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Impacts of the emerald ash borer (EAB) eradication and tree mortality: potential for a secondary spread of invasive plant species

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Abstract Since the discovery of the emerald ash borer in 2002, eradication efforts have been implemented in an attempt to eliminate or contain the spread of this invasive beetle. The eradication protocol called for the removal of every ash tree within a 0.8 km radius around an infested tree. In 2005 this study was established to identify environmental changes attributed to the eradication program and measure subsequent shifts in forest community composition and structure. We conducted this study in Ohio and compared areas that received the eradication treatment (ash trees cut down), to areas that were left uncut, (ash still standing). The goal of this project was to identify how the plant community is responding in these two areas. The eradication protocol accelerated the formation and size of gaps within the forest and thus increased the duration and intensity of light penetrating through to the forest floor. In addition, the use of track vehicles for removal of cut trees resulted in significant soil compaction. The resultant plant community had greater species diversity (H'). When specific species composition differences were compared, an increase in the establishment of invasive plant species was detected in areas that received eradication efforts compared to those that did not. Invasive species accounted for 18.7% of the total herbaceous cover in this highly disturbed environment which included *Cirsium arvense*, *Rhamnus cathartica* and 2 species of *Lonicera*. In contrast, invasive species accounted for <1% of the total herbaceous cover in the undisturbed uncut areas.

Keywords Emerald ash borer · Invasive species · Eradication · Disturbance · Soil compaction · Light environment

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Introduction

The introduction of invasive species has negative effects on biological diversity at the local level (Pimentel et al. 2000; Sandlund et al. 1999; Vitousek 1990), and can lead to severe disruption of ecological communities (Johnson and Padilla 1996; Moller 1996; Primack 1993). It is estimated that the economic impact of invasive species on US agriculture, forestry, fisheries and human health is at least $134 billion annually (US Congress Office of Technology Assessment 1993). Eradication efforts to combat the spread of an invasive species present themselves as a feasible alternative. However, the cost-benefit analyses of eradication programs typically underestimate the cost (both financially and ecologically) and overestimate the benefit (Myers et al. 1998).

The introduction of forest pests has a profound impact on native plant community structure due to the lack of natural enemies or the lack of plant defense mechanisms (Mack et al. 2000). The gypsy moth (*Lymantria dispar* (L.)) is a well documented forest pest that has had devastating impacts on native forests wherever established (Abrams 1998; Campbell and Sloan 1977; Fajvan and Wood 1996; Liebhold et al. 1995) including significant mortality among stressed trees during a single defoliation event (Davidson et al. 2001). Since the first documented outbreak of gypsy moth in 1865, eradication efforts have been heavily implemented to eliminate this pest. While eradication is no longer a feasible option, control mechanisms to manage the spread of the gypsy moth continue today; annually the control of gypsy moth populations cost $11 million (Campbell and Schlarbaum 1994).

A recent example of a successful invasive species to the continental United States is the emerald ash borer (EAB), *Agrilus planipennis* Fairmaire (Buprestidae) This metallic green wood-boring beetle is native to Asia (Haack et al. 2002) and relies on ash species (*Fraxinus* spp.) to complete its life cycle. Larvae feed on the cambium layer of the ash interfering with the translocation of water and nutrients. Once infested, most ash trees typically dies within 2–4 years (Herms et al. 2004).

The first EAB infestation in the United States was identified in Detroit, MI in 2002, but its introduction likely occurred at least 5–10 years prior (McCullough and Roberts 2002). It is likely to have been imported as larvae in ash wood crating or pallets (Stone et al. 2005). Since discovery, EAB has killed over 15 million ash trees in Michigan (Poland and McCullough 2006) with even greater mortality estimated across all other states to date. Currently EAB is known to be established in 13 states and Canada. All of the 16 *Fraxinus* species native to the United States are susceptible to and attacked by EAB. A few of these species have protective status, including *Fraxinus profunda* (endangered in the states of New Jersey and Pennsylvania and threatened in Michigan) and *Fraxinus quadrangulata* which is threatened in Iowa and Wisconsin (USDA 2008). In Ohio where this study was conducted, there are five native ash species (*F. americana*, *F. pennsylvanica*, *F. profunda*, *F. nigra* and *F. quadrangulata*). There are approximately 279 million ash trees in Ohio (USDA Forest Service) about 6% of all trees by number and volume (Widmann 2008). EAB has been found in 47 of Ohio's 88 counties. No landscape or forest attributes have been identified to date that make a stand susceptible to EAB attack (Smith 2006).

