two of the largest DBH classes had greater dieback than trees in the smallest DBH classes in 2022 and 2023; 3) percent mortality was negatively correlated with canopy cover in 2022 but not in 2023; 4) the change in dieback and mortality from 2022 to 2023 was though positively correlated with ash abundance (basal area, dominance, and importance value); and 5) the annual change in dieback and mortality were negatively correlated with tree species diversity and evenness.

Consistent with Smith et al. (2015), we did not find any relationships between EAB-induced ash tree mortality and site attributes in any given year. The higher dieback rating of some of the larger DBH categories in both years might be indicative of EAB preference. Trees in these classes might be more apparent, and thus, more likely to be randomly selected by EAB when they enter a host patch (Saint-Germain et al., 2007). Although we did not measure EAB oviposition, the increased tree dieback and mortality may be due to increased oviposition and greater larval survivorship due to feeding in the thicker phloem in larger trees. Although, in a common garden study, Rigsby et al. (2014) found that DBH was not associated with oviposition preferences of EAB. Jennings et al. (2014) also reported that observed oviposition rates and mating adults were not affect by tree diameter in forests in Michigan, but rather by the degree of canopy dieback, such that trees with intermediate levels of dieback (3-4 out of 5) had greater numbers of eggs and adults.

These results also have implications for EAB management through classical biological control. Currently, four Hymenopteran parasitoids are approved for release in the United States, *Spathius agrili* Yang, *Spathius galinae* Belokobylskij, *Tetrastichus planipennisi* Yang, *Oobius agrili* Zhang and Huang (USDA-APHIS, 2021). Of these, only the larval parasitoid *S. agrili* and egg parasitoid *O. agrili* will be approved for release in the Southeast. Although *S. agrili* has never established in northern states, studies of the closely related *S. galinae* show that it is much less likely to parasitize EAB larvae in larger trees due to increasing bark thickness (Murphy et al., 2017). Biocontrol in the Southeast may then be optimized by releasing *S. agrili* at sites with primarily smaller trees where bark thickness will not preclude oviposition. Similarly, sites with predominantly larger trees may prioritize release of *O. agrili*. EAB eggs are lain on the bark surface and *O. agrili* parasitism is therefore less constrained by tree size.

In 2022, we found a negative relationship between percent ash mortality and canopy cover across our sites. However, this relationship was not significant for either 2023 or for the difference in percent mortality over two years. EAB is known to favor open trees in full light conditions (McCullough et al., 2009). Thus, our results are in contrast where lower tree dieback was in more open sites. It’s possible that these ash trees in more open sites were growing more vigorously due to less competition from surrounding vegetation and greater access to nutrients and may have been better able to defend themselves in the early stages of EAB invasion. Considering that this pattern was absent in 2023 suggests that eventually these trees in open areas eventually succumbed to EAB with an increase of their populations within and across sites.

We found positive relationships between the annual change in ash mortality and dieback with ash basal area, dominance, and relative importance value. We also found negative relationships between the change in mortality and dieback and Shannon-Weiner diversity index and evenness across all sites. These results are not necessarily surprising, since sites with more ash trees tended to have lower tree diversity. However, it is interesting that we did not find relationships between changes in percent ash mortality and dieback rating with ash density. Hence, sites with more phloem resources (i.e., more ash basal area and where this species is more dominant) are more likely to be experience mortality in Georgia. The resource concentration hypothesis states that greater insects, and hence greater herbivory is expected in areas of high host density (Root, 1973). Despite not finding any effects of density, our results may still be consistent with this hypothesis. The original paper was conducted on collards (*Brassica oleracea* L.) in meadow ecosystems. In forests, the more complex age structure and variation in size means that metrics that account for the size and number of trees are likely better descriptors of resource availability under the resource concentration hypothesis. This would be especially true for woodboring beetles such as EAB, that rely on the total volume of phloem resources across a patch rather than simply host density. These results are also consistent with our observation that across all sites and in both years, trees in the largest DBH classes had higher mean dieback rating (Fig. 3.3). It is possible that EAB is selecting patches with greater resource availability across the landscape and the largest trees within patches, a trend that may be considered for prioritizing chemical control where it's more feasible.

Knight et al. (2013) found evidence for resource dilution patterns of EAB-induced dieback and mortality in Ohio, where sites with lower ash density reached complete mortality quicker than sites with higher ash density. Landscape differences between Ohio and Georgia might mediate the observed changes. For example, in the Georgia Piedmont, ash is highly heterogenous, and so EAB might stay in concentrated host patches longer. In states where ash is more abundant and continuously distributed across the landscape, EAB adults can perhaps disperse more readily between ash patches. Knight et al. (2013) also assessed these patterns over five years while our study was over one year only. Hence, it will be beneficial to monitor these sites to determine if these relationships persist over longer timeframes.

Overall, our results suggest that as EAB continues to spread in the Southeast, mortality and dieback rates might not be consistent across the landscape. Most importantly, the rate of EAB induced ash mortality and dieback was associated with site characteristics that described the relative amount of available ash phloem resources. These data suggest that managing agencies or private landowners in the Southeast might want to prioritize areas with a higher ash basal area for management interventions (i.e., biocontrol and/or chemical control) and to increase tree species diversity to reduce EAB impacts. Since some southeastern states are relatively recently infested by EAB and it has not been detected in others yet, determining areas where EAB is more likely to cause a greater rate of mortality may be crucial to preserving the native ash resource in the region.

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Figure Legend

Figure 3.1 Ten tree species with highest proportions of total basal area across 15 sites in Northeast Georgia. Error bars represent 95% confidence intervals (A). Tree species density (trees/ha) for ten most common tree species encountered across all 15 sites in Northeast Georgia (B).

Figure 3.2 Correlation between mean site canopy openness (%) and percent emerald ash borer mortality in 15 sites in Northeast Georgia.

Figure 3.3 Mean ash dieback ratings in 2022 across diameter at breast height (DBH) classes in 15 sites in Northeast Georgia (A). Mean ash dieback ratings in 2023 across DBH classes in 15 sites in Northeast Georgia (B). We calculated dieback ratings for each ash tree on a 1-5 scale where 1 = a healthy tree and 5 = a dead tree. Letters indicate significance according to post-hoc Dunn's Test with Bonferroni adjusted P-values.

Figure 3.4 Correlation between change in percent ash mortality and: ash basal area (A) ash dominance (% of site basal area made up by ash) (B), and ash importance value (calculated as the sum of ash density, ash dominance, and relative frequency) (C) across all sites in Northeast Georgia.

Figure 3.5 Correlation between the change in mean dieback rating and: ash basal area per hectare across all sites (A) dominance (% of site basal area made up by ash) across all sites (B), and ash importance value (calculated as the sum of ash density, ash dominance, and relative frequency (C) across all sites in Northeast Georgia.

Figure 3.6 Correlation between the change in percent ash tree mortality and evenness across all sites with emerald ash borer infestations in Northeast Georgia.

Figure 3.7 Correlation between the change in ash tree dieback rating and: Shannon Diversity Index across all sites (A) and evenness (B) across all sites in Northeast Georgia.

