

Effects of prescribed fire and stem-injection herbicide on *Ailanthus altissima* demographics and survival



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ABSTRACT

Ailanthus altissima (Mill.) Swingle, tree-of-heaven, Chinese sumac, stink tree) is a nonnative invasive tree that is common throughout much of the eastern United States. It can invade and expand dramatically when forests are disturbed. Anecdotal evidence suggests that fire might facilitate its spread, but the relationship between fire and this prolific invasive tree is poorly understood. To better understand the impacts of fire on *Ailanthus*, we conducted a study at Tar Hollow State Forest in southeastern Ohio, USA where *Ailanthus* is widely distributed. Our objective was to quantify the direct effects of prescribed fire on the demography of *Ailanthus* populations, with and without the pre-burn application of the stem-injected herbicide, imazapyr. We quantified demographic attributes (survival, density, and sprouting) of *Ailanthus* germinants, seedlings, saplings, and trees in a field experiment with four treatment combinations: (1) no fire and no herbicide, (2) herbicide, (3) fire, and (4) herbicide plus fire. *Ailanthus* stems > 3 cm dbh were stem-injected with the herbicide, imazapyr in late summer to early fall followed by prescribed fire in April. We determined that herbicide plus prescribed burn treatment was highly effective in killing large saplings and trees, with no resprouting evident after four growing seasons. In the season immediately after the burn with and without herbicide treatment, small *Ailanthus* sapling and seedling abundance increased, but did not persist. Over time, *Ailanthus* germinants and sprouts from top-killed stems were poor competitors with faster-growing post-fire woody regeneration as forest floor shading increased. In addition, the late growing season application of imazapyr was highly effective in killing *Ailanthus* and subsequent sprouts. This study, the first to quantify the direct effects of fire on *Ailanthus*, demonstrates that prescribed fire alone does not appear to facilitate the spread of *Ailanthus*.

1. Introduction

The use of prescribed fire on public lands has increased rapidly in the last decade as a tool to favor oak (*Quercus*) regeneration, enhance wildlife habitat, and reduce fuel buildup; however, its use may also increase the risk of invasion and expansion of non-native plant species (NNIS). Wildfire has often been shown to favor some NNIS in the western United States (Zouher et al., 2008). These effects are often related to invasion establishment and positive feedback cycles in relation to fire of non-native grasses and forbs (Brooks et al., 2004). In the eastern United States, NNIS are abundant and represent one of the greatest management and ecological challenges for natural resource managers (Webster et al., 2006). However, because large and high severity wildfires are infrequent and the widespread use of prescribed burning on public land is relatively recent, much less is known about the effects of fire on invasives in the eastern USA (Dibble et al. 2008). Research that characterizes how woody plant invasives respond to management practices such as prescribed fire and thinning are just emerging (Pile et al., 2017a, 2017b; Rebeck et al., 2017).

Ailanthus altissima (Mill.) Swingle, tree-of-heaven, Chinese sumac, stink tree) is an invasive of particular prominence in the eastern United States, yet there is limited knowledge of forest management impacts on

its growth and reproduction. Results from limited studies and anecdotal information suggest that fire may increase *Ailanthus* density in North America (Pomp 2008; Guthrie et al., 2016). In the Insubric region of the southern Swiss Alps, Maringer et al., (2012) documented that *Ailanthus* and black locust (*Robinia pseudoacacia*) invaded and grew in high abundance within areas of high-light and bare soil created following a high intensity wildfire.

Ailanthus has been present in North American landscapes for over two hundred years (Hu, 1979) and is widely distributed throughout the East and Midwest. It is found on all continents with the exception of Antarctica (Kowarik and Säumel, 2007). It is most often abundant in open sites such as roadsides (McAvoy et al., 2012). However, it can invade disturbed sites in forests via prolific wind-dispersed seed, form a persistent seed bank (Rebeck and Jolliff, 2018) and expand via clonal growth. High mortality of *Ailanthus* seedlings has been observed under closed canopy conditions (Kowarik, 1995; Kota et al., 2007). Carter and Fredericksen (2007) reported lower seedling densities in mature forest stands (320 ± 147 seedlings ha⁻¹) compared with recently logged stands ($10,138 \pm 6692$ seedlings ha⁻¹) on non-industrial private forestlands of Virginia, USA.

Ailanthus is present in many oak forest landscapes where the use of prescribed fire is increasing to meet multiple land management

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objectives including the promotion of oak advanced reproduction. At the Tar Hollow State Forest in southeastern Ohio, we witnessed the rapid expansion of *Ailanthus* following thinning and burning treatments on a study assessing the role of fire and fire surrogates in oak regeneration (Hutchinson et al., 2004). A better understanding of how *Ailanthus* responds to prescribed fire is needed so that proactive control strategies can be developed and integrated into prescribed fire and timber management programs at a landscape level.