In 2002 the USDA Animal and Plant Health Inspection Service (APHIS) established a Science Advisory Panel to develop a plan to contain and eventually eradicate EAB. Measures to control the outbreak include tree removal and the establishment of quarantine areas, which regulate the potential movement of infested wood materials (USDA-APHIS 2003; Stone et al. 2005). The funding for this multistate level eradication effort came with the support of APHIS. According to the APHIS protocol, after an infested tree has been identified, every ash tree (>2.5 cm DBH) within a 0.8 km radius is felled and chipped to destroy EAB larvae (Poland and McCullough 2006; Stone et al. 2005). The remaining stumps are treated with herbicide and the chips are burned in an electricity co-generation plant (Poland and McCullough 2006). This eradication protocol as it is applied on site is supposed to have as minimal an impact as possible. However these measures create immediate environmental changes to the habitat with abrupt environmental consequences.

The eradication protocol may influence the composition of existing plant communities by unnaturally accelerating the formation of gaps in the forests due to the total removal of all ash trees. This may increase the duration and intensity of light reaching the forest floor. Second, the heavy, tracked vehicles used to remove trees may lead to soil compaction, as most soils become compacted during the first few passes of a vehicle (Lockaby and Vidrine 1984; Shetron et al. 1988). Our study performed a comparison of plant communities between areas that received an eradication protocol treatment to areas that received no treatment.

This project identifies the biological consequences of implementing the emerald ash borer eradication program in Ohio. What are the shifts in dominance and distribution of plant community species? A secondary spread of invasive plants is predicted in these highly disturbed cut areas.

Materials and methods

Study site description

This study was conducted at Pearson Metropark (Pearson) in northwest Ohio (Lucas County). Established in 1935, Pearson is 620-acres of the relic Great Black Swamp, a glacial swamp forest. Situated on the eastern edge of the city Oregon, it is surrounded by residential neighborhoods and agricultural fields. The forest in Pearson is characterized by dense hardwoods dominated by several ash, maple and hickory species (Table 1). The soil substrate is composed of heavy clay, specifically >90% of the park is Latty silty clay (Soil Survey Staff 2008). In early spring of 2005 the Ohio Department of Agriculture identified three EAB infested trees in Pearson which was identified as the leading eastern edge of the infestation, and was selected to receive the most intensive eradication efforts.

Implementation of EAB eradication began in Pearson in April of 2005 and lasted approximately 3 months. Following the APHIS eradication protocol described above, self-propelled, tracked, hydro-axes were used to cut down ash trees within 0.8 km radius of an infested tree. Some tree cutting truck chippers were used on-site to chip small branches and limbs, but skidders were primarily used to transport the trees out of the forest to staging areas. Logs were then transported to an off-site location for processing by a large drum chipper mill. Preexisting trails were used when possible with additional “cut-roads” created as needed to access all ash trees. Approximately 5,000 total trees, ranging from saplings to trees >40 cm DBH, were felled from a total of 83 acres within Pearson.

Experimental design and biotic sampling

In April 2005, prior to the initiation of the EAB eradication protocol (tree removal) using a block

Table 1 Relative density, frequency, coverage and importance values (IV) for the 10 most important tree species across the entire forest

Species	Relative density	Relative frequency	Relative coverage	Importance value (IV)	Importance value without ash
Fraxinus pennsylvanica (ash)	21.64	10.81	28.84	61.30	–
Tilia americana (basswood)	13.70	11.71	12.53	37.94	48.22
Fraxinus rubrum (red maple)	12.33	8.11	13.36	33.80	43.61
Ulmus americana (slippery elm)	10.41	7.21	8.69	26.31	33.58
Acer negundo (box elder)	7.40	8.11	5.66	21.17	26.49
Acer saccharinum (silver maple)	6.58	4.50	11.68	22.76	29.86
Carya ovata (shagbark hickory)	4.66	6.31	3.14	14.11	17.43
Carya cordiformis (bitternut hickory)	3.84	6.31	2.43	12.57	15.38
Carpinus caroliniana (blue beech)	2.74	4.50	0.66	7.91	9.48
Crataegus spp. (hawthorn)	2.74	4.50	0.84	8.09	9.73