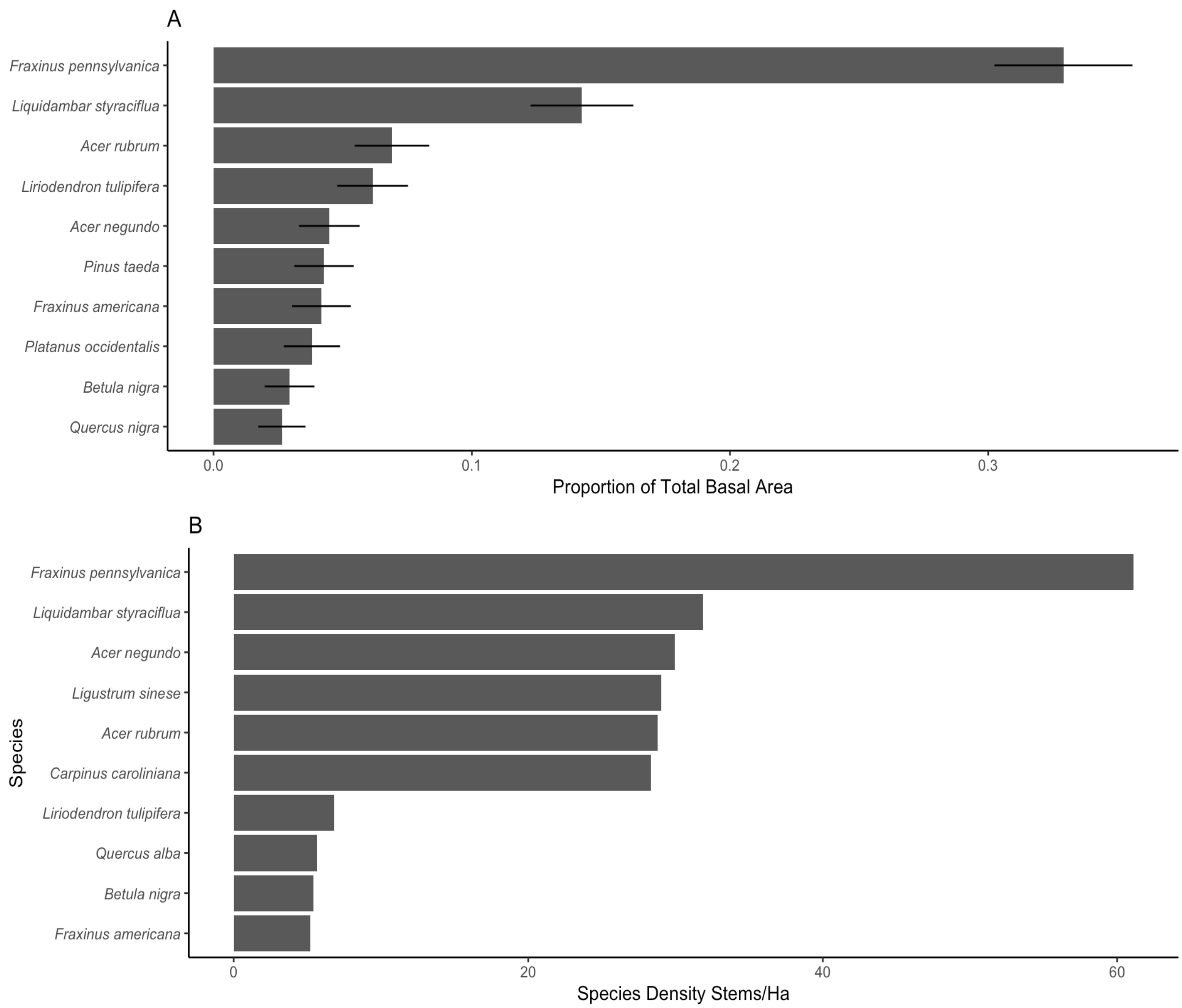
A
Fraxinus pennsylvanica
Liquidambar styraciflua
Acer rubrum
Liriodendron tulipifera
Acer negundo
Pinus taeda
Fraxinus americana
Platanus occidentalis
Betula nigra
Quercus nigra
0.0
0.1
0.2
0.3
Proportion of Total Basal Area
B
Species
Fraxinus pennsylvanica
Liquidambar styraciflua
Acer negundo
Ligustrum sinese
Acer rubrum
Carpinus caroliniana
Liriodendron tulipifera
Quercus alba
Betula nigra
Fraxinus americana
0
20
40
60
Species Density Stems/Ha

Figure 3.1

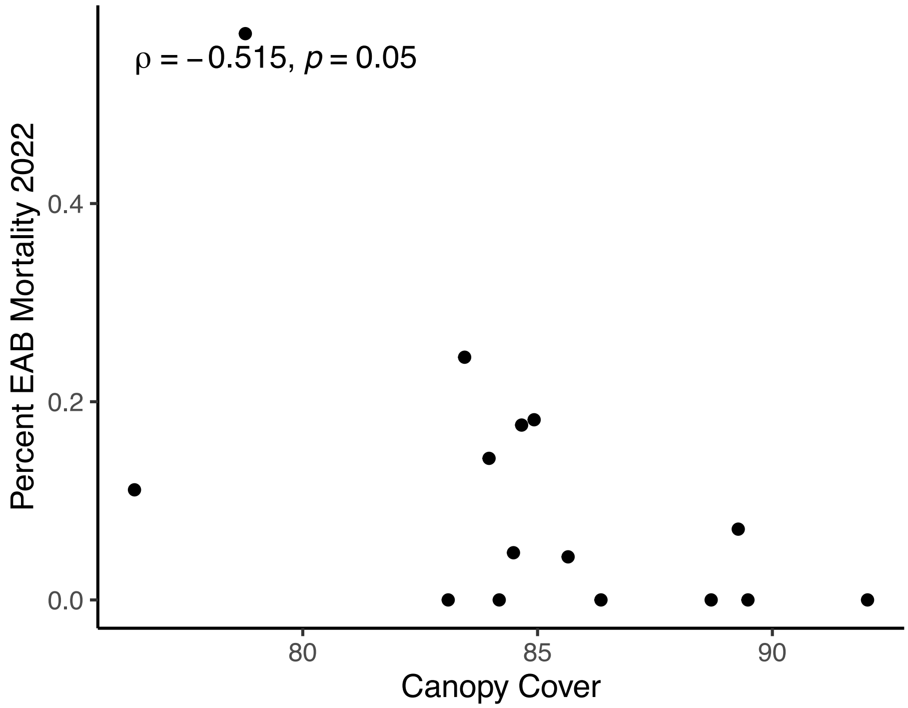
ρ = −0.515, p = 0.05
Percent EAB Mortality 2022
0.4
0.2
0.0
80
85
90
Canopy Cover