Very little is known about the direct and immediate effects of fire on *Ailanthus*. Lewis (2007) reported that *Ailanthus* saplings were easily top-killed by fire but resprouted to preburn density levels within six months. The USDA Forest Service Fire Effects Information System (FEIS) proposes that given its prolific sprouting capacity, fire would be likely to promote *Ailanthus* populations (USDA Forest Service, 2018). Managers have reported observing increases in *Ailanthus* via seed germination immediately following fires. However, in landscapes with very small populations of *Ailanthus*, it typically does not invade burned sites (e.g., Hutchinson et al., 2005). In addition, prescribed fire has been found not to be a good predictor of *Ailanthus* presence (Rebbbeck et al., 2017). It remains unknown whether a post-burn *Ailanthus* expansion would inevitably occur if the propagule pressure was high. It may be that fire, by reducing litter and increasing light to the forest floor, creates improved conditions for *Ailanthus* establishment, as other disturbances have been shown to do (Kota et al., 2007).

Our first goal was to better understand the direct and indirect effects of prescribed fire on *Ailanthus* trees and saplings and regeneration dynamics. The second was to determine the effectiveness of a stem injection herbicide treatment of trees and large saplings prior to a prescribed burn in reducing post-fire sprouting. Because *Ailanthus* is considered a high light-demanding species, light levels were monitored over the course of the study to determine how changes may influence *Ailanthus* seedling establishment and resprouting success. In addition, the abundance of non-*Ailanthus* understory vegetation was measured post-treatment to estimate its competitive relationship to *Ailanthus* regeneration. Fire severity was quantified to characterize its impact on the mortality of varying *Ailanthus* stem sizes. We hypothesized that herbicide treatments would reduce the risk of *Ailanthus* expansion after prescribed fire by reducing sprouting and seed source. This basic ecological information is critical to better ascertain *Ailanthus* response to prescribed fire and to develop management prescriptions to reduce the threat of post-fire invasion.

2. Materials and methods

2.1. Study site and treatments

This study was conducted at the 6618 ha Tar Hollow State Forest (THSF; 39°21'N; 82°46'W), located within the Southern Unglaciated Allegheny Plateau in southeastern Ohio. The topography is dissected, consisting of sharp ridges, steep slopes, and narrow valleys. The uplands consist of mixed-oak overstory and the lower slopes and coves are mixed mesophytic hardwood forests (Braun, 1950). After European settlement in the early 1800s, this landscape, like most in the region, had a history of exploitive timber harvesting and farming. By the 1930s, when land was being purchased to create THSF, much of the land had become severely eroded and unproductive for agriculture. Today, THSF is almost completely forested and has had an active timber harvesting program, a component of its multiple-use management, for more than 50 years. Utilizing Ohio Department of Natural Resources (ODNR), Division of Forestry digital and historic paper timber management records (earliest records from 1938), we identified potential study areas which had: (1) no recent heavy (shelterwood or clearcut) timber harvest activity within the last 20 years; and (2) no recent fire history (no prescribed burn or wildfire within the last twenty years). Seventy-five percent of the 39 experimental plots were located within areas that had been selectively harvested or lightly thinned in the recent past,

Table 1

Pretreatment vegetation composition and distribution in summer 2009 with study plots at Tar Hollow State Forest.

Variable	
A. <i>Ailanthus</i> density (stems ha⁻¹)	
Trees and saplings (> 3 cm dbh)	622.2 ± 71.9
Large seedlings (≤ 3 cm dbh)	6498.8 ± 1892.4
Germinants	7644.2 ± 1532.3
B. Overstory* basal area (m² ha⁻¹)	
Total	29.3 ± 1.3
<i>Ailanthus</i>	8.2 ± 1.1
Oak-hickory	6.8 ± 0.9
Shade intolerants ^a	8.6 ± 1.3
Shade tolerants ^b	5.2 ± 0.7
C. Saplings (> 3–9.9 cm dbh)	
Dominant species group (% of plots)	
Shade intolerants	23.1
Shade tolerants	43.6
Mixed shade-intolerants and tolerants	30.6
No dominant non- <i>Ailanthus</i> species group observed	2.6
D. Shrubs, dominant species (% of plots)	
None	33.3
Spice bush, <i>Lindera benzoin</i>	51.3
<i>Rubus</i> spp.	5.1
<i>Smilax</i> spp.	5.1
Others (understory trees) ^c	5.1

* Overstory basal area represented by trees ≥ 10 cm dbh.

^a Shade intolerants (other than oak (*Quercus* spp.) and hickory (*Carya* spp.)) included yellow-poplar (*Liriodendron tulipifera*), sassafras (*Sassafras albidum*), white ash (*Fraxinus americana*) aspen (*Populus* spp.), hackberry (*Celtis occidentalis*), Eastern redbud (*Cercis canadensis*), black cherry (*Prunus serotina*), black walnut (*Juglans nigra*), black locust (*Robinia pseudoacacia*), and pitch pine (*Pinus rigida*).