The last column projects the changes in dominant canopy structure when all of the ash trees are removed from IV calculations design, eight 20 × 25 m plots were established to assess the status of the forest structure. Plot locations were randomly selected by GPS coordinates and included a 50–100 m buffer between plots and a 50 m buffer zone from any trail or forest edge. By July 2005 all ash tree-cutting eradication practices ceased due to lack of funding; therefore, not all ash trees within Pearson Metropark were cut. None of the original 8 plots had ash trees removed and were therefore designated as uncut comparative control plots. Based on the total area that did receive the eradication protocol, only 6 additional 20 × 25 m plots were able to be established and were designated as the cut plots. These 6 cut plots had buffer zones reduced to 50 m between plots and 10–20 m from trail or forest edge. A vegetational assessment of 14 plots (8 uncut plots and 6 cut plots) was conducted at the end of July 2005 and again in July 2006. Data collected included measurements of the canopy composition and structure, and the herbaceous understory. All trees, greater than 3 cm diameter at breast height (DBH), within the 20 × 25 m plots were counted, identified to species, and given a % canopy cover. This information was used to determine the Importance Value of each tree species which is calculated by incorporating the number of trees per species that exist in an area, the overall frequency of each species in the forest and the total coverage, based on basal area, of each species (Relative abundance + Relative frequency + Relative coverage = Importance value) (Curtis and McIntosh 1951). Within each of the 14 plots, seven 1 × 1 m subplots were established to determine the herbaceous understory community. Six of the subplots were placed around the perimeter of the 20 × 25 m plot (4 corners and 2 at 12.5 m along the edge) and an additional subplot was located in the center of the plot. Every plant encountered in these subplots was identified to species and given a % cover measurement. Voucher specimens of all plants were collected for proper species identification and deposited in the Kent State University Herbarium.

Abiotic sampling

Abiotic measurements were taken to determine changes in the light environment and the effect of soil compaction. The light environment was assessed by taking six hemispherical photographs of the canopy from every 20 × 25 m plot (4 corners and 2 from a center transect). These 180° fisheye images are taken from a height of 1 m off the forest floor and show a 10 m diameter view of the canopy (Oberbauer et al. 1993; Whitmore et al. 1993). Fisheye images were imported into Hemiview and used to calculate changes in solar radiation regimes. Hemiview determines the Global Site Factor (GSF), the proportion of global radiation (direct + diffuse) that occurs in the open sky compared to that which occurs under the forest canopy, and identifies the number and duration of sunflecks. A sunfleck occurs when direct radiation penetrates through to the forest floor. A sunfleck calculation scans the path of the sun from sunrise to sunset at 30 s intervals and determines start and stop times as well as the amount of radiation that occurs during each sunfleck.

Soil compaction measurements were taken in all fourteen plots. A soil penetrometer was use to determine the pressure in pounds per square inch (PSI) needed to penetrate through the soil profile. These compaction measurements were converted to kilopascal (kPa). Six readings (4 corners and 2 from a center transect) were taken at five consecutive depths 7.6, 15.2, 22.9, and 38.1 cm, in each plot. Soil temperature and percent moisture content were taken in the same six locations as compaction using a soil moisture meter (Aquaterr T-300).

Statistical analysis

Multivariate analysis of variance was used to determine the effects of treatment (cut and uncut), and year (2005 and 2006) on the light environment, and the effects of treatment and depth of soil profile (7.6, 15.2, 22.9, 38.1 cm) on soil compaction. One-way analysis of variance was used to examine the effect of treatment on species diversity. Plant community composition was analyzed with ordination using canonical correspondence analysis (CCA) (McCune and Medford 1999). Ordinations were performed with percent cover data from 73 species, with abiotic measurements (GSF, number of sunflecks, average duration of sunflecks, soil compaction, soil moisture and soil temperature). Treatment (cut and uncut) was also included as a covariate.