Figure 3.2

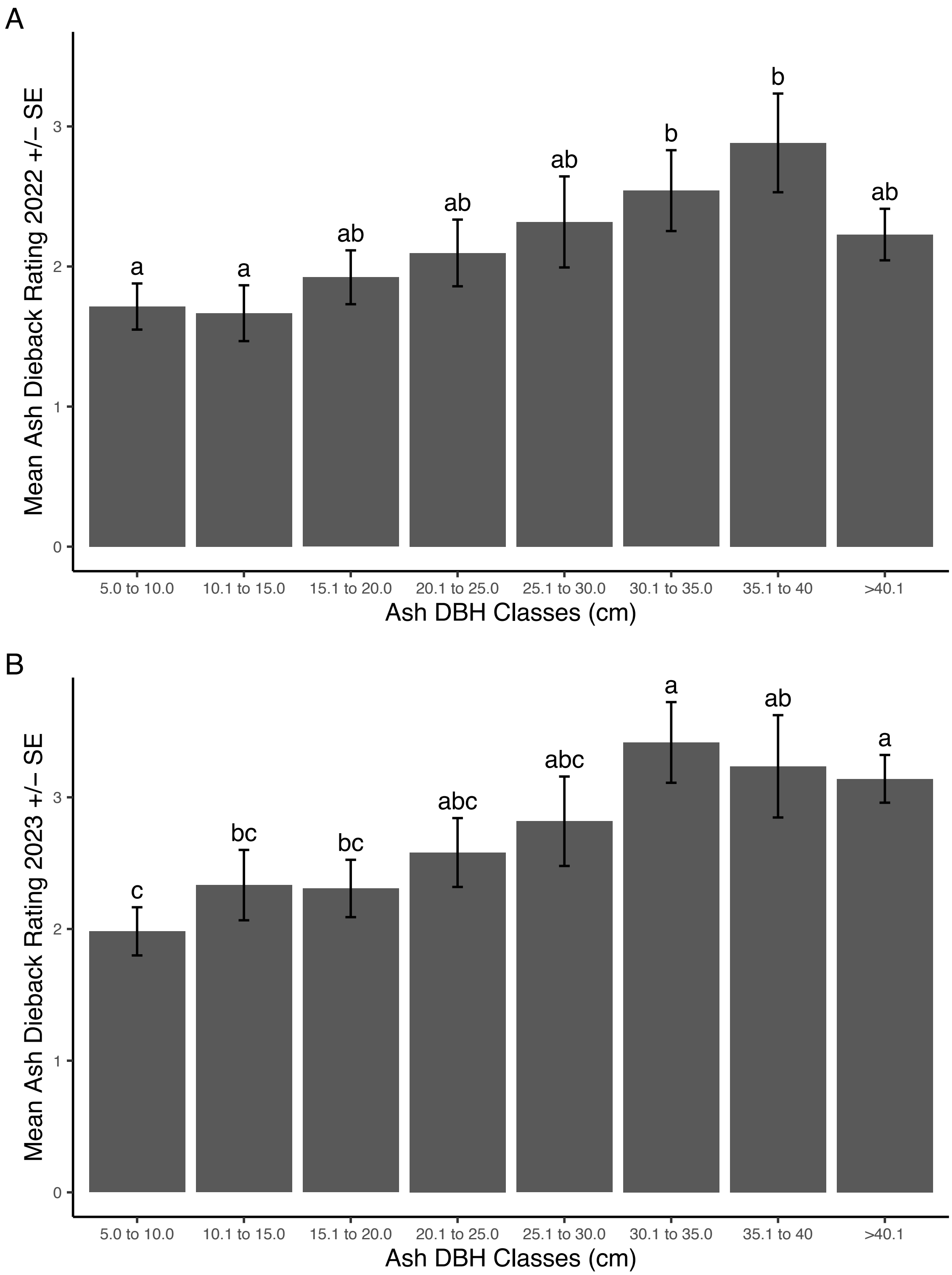
A
Mean Ash Dieback Rating 2022 +/− SE
Ash DBH Classes (cm)
5.0 to 10.0
10.1 to 15.0
15.1 to 20.0
20.1 to 25.0
25.1 to 30.0
30.1 to 35.0
35.1 to 40
>40.1
a
a
ab
ab
ab
b
b
ab
B
Mean Ash Dieback Rating 2023 +/− SE
c
bc
bc
abc
abc
a
ab
a
0
1
2
3

Figure 3.3

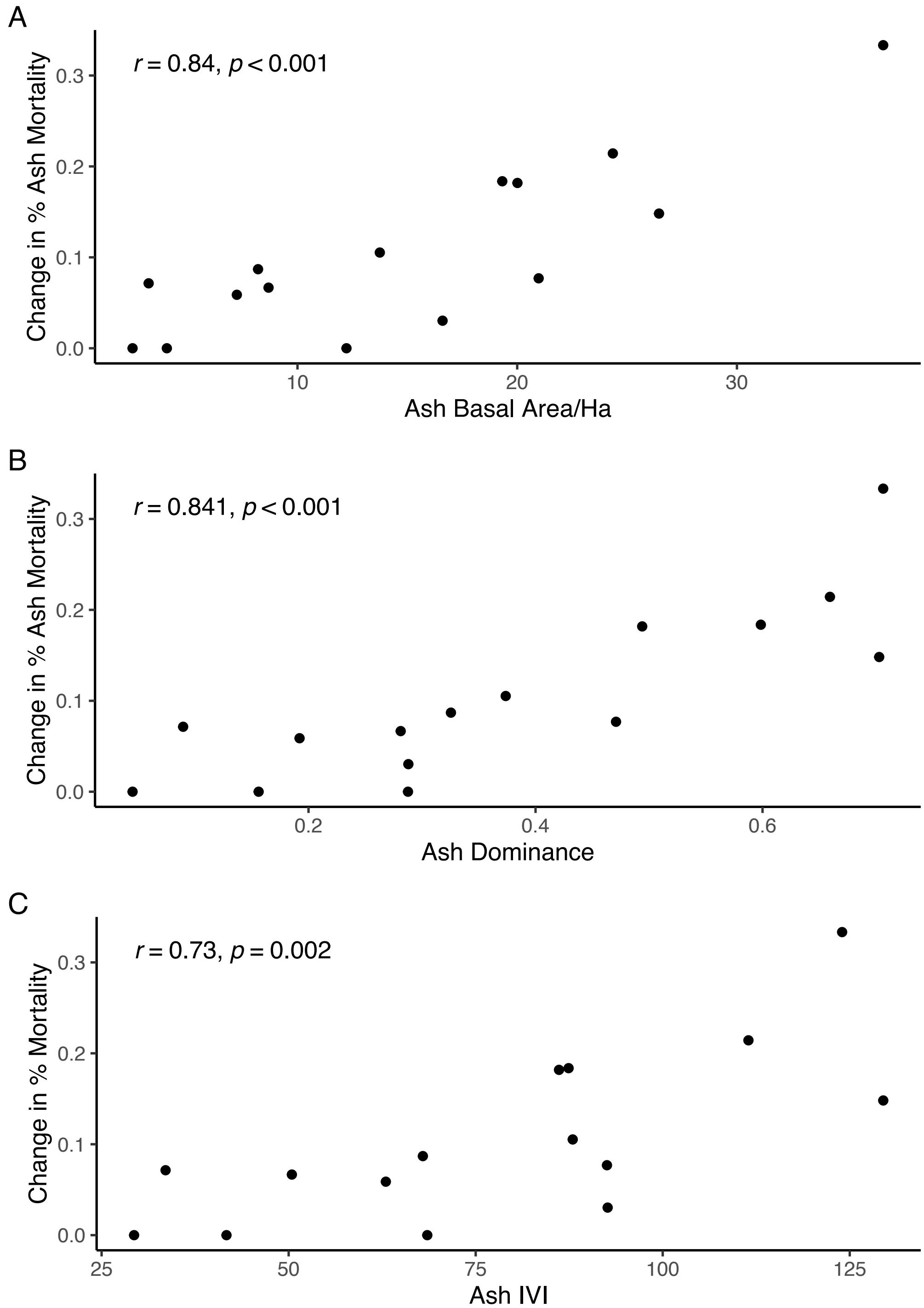
A
$r = 0.84$, $p < 0.001$
Change in % Ash Mortality
0.0
0.1
0.2
0.3
10
20
30
Ash Basal Area/Ha
B
$r = 0.841$, $p < 0.001$
Change in % Ash Mortality
0.0
0.1
0.2
0.3
0.2
0.4
0.6
Ash Dominance
C
$r = 0.73$, $p = 0.002$
Change in % Mortality
0.0
0.1
0.2
0.3
25
50
75
100
125
Ash IVI