^b Shade tolerants included red maple (*Acer rubrum*), sugar maple (*A. saccharum*), elm (*Ulmus* spp.), sourwood (*Oxydendrum arboreum*), American basswood (*Tilia americana*), and blackgum (*Nyssa sylvatica*).

^c Others – 1 plot was dominated by paw paw (*Asimina triloba*) and 1 plot was dominated by Eastern redbud.

(7–32 years ago), while 25% had no record of timber harvest. Slopes averaged 29.8% ± 1.7 and most plots were located on S-facing aspects.

Prior to treatments, *Ailanthus* (≥ 10 cm dbh) represented 28% of overstory basal area (BA) within sampling plots. Despite the location of plots on ridgetops and upper slopes, mesophytic tree and shrub species were common. Oak and hickory (*Carya* spp.) represented 23.2% of overstory BA, while other shade-intolerant tree species, primarily yellow-poplar (*Liriodendron tulipifera*) represented 29% (Table 1). Shade-tolerant species such as sugar (*Acer saccharum*) and red maple (*A. rubrum*) represented 17.7% of the total overstory BA. Shade-tolerant species were dominant in the sapling size class (> 3–9.9 cm dbh) in 43.6% of the plots. Spicebush (*Lindera benzoin*) was the dominant shrub in 51% of the plots, while 33% of the plots had no dominant shrub.

Pre-treatment densities of *Ailanthus* trees and saplings combined averaged 622 ± 72 stems ha⁻¹. Mean *Ailanthus* tree dbh ranged from 13.3 to 17.2 cm while mean sapling dbh ranged from 5.8 to 7.7 cm. Large *Ailanthus* seedlings (< 3 cm dbh) density averaged 6499 ± 1892 stems ha⁻¹ while germinant density averaged 7644 ± 1532 stems ha⁻¹ (Table 1).

A repeated measures nested incomplete split plot design was used to test the effects of prescribed fire (whole plot unit) and stem-injection herbicide treatments (subplot unit) on *Ailanthus* populations. Year (pre- and post-treatment) was the repeated measures variable. Four treatments—burn (B), herbicide (H), herbicide + burn (H + B), and control (no burn + no herbicide, C) were replicated in ten 10-m radius plots. In order to increase the likelihood of a successful burn, ridgetop and upper slopes with sufficient litter fuels were prioritized for ignition, while mesic areas and drainages were excluded. Plot locations were further selected based on the presence of a minimum of 5–10 *Ailanthus*