Results

Ash (*Fraxinus* spp.) is the dominant tree species in the Pearson Metropark forest, comprising the highest Importance Value (IV) (86.2% in the Cut plots and 42.2% in the Uncut plots) (Table 1). When ash species are eliminated from IV calculations all other tree species IV increase by an average of 39% in cut plots and 16% in uncut plots. The future projected dominant canopy species were similar in both treatments and include *Tilia americana*, *Acer rubrum*, *Acer saccharinum*, and *Ulmus rubra* (Table 2).

Results from 2005 and 2006 show that removal of ash trees resulted in significant differences in the light environment (Fig. 1). These included an increase in duration and intensity of light reaching the forest floor. There was a greater GSF measurement found in cut plots compared to uncut plots ($F_{1,162} = 56.87$, $P < 0.0001$) (Fig. 2). While the difference in the number of sunflecks was not significant ($F_{1,162} = 1.99$, $P = 0.16$), approximately 1.5 more sunflecks occurred in uncut plots compared to cut plots (Fig. 3a). However, while fewer sunflecks occurred in the cut plots, the duration of each sunfleck lasted on average more than 1.5 min longer ($F_{1,162} = 9.97, P = 0.002$). This is the equivalent of 24 min more of direct radiation penetrating through to the forest floor that day (Fig. 3b). There was a significant year effect for GSF, number of sunflecks and average duration of sunfleck ($F_{1,162} = 17.47$, $F_{1,162} = 13.37$, $F_{1,162} = 27.64$, $P < 0.0003$, respectively), such that all light measurements were slightly greater in 2005

Table 2 Importance values are calculated for the 10 tree species mentioned in Table 1 for each treatment

Species	Uncut plots	Cut plots with ash	Cut plots without ash
Ash	42.22	86.18	–
Basswood	38.95	36.68	53.29
Red maple	40.96	24.41	35.87
Slippery elm	34.53	15.68	21.80
Box elder	25.79	15.05	20.84
Silver maple	20.80	25.34	38.78
Shagbark hickory	17.44	9.58	13.19
Bitternut hickory	10.68	15.13	20.87
Blue beech	4.76	12.32	16.24
Hawthorn	6.21	10.87	14.02
Totals	242.35	251.24	234.89

The first column is the IV's for the uncut Control plots. The IV's for the cut plots are calculated twice: first with ash trees present (calculated with DBH estimated as the shortest diameter of a cut stump) and second without ash trees, representing the immediate shift in canopy dominance after the eradication cuttings

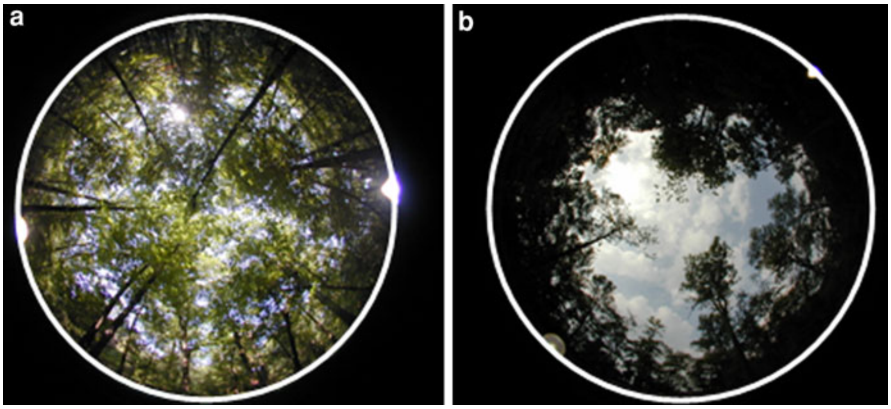

Fig. 1 Example of Hemispherical pictures. Picture **a** was taken from a portion of an uncut plot which has 6 ash trees that comprise 40% of the canopy cover from within that plot. Picture **b** was taken from a cut plot that had 6 ash trees cut down during the EAB eradication