Figure 3.4

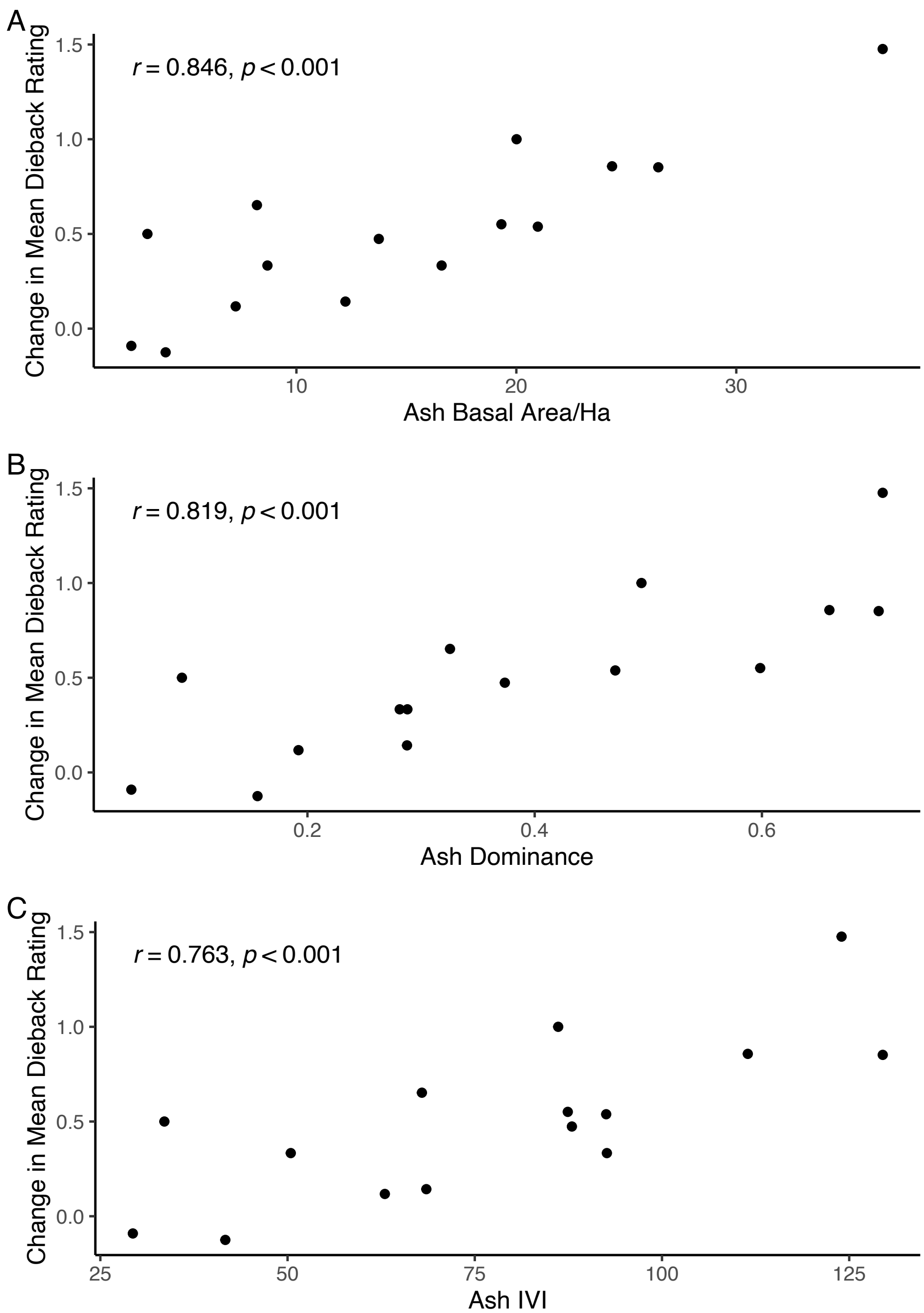
A
Change in Mean Dieback Rating
r = 0.846, p < 0.001
Ash Basal Area/Ha
B
Change in Mean Dieback Rating
r = 0.819, p < 0.001
Ash Dominance
C
Change in Mean Dieback Rating
r = 0.763, p < 0.001
Ash IVI