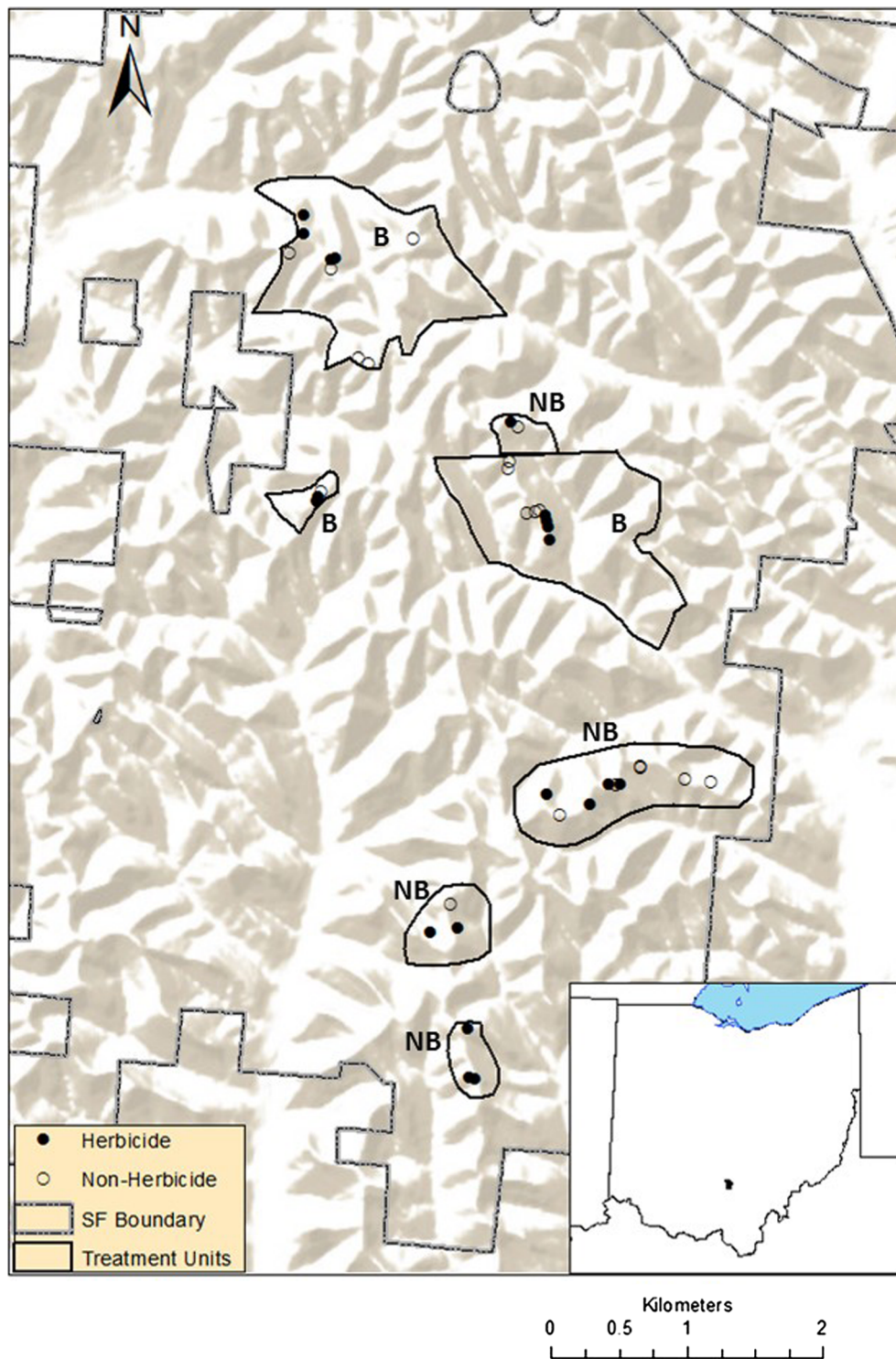


Fig. 1. Distribution herbicide (closed black circle) and no herbicide (open circle) plots within prescribed burn units labeled B and no burn units (NB) within study area at Tar Hollow State Forest in southeastern Ohio.

stems > 10 cm dbh within a plot. Plots (experimental units) were established within three burn units (a total of 269 ha) and four unburn units (a total of 132 ha) (Fig. 1). Because fire lines were shifted for operational reasons, two plots were not burned (one C and one H plot), one control plot (no herbicide) was burned, and an additional control plot was eliminated because of access issues, which resulted in a total of 21 burn plots and 18 non-burn plots. The herbicide treatment was randomly assigned to 10 burned and 10 unburned plots. Imazapyr (6% Arsenal® (BASF Specialty Chemicals, Research Triangle Park, NC);

53.1% active ingredient) was applied via stem injection (hack and squirt) into all *Ailanthus* stems ≥ 3 cm dbh in the plots and in a 10 m buffer surrounding each plot in late summer to early fall 2009 (Kochenderfer et al., 2012). Burn plots (n = 21) were located within three separate prescribed burn units (six plots within a 111-ha unit, ten plots within a 148-ha unit, and three plots within a 10-ha unit). The two larger units were burned simultaneously on April 13, 2010 and the smallest unit was burned on April 5, 2010 by the ODNR Division of Forestry. To estimate relative fire intensity, two Hobo® temperature

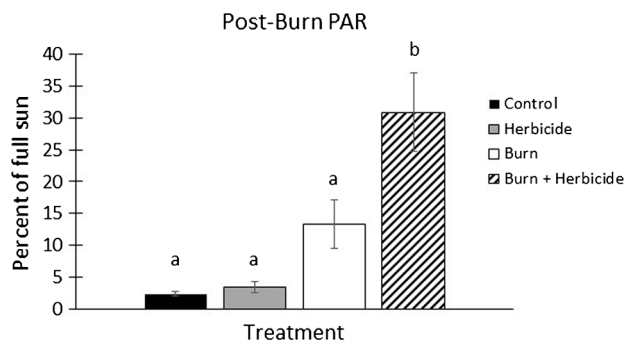


Fig. 2. Percent of full sun in 2011, estimated by measuring photosynthetically active radiation (PAR) within 10-m radius plots, approximately 16 months after a prescribed fire and 23 months following stem-injection herbicide applications. Different letters above the bars indicate a significant difference based on Tukey-Kramer least square means test comparisons between treatments.

Table 2

Weather, sensor temperatures during prescribed burns on April 13, 2010 between 13:00 and 16:00 EST at Tar Hollow State Forest, SE Ohio, USA.

Variable	Mean	Minimum	Maximum
Air temperature, °C	21.8	21.4	22.1
RH, %	27.3	24	28.3
Wind speed, mph	4.33	–	8.5
Days since last rain	6	–	–
Max paint tag temperature, °C	132.7 ± 92.9	< 79.4	343
Mean sensor temperature, °C	155.2 ± 76.7	56.8	337.8
Upslope probe temperature, °C	145.8 ± 75.2	52.2	367.5
Downslope probe temperature, °C	162.7 ± 71.6	61.5	308
Heat index*	16166 ± 3825	755	62,544
Visual estimates of plot area burned, %	95.8 ± 8	75	100

* Heat index = temperature over 50 °C summed over duration (integral under curve).