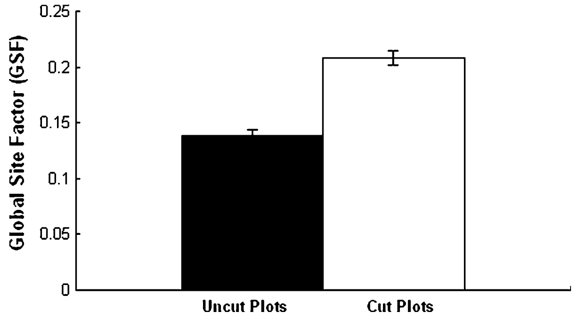

Fig. 2 The proportion of radiation (Global Site Factor) that occurs under the forest canopy is greater in cut than in uncut plots ($P < 0.0001$). *Error bars* represent ± 1 SE of the mean

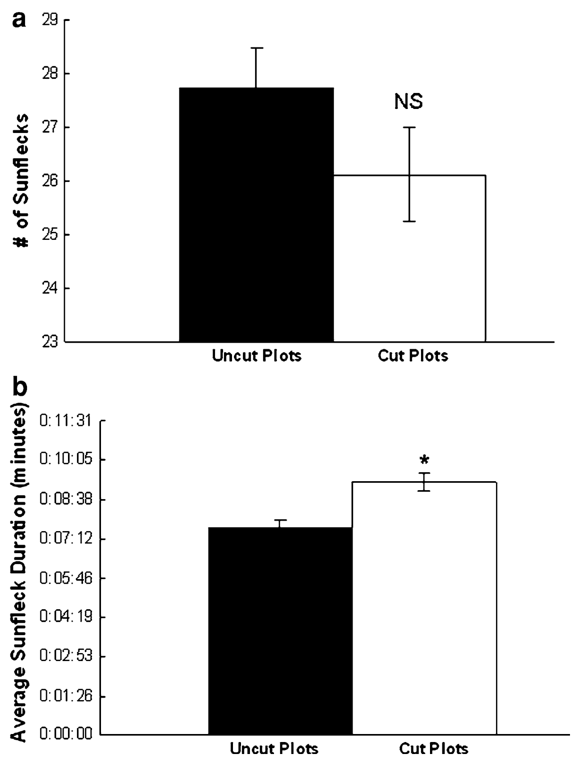

Fig. 3 The number of sunflecks (**a**) and average duration of each sunfleck (**b**) in uncut and cut plots. *Error bars* represent ± 1 SE of the mean. Significant difference at ($P = 0.0001$)

immediately following the tree cutting eradication efforts.

There where significant differences in soil compaction between cut plots and uncut plots ($F_{1,404} = 44.61$, $P < 0.0001$) in 2005. The degree of soil compaction was greater in the cut plots with significant differences detected at each depth of the soil profile (Fig. 4). Soil compaction results remained similar in 2006 with greater compaction detected in the cut plots than in the uncut plots ($F_{1,404} = 106.80$, $P < 0.0001$).

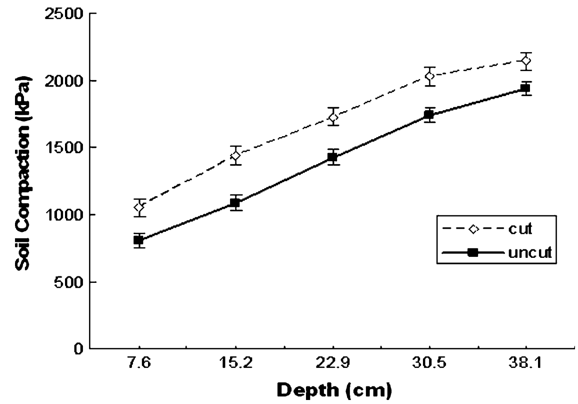

Fig. 4 Soil compaction in cut and uncut plots at five different depths in the soil profile from 2005. *Error bars* represent ± 1 SE of the mean