Figure 3.5

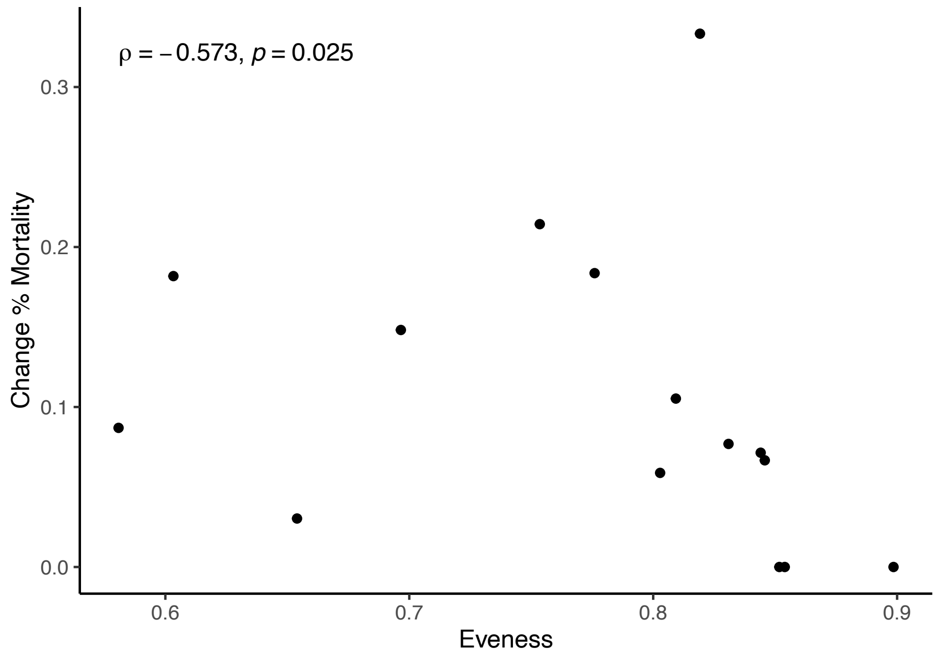

Figure 3.6

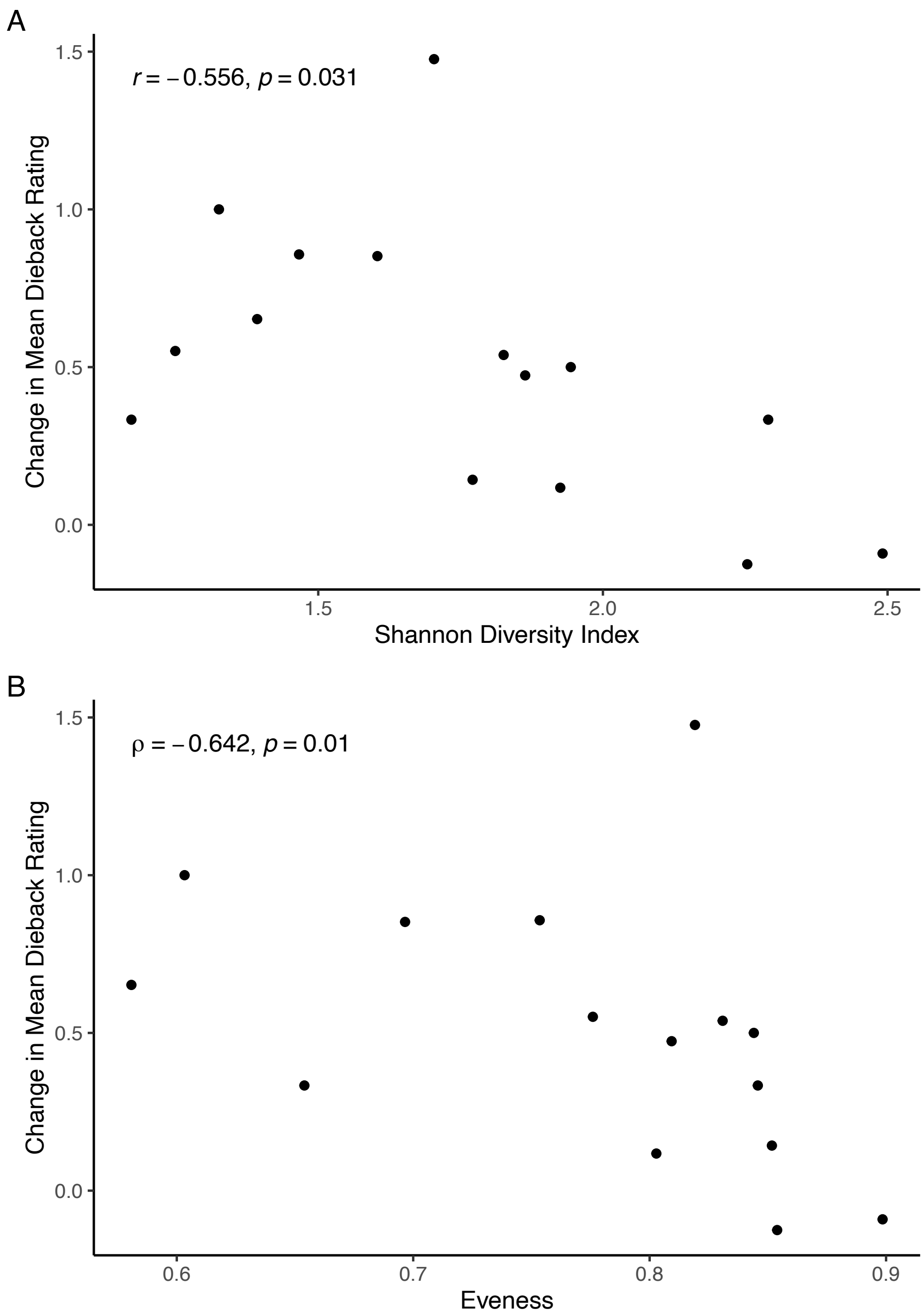
A
$r = -0.556$, $p = 0.031$
Change in Mean Dieback Rating
1.5
1.0
0.5
0.0
1.5
2.0
2.5
Shannon Diversity Index
B
$\rho = -0.642$, $p = 0.01$
Change in Mean Dieback Rating
1.5
1.0
0.5
0.0
0.6
0.7
0.8
0.9
Eveness

Figure 3.7

CHAPTER 4: CONCLUSIONS AND FUTURE DIRECTIONS

4.1. Park Users' Support for Emerald Ash Borer Management

This thesis had two specific goals related to the emerald ash borer (EAB) invasion on ash trees (*Fraxinus* spp.) in Georgia. My first goal was to determine park users level of support for two major EAB control methods, namely, biological and chemical control. I conducted an in-person intercept survey in four parks across Northeast Georgia with a known ash component and EAB induced dieback. My survey was designed to measure park users' awareness and knowledge of EAB and invasive species, their attitudes towards ash trees and insects, their risk perceptions of EAB and control methods, demographic information, and park use information such as recreation preferences. I conducted ordinal logistic regression on our final data set of 174 surveys and then performed stepwise backwards regression using AIC values to find a best fit model for both support for biocontrol and support for chemical control. Park users tended to be more supportive of biocontrol than chemical control, and they also perceived more risk from the latter treatment. Similarly, both support models included terms for risk perceptions. However, I also found that in both cases, recreation preferences influenced levels of support. Specifically, individuals who indicated that they use the parks for birdwatching were less likely to support chemical control, and individuals who use the parks to sit and enjoy nature were more likely to support biocontrol. Although not measured in this survey, these recreation preferences might relate to the values and value orientations of individuals in this group, a concept that should be explored further. Also, individuals who expressed more positive attitudes towards ash trees were more likely to support chemical control and level of concern about EAB increased support for biocontrol. These results may be important for managers seeking to use either biocontrol or chemical control to slow EAB induced mortality in public parks. Specifically, individuals who are in positions to make decisions about management may want to consider how recreationists use their parks to inform their messaging around a chosen control method. They may then elect to use messaging that reduces perceived risk of either biocontrol or chemical control.

4.2 Site Attributes Influence on the Rate of EAB Induced Dieback and Mortality

I also sought to determine if emerald ash borer induced mortality and dieback were associated with forest characteristics that relate to structure and diversity across Northeast Georgia. I collected data at 15 total sites across the Piedmont region of Georgia. At each site I calculated relevant site information including measurements of ash abundance (density, basal area, and dominance), site structure (total density and total basal area), and EAB induced mortality and dieback (percent mortality and canopy dieback rating). I then performed correlation analyses to determine if EAB induced mortality and dieback were correlated with any of site characteristics. In 2022 mean site canopy dieback rating was negatively correlated with percent canopy openness, although no other correlations were found in either 2022 or 2023. However, there were positive correlations between the change in site dieback rating and percent mortality from 2022 to 2023 with variables that were related to the amount of ash at each site, (e.g., ash basal area, dominance, and importance value). Similarly, there were negative correlations between change in site canopy dieback rating and Shannon-Weiner Diversity Index. The change in canopy dieback rating and percent mortality across all sites was also negatively correlated with evenness.

These results demonstrate that in any individual year, EAB induced mortality and dieback may not appear correlated with forest characteristics. However, the change in dieback and mortality, which is a rate, was positively correlated with metrics that describe the amount of ash across our sites. Although, this relationship was not captured between either the change in percent mortality or change in dieback rating and ash density. Hence, the rate at which ash dieback and mortality occur is positively correlated with the amount of available ash phloem at a site (due to more larger trees as indicated by basal area) and the proportion of ash at a site (as measured by dominance and importance value). The negative relationships with a few of the diversity metrics is not necessarily surprising, since sites with higher diversity would tend to have less ash basal area and a smaller proportion of ash. However, I found smaller correlation coefficients for these tests, which might demonstrate the effects of greater ash resources across the landscape are more influential in how dieback and mortality proceeds.

4.3 Future Directions for Research

An important result from my survey project was that recreation preferences play important roles in determining levels of support for EAB control. As noted above, this survey did not quantify values. However, it is possible that values and value orientations amongst these recreation groups influence their support for management. Future work on the human dimensions of EAB management could develop survey instruments that measure environmentally relevant values (Stern et al., 1998) to understand how values influence support for management. Our survey also took place across a subset of parks in Northeast Georgia. Although we felt that we obtained a diverse sample, examining support for management in different areas of the country would be beneficial. For example, similar lines of inquiry in areas of the country where ash is more prominent or where EAB has been around for longer time and is more well known by the public may yield different results. Awareness and knowledge were not significant in our models, however if EAB damage is more visible to park users' due to longer pest pressure or greater proportions of dead ash across the forest in other areas of the country, awareness might mediate concern about EAB in ways that would influence support for management. Future human dimensions work on the management of EAB could also examine specific knowledge of other such invasive species across different areas of the United States to see if there are spatial trends to support for biocontrol due to regional differences.