probes (Onset Computer Corporation, Bourne, MA, USA) and data loggers were deployed at 30 cm above the forest floor at each plot center (1 upslope and 1 downslope) and were programmed to measure and record temperature every 2 s during the burn. As a backup for temperature probe failures, Tempilaq® temperature-sensitive paint (LA-CO Industries, Inc., Elk Grove Village, IL, USA) aluminum tags were also installed at plot centers (Iverson et al., 2004). The probes in all three plots within the smallest burn unit malfunctioned. With the data captured from the logger-probes, maximum probe temperature and heat index, defined as the cumulative temperatures above 50 °C (see Iverson et al., 2004) were derived for each vegetation plot (Table 2). Mean heat index (integral under curve) ranged from 755 to 62,544 and averaged 16,166. Maximum probe temperatures averaged 337.8 °C across all burn units. Mean sensor temperature of all probes was 155.2 °C compared with 132.7 °C for aluminum tags deployed with temperature sensitive paint. All fires spread uniformly across the three burn units. The percent area burned in each plot estimated immediately following the burn averaged 95.8% and ranged from 75 to 100%.

Fire severity classes were also estimated in August 2010 using a modification of Brose and Van Lear (1998), based on a sum of four categorically scored variables: (a) sapling (3–10 cm dbh) top-kill or mortality: 1 = < 50%, 2 = 50–99%, or 3 = 100%; (b) midstory (10.1–25.5 cm dbh) top-kill or mortality: 0 = none, 1 = little to infrequent, or 2 = some to frequent; (c) midstory injury: 1 = none to little, 2 = some to frequent, or 3 = frequent; and (d) bark scorch: 1 = none to little, 2 = some, or 3 = frequent. A fire severity class was assigned based on the cumulative score of each categorical rating: low class scores ranged from 2 to 3, medium class scores ranged from 4 to 6, and high ranged from 7 to 8.

2.2. Vegetation sampling

Within each 10-m radius plot, overstory basal area (BA) by species was estimated prior to treatments (summer 2009) with a 10-BAF prism. In 2009, dbh of *Ailanthus* saplings (3–9.9 cm dbh) and trees (≥ 10 cm dbh) was recorded and the stems were mapped and permanently labeled. From 2009 to 2014 (excluding 2013), all small *Ailanthus* saplings (1.4-m tall to 2.9 cm dbh), large *Ailanthus* saplings, and trees were tallied within each 10-m radius plot. Within four 2 m-wide × 8 m-long belt transects (N, E, S, W of plot center) nested within the 10-m radius plot, new *Ailanthus* germinants (cotyledons still attached or evidence of past presence), small *Ailanthus* seedlings (< 50 cm height), and large *Ailanthus* seedlings (50 – 140 cm height) were also tallied and the four belt transects within each plot were combined. Tally data for all size classes were also collected post-treatments from 2010 to 2014, excluding 2013.

2.3. Mortal status and herbicide effectiveness

Plots were evaluated 12 months after the herbicide treatment and 3–4 months after the prescribed burn treatment to evaluate mortal status and crown vigor of all *Ailanthus* saplings and trees. In addition, herbicide effectiveness in the H plots, and fire severity in the B and H + B plots was also evaluated. Mortal status of *Ailanthus* saplings and trees within all treatment plots in August 2010 was rated using the following classes: 0 = healthy and alive; 1 = main stem dead with no sprouts; 2 = main stem top-killed with basal sprouts; or 3 = main stem alive with basal sprouts. Crown vigor (dieback and foliar discoloration) of trees and saplings within all plots was evaluated as follows: 1 = healthy with < 10% dieback and discoloration; 2 = 10–25% dieback and discoloration; 3 = 26–50% dieback and discoloration; 4 = 51–75% dieback and discoloration; 5 = 76–100% dieback and discoloration; or 6 = dead. Imazapyr-treated *Ailanthus* stems were rated for herbicide effects as class variables: 0 = healthy, no apparent effect; 1 = discolored and small stunted leaves, multiple typical stems with very small leaves; 2 = severely malformed or stunted sprouts or leaves; or 3 = complete kill, no sprouting. These herbicide assessments were repeated in 2012. During the final post-treatment stem tallies in 2014, visible signs of herbicide effects as well as associated root and basal sprouting within the H-only plots were monitored. On each plot in the burn units, stem char on the lower bole (0–25 cm) of each sapling and tree was rated as either absent, light (< 25% lower stem charring), medium (25–75% charring), or heavy (> 75% charring). The presence of basal sprouts (yes or no) and root sprouts (yes or no) was recorded.

2.4. Light sampling

Photosynthetically active radiation (PAR) was measured in each plot with an AccuPAR LP-80 ceptometer (Decagon Devices, Pullman, WA, USA) in August 2011, in the second growing season after treatments. A separate PAR measurement (1 m height) was taken at plot center and at every 2-m along each of the four 2-m wide by 8-m long belt transects. Each measurement was the mean of separate readings taken at 4 cardinal directions (N, E, S, and W). A plot average of PAR was used for analyses. A second ceptometer was placed in a nearby clearing to record PAR in full sunlight so that the PAR measured in each plot could be expressed as the percentage of full sunlight.