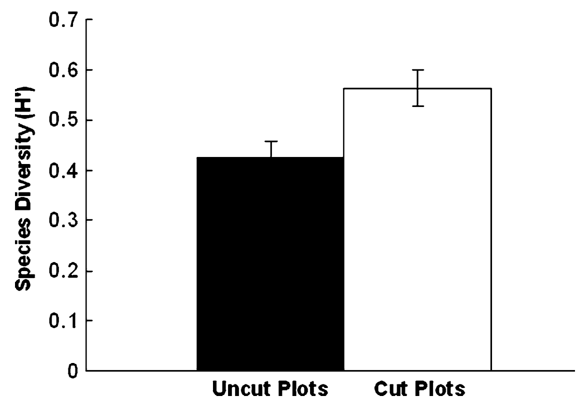

Fig. 5 Shannon's Diversity Index (H') for understory plant community in uncut and cut plots. Significantly greater diversity in cut plots ($P = 0.005$). *Error bars* represent $\pm$ 1 SE of the mean

Species diversity was assessed for the understory plant community using Shannon's Diversity Index (H') in 2006. Analysis of variance revealed higher diversity in the cut plots ($H' = 0.56$) where there was greater disturbance compared to the uncut plots or undisturbed controls ($H' = 0.43$) ($F_{1,96} = 8.43$, $P = 0.005$) (Fig 5). CCA analysis was performed to identify the species-specific distribution patterns. Differences in species composition were detected between the cut and uncut plots (Fig. 6). The uncut and cut plots were both strongly correlated with Axis

1 (both $R^2 = 0.574$) determining each end of the scale. In addition, GSF was also associated with Axis 1 ($R^2 = 0.229$) in the direction of the cut plots. The abiotic factors that correlated the highest with Axis 2 were as follows: number of sunflecks ($R^2 = 0.241$), average duration of each sunfleck ($R^2 = 0.256$), and soil moisture ($R^2 = 0.305$). While only 3% of the variance was explained by either Axis, it was the location of non-native plant species that was of interest. In Fig. 6 each dot represents a species or a cluster of species with non-native plants identified by X's. Out of 73 species total, 13 (17.8%) were non-native; all but one species were present within cut plots including 10 that were found exclusively within cut plots. The total percent cover by understory vegetation was not significantly different between the uncut and cut plots ($P = 0.81$); however, the compositional difference of community members was significantly different. There were only 3 invasive plant species, which made up less than 1% of the total herbaceous vegetation cover for all of the uncut plots combined (Table 3). *Alliaria petiolata*, the dominant invasive species in uncut plots, was found in 7 of 8 plots but on average had 2.0% cover per plot. *Rosa multiflora*, the only species not found in the cut plots, was merely found in one subplot of a single uncut plot. In contrast, there were 12 invasive plant species in the cut plots comprising 18.7% of the total herbaceous cover (Table 3). All species listed in Table 3 are non-native but not all of them are

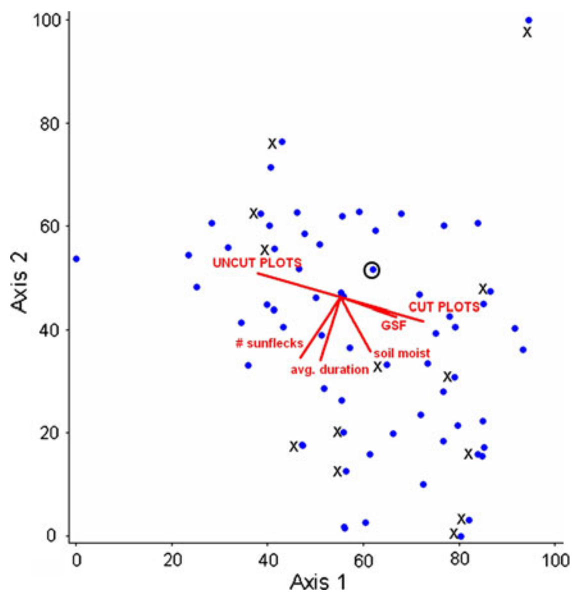

Fig. 6 Canonical Correspondence Analysis ordination plot generated with the percent cover of 73 plant species from 1 × 1 m subplots. An *X* denotes location of 13 non-native invasive species. The *black circle* identifies the location of *Fraxinus* spp. seedlings

Table 3 Invasive plant species in the understory of uncut and cut plots listed by total percent cover from all 1 × 1 m understory subplots