The pre-testing for this survey suggested that we remove social trust questions, which asked participants how much they trust managing agencies to implement EAB control measures. Social trust could still be a useful concept to examine regarding EAB management however, since it has shown utility in human dimension studies of other conservation issues (Harper et al., 2015; Siegrist & Cvetkovich, 2000; Vaske et al., 2018). In this case, park users may not have been an appropriate stakeholder group to ask such questions. Future work that implements social trust as a variable to see how it influences levels of support would continue to develop our understanding of the human dimensions of EAB management. Finally, recent work from Kentucky examined landowners with ash on their properties intentions to manage for EAB (Adhikari et al., 2023). They found that severity of risk perceptions influenced landowners intentions, but also found that knowledge of management options, economic objectives, and group efficacy also were significant. Continued studies on with different stakeholder groups such as homeowners or individuals with street trees near their properties would paint a more complete picture of the support for EAB management.

Continued monitoring of the rate of EAB induced mortality and dieback across Georgia would be beneficial. This project only captured a year-to-year change in mortality and dieback, and longer-term monitoring might provide more information about how quickly various sites succumb to EAB. Green ash (*Fraxinus pennsylvanica* Marshall) was much more prevalent across these sites than white ash, with only two sites having a major white ash (*Fraxinus americana* L.) component. Even though white ash is still susceptible to EAB, it is known to be a less preferred host than green ash (Anulewicz et al., 2007; 2008). Future work could quantify the rate of mortality and decline amongst different ash species in the Southeast to better understand EAB's movement across southeastern forests.

More broadly, there are important questions regarding EAB's movement through the Southeast that may be examined. For example, there has been a great deal of work on the northern limits to EAB's range based on larval cold tolerance (Crosthwaite et al., 2011; DeSantis et al., 2013), but less work has examined the southern limits to EAB's spread due to warm winter temperatures. In a laboratory study, Duan et al. (2021) demonstrated that diapausing EAB J-larvae require a chilling period of at least 2 months at 12.8° C prior to emerging as adults. Another laboratory study showed increased rates of prepupal mortality when diapausing prepupae were taken from the field in later winter and incubated at 25° C and 28° C relative to a 13° C control group (Richins, 2022). Although these prepupae may have completed their chilling requirements, pupation was not observed in this study. It is possible that this diapause is obligate for EAB, as is the case for its native congener the bronze birch borer (Muilenburg & Herms, 2012). If this is the case, then there are locations at the southern range of ash in North America where winter temperatures may remain warm enough such that EAB will not be able to complete development. Further investigation of such field studies would help determine EAB's potential for spread further south and continue to cause EAB mortality.

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SUPPLEMENTAL MATERIAL

Supplement A

Evaluating the establishment of emerald ash borer biological control parasitoids in Georgia

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SA.1 Objective

Emerald ash borer (*Agrilus planipennis* Fairmaire; EAB) (Coleoptera: Buprestidae) is currently leading to the widespread death of ash trees (*Fraxinus* spp.) across North America (Chapter 1). It has been present in Georgia since 2013 and the only long-term management option to protect native ash trees is biological control in forested ecosystems (Chapter 1). The objective for this project is to evaluate the potential for biocontrol of EAB through releases of two species of Hymenopteran parasitoids, *Spathius agrili* Yang and *Oobius agrili* Zhang and Huang, in collaboration with USDA-APHIS over two years, and to recover them subsequently.

SA.2 Methods

SA.2.2 Site Selection

Site selection for biocontrol release sites began in the spring of 2021 and continued throughout the fall. We evaluated sites across seven counties in Northeast Georgia that were known to have ash present in them. At each site, we completed a visual assessment to determine if it would be suitable for biocontrol. We assessed the relative amount of ash trees present at each site, the presence of EAB signs and symptoms, the amount of ash dieback due to EAB, and if the site was isolated or connected to other woodlots. We deemed sites to be suitable if >25% of the trees were ash in a variety of size classes, there were low-moderate EAB populations based on signs and symptoms, there was connectivity to other woodlots, and they were greater than 16.2 hectares. In many cases we were unable to find wooded areas with ash that were greater than 16.2 hectares. In those cases, we prioritized sites that were connected to other woodlots that also contained ash. These targets are in accordance with the USDA-APHIS Emerald Ash Borer Biological Control Release and Recovery Guidelines (2021). Our final list of potential sites consisted of 19 sites, five of which were approved for releases by USDA-APHIS. The five release sites were in the following five counties in the Piedmont region of Georgia: Clarke, Greene, Gwinnett, Jackson, and Oglethorpe. They were generally situated on low-lying bottomlands or on riparian corridors with green ash (*Fraxinus pennsylvanica* Marshall) as the major ash species, however the site in Jackson represents an exception, it is a mid-slope mixed hardwood forest with white ash as the major ash species, although green ash is present too.

SA.2.2 Site Characteristics

At each site, we established three 10-m radius plots along a transect that would serve as our release plots. We placed our first plot intentionally in an area where ash was present. Our second and third plots were placed at least 50 m away around the next area containing ash trees that was found. This was done since ash is relatively uncommon in Georgia, and even in suitable habitats ash is distributed highly heterogeneously. Randomly distributing plots within a site therefore could have resulted in plots with few ash trees to conduct parasitoid releases on, which would not have been ideal for biocontrol releases.

Within each plot we measured and identified each tree over 5 cm DBH to characterize each site in terms of structure and composition. Each ash tree was also rated on a dieback scale of 1-5, where 1 is a healthy tree and 5 is a dead tree and 2-4 represent progressive levels of dieback (Smith et al., 2015). We also calculated percent ash mortality and average ash dieback rating at each site so that we could track these metrics from year to year and we measured canopy openness within each plot using a handheld densiometer. The unit of replication for analysis was the site level so data were averaged to the whole site. All ash trees within each plot were tagged with a specific code, which allowed us to track individual trees from year to year and note changes in EAB mortality and dieback (Chapter 3).

SA.2.3 Biocontrol Releases

In 2022, we released approximately 200 female *O. agrili* per week at each site for three weeks from May 3rd to May 18th and again from June 2nd to June 24th. In total, approximately 1,400 *O. agrili* were released per site over the course of the summer. All *O. agrili* were released as developing pupae within parasitized EAB eggs held on coffee filter paper in small medicine vials. Each vial was left out in the field for six weeks to allow all the parasitoids to emerge, after which they were removed. In 2023, we began releases of *O. agrili* at all sites on April 26th and continued for three weeks until May 10th. We released approximately 200 female *O. agrili* per site on June 7th, 400 females on June 15th, and 200 females on June 22nd. In 2023 all sites again received approximately 1,400 female *O. agrili*.

Releases of the larval parasitoid, *Spathius agrili*, began in late June in 2022. We released approximately 200 female *S. agrili* per week at each site from June 23rd to August 4th. Some sites (Gwinnett, Jackson, and Oglethorpe) also received an additional week of releases the week of June 8th. All *S. agrili* were released as adults directly into each plot. In 2023 releases of *S. agrili* again began in June. We released approximately 200 female *S. agrili* per week at each site from June 15th to July 27th. In each year, all sites received approximately 1,400 female *S. agrili*.

SA.3 Future work

We will be doing recovery of these two parasitoids in 2024 and 2025. Various methods will be used such as yellow pan trapping according to the guidelines or deploying sentinel bolts as described in (Quinn et al., 2022). Since we have data for ash tree dieback and mortality at the very start of releases, and also data from control (non-released) sites, we will be able to compare the potential impact of these parasitoids in suppressing EAB populations. This is the first time that biocontrol has been attempted for EAB in Georgia, and we hope that the parasitoids establish and slow the spread of this invasive pest in novel ecosystems.