2.5. Post-treatment non-*Ailanthus* vegetation composition

Additional vegetation sampling was conducted in 2014 to determine how native tree regeneration responded to the treatments as well as how shrub and herbaceous cover might affect *Ailanthus* regeneration. The post-treatment density of non-*Ailanthus* tree seedling and small sapling (two broad groups: oaks + hickories and all other native

species), and percent cover of shrubs, grasses, forbs, *Rubus* spp., and *Smilax* spp. were quantified within each plot. Tree regeneration (> 50 cm tall to 2.9 cm dbh) density and shrub cover estimates were made within the four 2 m-wide × 8 m-long belt transects previously described to count *Ailanthus* germinants and seedlings. The dominant cover species within each belt transect was identified.

2.6. Statistical analyses

For statistical analyses, all *Ailanthus* stem density data collected on plots (the sampling unit) were converted to stems ha⁻¹ and grouped into separate size classes: (1) germinants (attached cotyledons), (2) small seedlings (< 0.5 m tall), (3) large seedlings (0.5 m height to < 3.0 cm dbh), (4) saplings (3–9.9 cm dbh), and (5) trees (≥ 10 cm dbh). Classes were then created for all stem size classes densities, because of model convergence issues. Germinant and small seedling size classes included: none (0 seedlings ha⁻¹), low (1–2000 seedlings ha⁻¹), medium (2001–7000 seedlings ha⁻¹), and high (> 7000 seedlings ha⁻¹). Large seedling size classes were assigned these density classes: none (0 stems ha⁻¹), low (1–1000 stems ha⁻¹), medium (1001–3000 stems ha⁻¹), and high (> 3000 stems ha⁻¹). Sapling and tree classes were assigned the following density classes: none (0 stems ha⁻¹), low (1–100 stems ha⁻¹), medium (101–500 stems ha⁻¹), and high (> 500 stems ha⁻¹). These categorical density data were analyzed separately for each of the four stem size classes using pretreatment data (2009) and the 4-year post-treatment (2014) data only, although intermediate years of data are presented to illustrate ephemeral responses to treatments.

Using SAS 9.4 GLMMIX (SAS Institute Inc., Cary NC, USA), a repeated measures nested incomplete split design was used for the density classes of the germinant, small and large seedling stem size classes. Burn was the whole plot unit, herbicide the subplot and year was the repeated measures portion of the design. Because there were only two time periods an unstructured covariance structure was used to model the correlation between years. A negative binomial distribution with a log link function was the most appropriate model (Littell et al. 2006) for germinants and seedlings. Residuals from all models were assessed for normality using SAS 9.4 UNIVARIATE procedure by plotting and fixed effects were evaluated for homogeneous variance using Levene's test (Brown and Forsythe, 1974). Because tree and sapling density data was limited, the complicated model structure was not supported for these two size classes, instead only plot data within the burn units were analyzed to test the effect of herbicide and year on tree and sapling density classes by excluding all no burn plots. Tukey-Kramer tests were run on Least Square Mean comparisons for response variables with significant fixed and random interaction effects. The Kenward-Rogers denominator degrees of freedom adjustment was used for all models. Differences were considered significant at $p \leq 0.05$.

To assess the relationship between visual plot level fire severity ratings and probe temperature data collected (maximum probe temperature and heat index) during the prescribed burns, linear regressions were run. We also assessed the relationship between fire severity and the immediate post-burn response of *Ailanthus* saplings and trees to the prescribed burns. Logistic regression models of 2010 mortal status (alive (top-killed) or dead) in response to the heat index (derived from probe temperature data collected during prescribed burns), were run separately for *Ailanthus* trees (≥ 10 cm dbh), large (≥ 3–9.9 cm dbh), and small (1.4-m tall to 2.9 cm dbh) saplings (representing a subset of the large seedling size class) in the burn-only treatment. The impact of burn and herbicide treatments on photosynthetically active radiation (PAR) light levels measured in 2011 was tested using an incomplete split-plot design. Burn was the whole plot unit with herbicide treatments nested within.

3. Results

3.1. Response of *Ailanthus* to herbicide treatments

In 2010, one year after stem-injections of imazapyr, herbicide effectiveness was evaluated on all treated stems (≥ 3 cm dbh; $n = 600$ stems). For trees treated with herbicide only, 72.4% were completely killed. Of the 27.6% of trees that survived, all exhibited herbicide effects: 57.6% had discolored, multiple stems and very small leaves, while the remaining 42.4% live trees had malformed and stunted sprouts and leaves. Not surprisingly, saplings in the herbicide only plots were more susceptible to the effects of herbicide than trees with 85.7% dead, and among the survivors, 53.9% had malformed, stunted sprouts and leaves, and 46.1% had discoloration, multiple stems and very small leaves.