Species	Total % cover	Number of plots where species occur
Uncut plots		
Alliaria petiolata (garlic mustard)	0.8	7
Urtica dioica (stinging nettle)	0.1	2
Rosa multiflora (multiflora rose)	0.03	1
Cut plots		
Cirsium arvense (Canada thistle)	10	6
Lonicera japonica (Jap. honeysuckle)	3.2	1
Rhamnus cathartica (buckthorn)	3.0	5
Lonicera spp. (bush honeysuckle)	1.1	2
Taraxacum officinale (dandelion)	0.7	5
Arctium minus (burrdock)	0.2	2
Alliaria petiolata (garlic mustard)	0.2	4
Glechoma hederacea (gill-over-the-ground)	0.1	1
Prunella vulgaris (common selfheal)	0.1	2
Persicaria maculosa (ladies thumb polygonum)	0.04	1
Urtica dioica (stinging nettle)	0.02	1
Melilotus officinalis (yellow sweet clover)	0.02	1

Number of plots where an invasive species was found (8 total uncut plots and 6 total cut plots). Invasive species made up <1% of the total herbaceous cover for the uncut plots and 18.7% for the cut plots. Some of the non-native species listed are not generally considered invasive considered invasive however, 93% of the invasive species cover is attributed to 4 highly invasive plant species. Over half of that cover was attributed to a single species *Cirsium arvense* with *Rhamnus cathartica* and 2 species of *Lonicera* constituting another 40%.

Ash seedlings were found in the understory of all 8 uncut and all 6 cut plots and comprise 5.2 and 6.3% of the total herbaceous cover for those treatments, respectively in 2006 (Fig. 6).

Discussion

Long term impacts of the EAB eradication protocol are unknown, however, differences in the abiotic environment were immediately detected after the implementation of the eradication protocol in 2005. The higher light environments caused by artificially created gaps and the greater degree of soil compaction created a disturbance in the forest which facilitated a secondary spread of invasive plant species. It should be noted that currently eradication is no longer a program objective of USDA APHIS Emerald Ash Borer Program as such wide-scale tree removal is not supported.

Herbaceous plants are often used to signify habitat quality or health (Andreas et al. 2004) as plants are sensitive to disturbance events, natural or anthropogenic in nature (Collins et al. 1985). Factors affecting the composition and diversity of the understory herbaceous community include canopy composition and light availability (Crozier and Boerner 1984; Hausman 2001). Light availability is often a limiting resource for plant growth. It has been shown that invasive plants species can leaf out earlier (Harrington et al. 1989) and are capable of attaining higher photosynthesic capacity than native species (Pattison et al. 1998; McDowell 2002) thus outcompeting native plants for light. In forested ecosystems a short duration sunfleck (5–7 min) can contribute 47–68% of the potential light for understory seedlings and saplings to live (Canham et al. 1990). From this study, the disturbed cut plots have significantly higher light environments with average sunfleck duration lasting over 9 min. In these high light plots there are greater numbers of invasive species which also make up a greater proportion of the understory cover. Non-native plants are often found in areas of previous disturbance events. Small and McCarthy (2002) found a greater frequency of non-native species during the spring in resource rich areas with a previous clearcut disturbance whereas fewer non-native species were found in mature forests with greater canopy cover.

Consistent with our results, it has been shown that the highest degree of soil compaction occurs in the top 30 cm of the soil profile which is also where most plant root biomass occurs (Wingate-Hill and Jakobson 1982). The effects of soil compaction include a decrease in soil aggregates, soil porosity, and filtration capacity and an increase in soil bulk density, runoff and erosion. Severe soil compaction has been shown to inhibit germination and woody seedling growth (Kozlowski 1999). In addition to the interference with forest regeneration, soil compaction can also contribute to the mortality of standing trees. From a post partial harvest study, proximity to skid trail was the most important determinant for elevated residual tree mortality with soil compaction identified as the principle cause (Thorpe et al. 2008). The soils in this study have a high clay content and are likely to remain compacted for years to come. As a result, this soil compaction will constrain forest regeneration and has the potential to restructure both the understory community as well as canopy composition.