SA.4 References

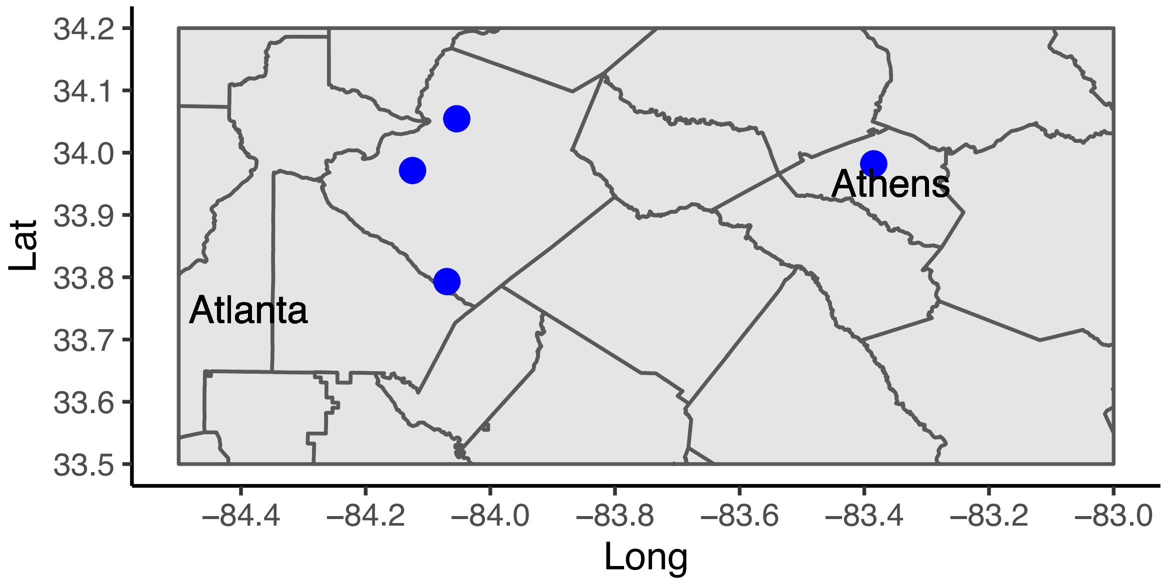
Smith, A., Herms, D. A., Long, R. P., & Gandhi, K. J. K. (2015). Community composition and structure had no effect on forest susceptibility to invasion by the emerald ash borer (Coleoptera: Buprestidae). *The Canadian Entomologist*, *147*(3), 318–328. https://doi.org/10.4039/tce.2015.8

Quinn, N. F., Gould, J. S., Rutledge, C. E., Fassler, A., Elkinton, J. S., & Duan, J. J. (2022). Spread and phenology of *Spathius galinae* and *Tetrastichus planipennisi*, recently introduced for biocontrol of emerald ash borer (Coleoptera: Buprestidae) in the northeastern United States. *Biological Control*, *165*, 104794. https://doi.org/10.1016/j.biocontrol.2021.104794

USDA-APHIS. (2021). Emerald Ash Borer Biocontrol Release and Recovery Guidelines. https://www.aphis.usda.gov/plant_health/plant_pest_info/emerald_ash_b/downloads/eab-field-release-guidelines.pdf

Supplement B

Map showing the locations of parks where intercept surveys were conducted in Clarke County, and Gwinnett County, Georgia.



Supplement C

Sociodemographic characteristics of respondents (n = 174) to an intercept survey measuring support for emerald ash borer control in Northeast Georgia. Note that respondents were allowed to select up to three recreation preferences.

	Number	%
Gender		
Male	81	46.6
Female	89	51.2
Prefer not to say	4	2.3
Age		
18-24 years	21	12.1
25-34 years	33	19.0
35-44 years	24	13.8
45-54 years	29	16.7
55-64 years	30	17.2
$\geq$65 years	37	21.3
How long it takes to drive to the park		
1-10 minutes	93	53.5
11-20 minutes	53	30.5
Over 20 minutes	21	21.1
Walk to park	7	4.0
How often park was visited in the past month		
0	18	10.3
1-3	62	35.6
4-6	30	17.2
7-9	14	8.1
10+	50	28.7
Recreation Preferences		
Walking/Hiking	158	46.3
Running/Jogging	53	15.5
Cycling	21	6.16
Birdwatching	36	10.6
Picnics/BBQ	7	2.1
Sitting and enjoying nature	55	16.1
Other	11	3.2

Supplement D

Respondents' were asked about their prior knowledge of the reasons that not native species can be considered invasive (n = 174).

Responses were used to generate invasive species knowledge scores.

Invasive species knowledge questions	**Median**	**Yes**		**No**		**I'm not sure**	
		Number	**%**	**Number**	**%**	**Number**	**%**
Before today were you aware that non-native species are considered invasive if they cause...							
Harm to the environment	Yes	138	79.3	18	10.3	18	10.3
Economic damage	Yes	126	72.4	25	14.4	23	13.2
Harm to people and pets	Yes	122	70.1	28	16.1	24	13.8

Supplement E

Respondent's attitudes towards park trees evaluated by asking them to rate the importance of various tree characteristics (n = 174).

	Median	Not at all important		Slightly important		Moderately important		Important		Very important		Factor Loading	Cronbach's alpha
		Number	%	Number	%	Number	%	Number	%	Number	%		
How important to you are the following characteristics of trees in the park?													0.85
That the park has trees that provide shade	Very important	4	2.3	10	5.8	20	11.5	36	20.7	104	59.8	0.57	
The park contains a variety of trees with different leaf shapes and colors	Important	5	2.9	13	7.5	29	16.7	47	27.0	80	46.0	0.85	
The park contains trees that flower	Important	7	4.0	15	8.6	32	18.4	47	27.0	73	42.0	0.81	
The park contains trees that are different shapes and heights	Important	5	2.9	13	7.5	35	20.1	46	26.4	75	43.1	0.86	

Supplement F

Respondents' attitudes towards insects. Note that a single question regarding attitudes towards butterflies was left out since it could not separate on to a single factor with the rest of the insect questions (n = 174).

	Median	Strongly dislike		Dislike		Neutral		Like		Strongly like		Factor Loading	Cronbach's alpha
		Number	**%**	**Number**	**%**	**Number**	**%**	**Number**	**%**	**Number**	**%**		
On a scale of strongly dislike to strongly like, how much do you like the following insects in this park													0.83
Bees	Like	8	4.6	5	2.9	35	20.1	41	23.6	85	48.9	0.43	
Ants	Neutral	19	10.9	22	12.6	73	42.0	34	19.5	26	14.9	0.79	
Flies	Dislike	43	24.7	54	31.0	58	33.3	10	5.8	9	5.2	0.78	
Wasps	Dislike	56	32.2	41	23.6	41	23.6	22	12.6	14	8.1	0.75	
Mosquitoes	Strongly dislike	106	60.9	45	25.9	19	10.9	1	0.6	3	1.7	0.66	
Beetles	Neutral	12	6.9	17	9.8	85	48.9	44	25.3	16	9.2	0.66	

Supplement G. Respondents' risk susceptibility to both biocontrol through the use of parasitoid wasps and chemical control through the use of systemic insecticides to control emerald ash borer in parks in Northeast Georgia (n = 174).