3.2. Fire severity

Of the 21 plots in burned units, 24% were rated as low fire severity, 52% as medium severity, and 24% as high severity. In addition, evidence of burning (fuel consumption, woody seedling top-kill, or scorch) within a 1-m radius immediately surrounding each individual sapling or tree was frequently observed (88–100%), which corroborated visual plot level estimates of % burn (Table 1). The relationship of fire severity and either maximum probe temperatures ($y = 6.507 + 0.140x$, $r^2 = 0.11$), or heat index ($y = 4247.2x - 16812$, $r^2 = 0.352$) were poorly correlated.

3.3. *Ailanthus* response to burning alone and with herbicide

In the first post-treatment growing season (2010) following herbicide and burning, few *Ailanthus* trees and saplings were alive within the H + B plots with 98.8% of trees dead and 1.2% top-killed and sprouting, while 99.4% of saplings were dead and 0.6% top-killed and sprouting (Fig. 3). However, in the B only treatment, no *Ailanthus* tree mortality was observed as 62.9% were top-killed but resprouted. Logistic regression models of *Ailanthus* tree and sapling mortality as a function of heat index in the burn-only plots were not significant. However, the mortality of tagged stems < 3.0 cm dbh (1.4–m tall to 2.9 cm dbh; large seedlings) increased as heat index increased ($p = 0.033$). Among the tagged *Ailanthus* stems, mortality in the B plots was only observed in the three smallest size classes (< 10 cm dbh), as 29.5% of stems were killed and 67% were top-killed and resprouted (Fig. 3). However, mortality of saplings and trees exceeded 98% in the H + B plots (Fig. 3). Crown vigor (76–100% dieback or dead) was poorest in H + B trees and saplings, while 93.4% and 66.1% of the B-only saplings and trees, respectively, displayed these same levels of poor vigor. Almost all saplings and trees in the B plots had light stem char, 92.8 and 95%, respectively, and an additional proportion, ~5%, had heavy stem char. In the H + B plots, 81.4% of saplings and 70.2% of trees had light stem char, while 6.4% of saplings and 8.3% of trees had heavy stem char. As expected, the presence of basal sprouts (64.5% of trees) and root sprouts (66.1% of trees) from *Ailanthus* trees and saplings was greatest in B plots.

Although not statistically tested, *Ailanthus* tree and sapling density in all treatments remained lower than controls in 2010, 2011, and 2012 (Fig. 4). In 2014, tree and sapling densities were lower in H plots than in all other treatments (Fig. 4). However, as resprouts grew into the sapling layer in the B plots, stem densities were not different from controls. Within control plots, *Ailanthus* sapling density declined due to natural mortality and was significantly lower in 2014 than in 2009. The repeated measures models comparing pre-treatment (2009) and post-treatment (2014) detected significant H × Y effect ($p > 0.018$) for tree

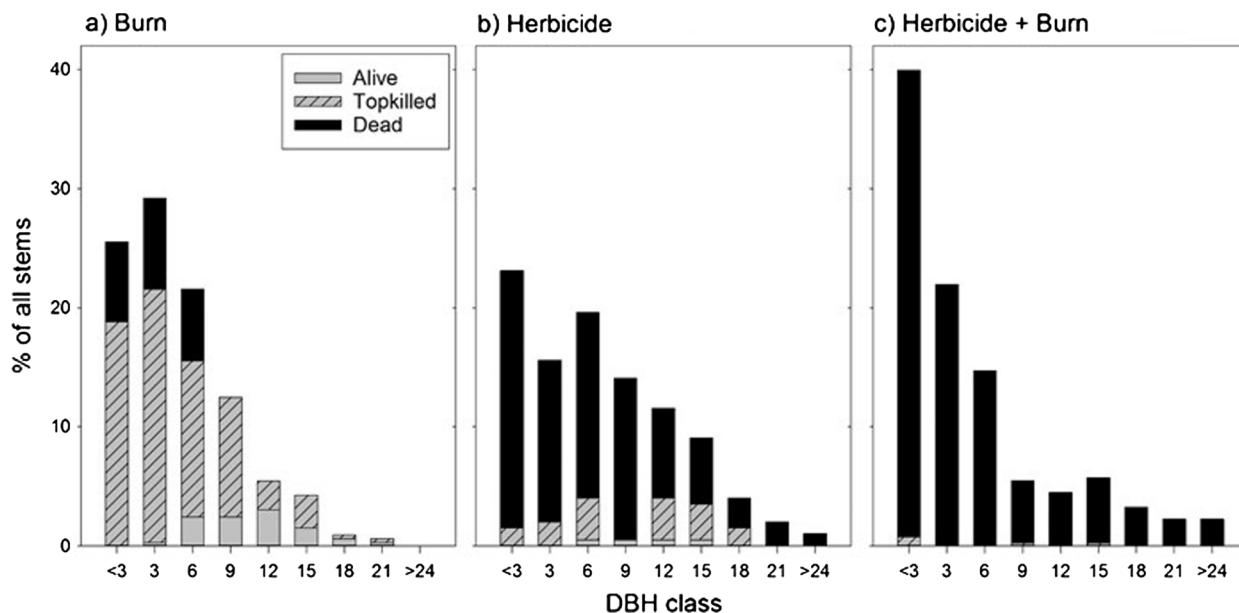


Fig. 3. Mortal status, evaluated four months after the prescribed fire in 2010, of *Ailanthus* saplings and trees represented as percent of stems that were alive, top-killed or dead within burn, herbicide, and burn + herbicide by size classes < 3 cm, 3–5.9, 6–8.9, 9–11.9, 12–14.9, 15–17.9, 18–20.9, 21–23.9, and > 24 cm dbh.