It is important to note at this time that it is unclear whether the increase in invasive species is attributed to an increased light environment or to greater soil compaction or to the combination of factors. The continued spread of EAB will ultimately increase mortality of ash trees and thus increase gap formation and light environment. This ash mortality however occurs more slowly over a period of about 6 years (Knight et al. 2007). We are currently identifying subsequent changes in the understory plant community and the degree of invasive species establishment under these slower EAB induced ash dieback. However, eradication efforts potentially caused a compounding negative effect by creating an increase in the light environment all at once and physically disturbing the soil which subsequently facilitated a secondary spread of invasive plant species. These results address the importance of the magnitude of scale or intensity of disturbance on community structure. Often invasive plant species have the ability to dominate and outcompete native species in disturbed habitats. Ross et al. (2002) found that increases in anthropogenic disturbance in various fragment sizes reduces native plant species richness and increases invasive species richness. Furthermore, it is unclear whether the invasive species that we are detecting in this study are "new" arrivals from an external source or were already present in the seed bank. It is possible that the source seeds of these invasive species were introduced as hitchhikers on the vehicles used during EAB eradication. Ongoing research compares differences in seed bank composition between eradication zones and control areas to determine potential invasion pathways. Addressing these concerns will identify patterns of invasive species establishment at various scales of disturbance as well as disturbance tolerance thresholds of the native plant species.

The future of ash trees in this forest depends upon the presence of ash in the seed-bank and continued regeneration. While ash saplings were found throughout the forest and within each of the treatments of this study, little is known about the longevity of ash seeds in the seed-bank. The ultimate persistence of ash as a canopy tree based on the presence of ash saplings may be overly optimistic. Ash saplings are under a secondary pressure caused by deer herbivory. Most of the ash saplings that occurred within the cut plots showed a shub-like growth form. These ash saplings with stunted growth and multiple branching stems demonstrate characteristic signs of previous browse events (Hausman, personal observations). Under continued deer herbivory pressure, one can only speculate as to the persistence of ash in this system. The eastern hemlock is also particularly susceptible to deer overbrowsing pressure (Frelich and Lorimer 1985). In areas infested with the hemlock wooly adelgid (HWA), Weckel et al. (2006) have documented a lack of hemlock sapling regeneration across a 40 year time span. Furthermore, there was also a lack of regeneration in white ash (*Fraxinus americana*) in the HWA infested areas. This study demonstrates the potential indirect effect invasive forest species have on non-target species. HWA and EAB represent parallel examples of high impact invasive pests capable of restructuring forest composition. The combined pressures of the forest pest with increasing deer browse may ultimately lead to the functional extirpation of these two trees from their respective forest systems.

As of 2006 the USDA-APHIS spent $100 million dollars on attempted eradication of EAB, research and reforestation efforts (Lucik and Redding 2006). The emerald ash borer eradication efforts not only failed at their attempt to control the spread of this beetle but ultimately facilitated a secondary spread of invasive plant species. The initial colonization of invasive plants exposes vulnerability in the habitat with potentially lasting and detrimental impacts for the native plant community. Once established invasive plant species can alter ecosystem functioning and reduce native plant species abundance and survival. For example, the invasive Amur honeysuckle (*Lonicera maackii*) negatively affects plant species richness and abundance of herbaceous and tree seedlings growing below *L. maackii* crowns (Collier et al. 2002). Even though lasting impacts on the environment are yet to be determined, preliminary evidence indicates that at least some detrimental damage (i.e. soil compaction) caused by the eradication of EAB will likely take decades or longer to recover. This project exposes a potential and yet substantial concern for government agencies and land managers. With the likely continued spread of the emerald ash borer, one must question whether control mechanisms should be implemented in sensitive natural areas. Thus far EAB eradication efforts have been expensive and labor intensive with detrimental ecological consequences. The establishment of invasive species in the wake of eradication efforts adds a new variable for calculating future cost-benefit analysis of control or management objectives. Therefore, in the pursuit of finding suitable forest pest eradication measures, greater consideration should be given to determining disturbance intensity and the ecological impacts that influence invasive plant species establishment. Sometimes not implementing eradication is the best option.

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