	Median	No Risk		Low Risk		Moderate Risk		High Risk		Factor Loading	Cronbach's alpha
		Number	%	Number	%	Number	%	Number	%		
Susceptibility to risk from biocontrol											
What risk do you think there is…											0.86
The wasps will harm the environment	Low Risk	34	19.5	79	45.4	51	29.3	10	5.8	0.65	
The wasps will harm humans (stinging, allergies, etc.)	Low Risk	40	23.0	74	42.5	43	24.7	17	9.8	0.87	
The wasps will harm pets (stinging, allergies, etc.)	Low Risk	39	22.4	81	46.6	42	24.1	12	6.9	0.95	
Susceptibility to risk from chemical control											
What risk do you think there is…											0.92
Applying chemicals in or around trees will harm the environment	Moderate Risk	11	6.3	41	23.6	69	39.7	53	30.5	0.81	
Applying chemicals in or around trees will harm humans	Moderate Risk	17	9.8	62	35.6	45	25.9	50	28.7	0.96	
Applying chemicals in or around trees will harm pets	Moderate Risk	13	7.5	44	25.3	59	33.9	58	33.3	0.90	

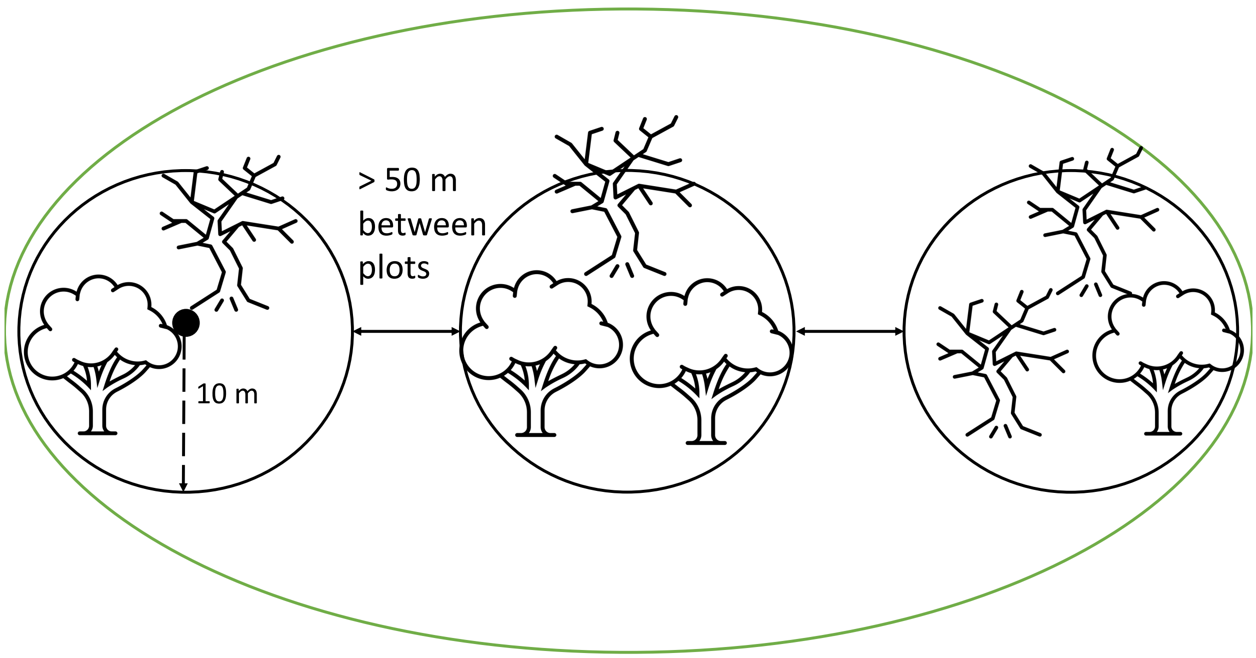
Supplement H. Respondents' risk sensitivity from both biocontrol through the use of parasitoid wasps and chemical control through the use of systemic insecticides to control emerald ash borer in parks in Northeast Georgia (n = 174).

	Median	Not at all concerned		Slightly concerned		Moderately concerned		Concerned		Very concerned		Factor Loading	Cronbach's alpha
		Number	%	Number	%	Number	%	Number	%	Number	%		
Risk sensitivity to biocontrol													
How concerned are you that…													0.96
Wasps used for biocontrol will attack native species in this park	Slightly concerned	36	20.7	52	29.9	44	25.3	27	15.5	15	8.6	0.70	
Wasps used for biocontrol will harm human health	Slightly concerned	56	32.2	55	31.6	30	17.2	21	12.1	12	6.9	0.96	
Wasps used for biocontrol will harm pets	Slightly concerned	54	31.0	53	30.5	28	16.1	28	16.1	11	6.3	0.92	
Risk sensitivity to chemical control													
How concerned are you that…													0.91
Chemical control will have unintended environmental impacts	Moderately concerned	8	4.60	41	23.6	39	22.4	46	26.4	40	23.0	0.88	
Chemical control will harm human health	Moderately concerned	16	9.20	42	24.1	36	20.7	44	25.3	36	20.7	0.92	
Chemical control will harm pets	Moderately concerned	15	8.62	39	22.4	35	20.1	47	27.0	38	21.8	0.92	

Supplement I. Locations of all plot centers across all sites in Northeast Georgia. Plots were geo-referenced using a handheld GPS.

Site	Plot 1		Plot 2		Plot 3	
Martin Farm Park	34.04406° N	84.06479° W	34.04363° N	84.06613° W	34.04302° N	84.06757° W
Cook Property	34.01904° N	83.52903° W	34.01784° N	83.52934° W	34.01773° N	83.52833° W
Galau Property	33.86031° N	83.24772° W	33.86001° N	83.24841° W	33.85986° N	83.24289° W
Shoal Creek Property	33.88874° N	83.29866° W	33.89137° N	83.29899° W	33.88702° N	83.29937° W
Oconee Forest Historic Site	33.73239° N	83.29964° W	33.73442° N	83.29005° W	33.73516° N	83.28990° W
Whitehall West	33.88846° N	83.36677° W	33.88846° N	83.36725° W	33.88838° N	83.36583° W
Whitehall East	33.88011° N	83.36040° W	33.88051° N	83.35986° W	33.88047° N	83.35932° W
Oconee Forest Creek	33.72306° N	83.27062° W	33.73164° N	83.27032° W	33.73115° N	83.27075° W
Oconee Forest North	33.78396° N	83.24224° W	33.78384° N	83.24301° W	33.78349° N	83.24347° W
State Botanical Gardens	33.90014° N	83.38711° W	33.89994° N	83.38640° W	33.89982° N	83.38567° W
George Pierce Park	34.05394° N	84.05349° W	34.05448° N	84.05384° W	34.05515° N	84.05396° W
Yellow River Park	33.78987° N	84.07100° W	33.79300° N	84.06958° W	33.79378° N	84.07068° W
McDaniel Farm Park	33.96978° N	84.12223° W	33.97133° N	84.12477° W	33.97142° N	84.12662° W
Oconee Heritage Park	33.76093° N	83.44205° W	33.76094° N	83.44305° W	33.76094° N	83.44466° W
Sandy Creek Nature Center	33.98137° N	83.38232° W	33.98146° N	83.38305° W	33.98186° N	83.38391° W

Supplement J. General diagram showing the plot design at each site in Northeast Georgia. We used at least 50 m of spacing between the edges of plots. Each plot was centered among areas containing ash to capture the heterogenous distribution of ash across the landscape.



Supplement K. Locations of all sites plotted with mean canopy dieback rating in Northeast Georgia. We determined that there were no clear spatial trends in EAB induced mortality and dieback across Northeast Georgia that we would need to account for in our analyses.