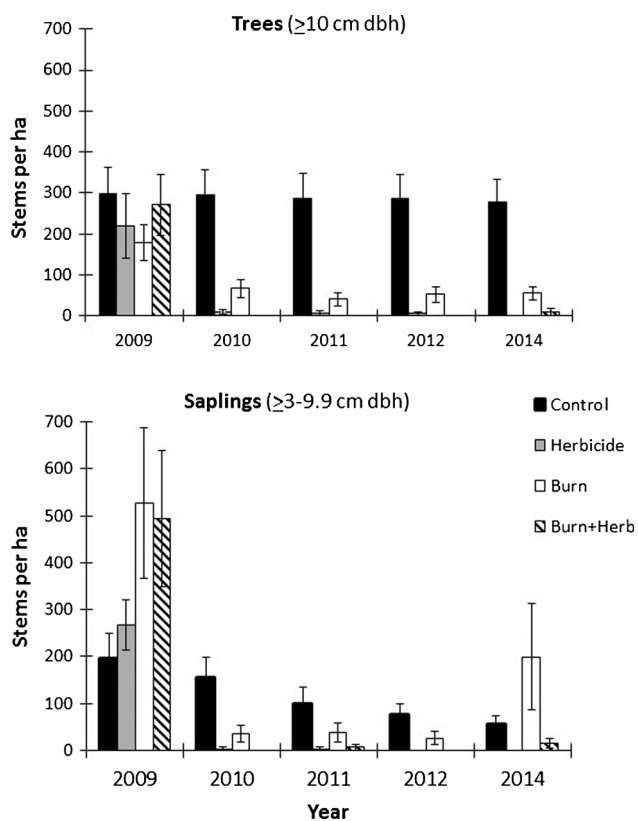


Fig. 4. Stem densities of live *Ailanthus* trees and saplings following herbicide and burn treatments. Live stems are represented by both original stems and stems top-killed but sprouting after treatments. Only pre-treatment (2009) and post-treatment (2014) data were tested in statistical models.

densities (Table 3).

Prior to treatments, *Ailanthus* germinant densities averaged 7644 seedlings ha⁻¹ across all plots (Table 1). However, in the first growing season after herbicide and fire treatments, germinant densities dramatically increased within all treatment plots relative to controls, 5.7 fold in H + B plots, 4.7 fold in B plots and 2.5 fold in H plots

Table 3

Statistical summary testing the effects of treatments on live *Ailanthus* stem density classes testing pretreatment (2009) and posttreatment (2014) for each stem size class*. In fall 2009, trees and saplings were stem-injected with herbicides and in April 2010 prescribed burns were conducted.

Effect	Large Seedlings		Small Seedlings		Germinants	
	df	p-Value	df	p-Value	df	p-Value
Burn (B)	1, 1.9	20.485	1, 4.8	0.666	1, 6.9	0.038
Herbicide (H)	1, 38.9	0.126	1, 46.5	0.038	1, 7.0	0.164
B × H	1, 38.9	0.499	1, 46.5	0.298	1, 7.0	0.043
Year (Y)	1, 7.0	0.008	1, 7.0	< 0.001	1, 7.0	< 0.001
B × Y	1, 7.0	0.900	1, 7.0	0.240	1, 7.0	< 0.001
H × Y	1, 7.0	< 0.001	1, 7.0	0.002	1, 7.0	0.066
B × H × Y	1, 7.0	0.927	1, 7.0	0.917	1, 7.0	0.045
	Trees ^a		Saplings ^a			
Herbicide	1, 19	0.028	1, 19	0.038		
Year	1, 19	< 0.001	1, 19	< 0.001		
H × Y	1, 19	0.018	1, 19	0.087		

* Stem size classes: large seedlings = 0.5–1.4 m tall & < 3.0 cm dbh; small seedlings < 0.5 m tall; trees ≥ 3 cm dbh; saplings = 3–9.9 cm dbh. Each stem size class was analyzed separately with a repeated measures nested split plot design to test the effects of prescribed fire (whole plot unit) and stem-injection herbicide treatments (subplot unit) on *Ailanthus* populations. Year (pre- and post-treatment) was the repeated measures variable. Density classes (none = 0, low = 1–2000, medium = 2001–7000, and high > 7000 stems ha⁻¹) were created for germinant, small and large seedling size classes because of model convergence issues. Separate density classes were created for trees and saplings as follows: none = 0, low = 1–100, medium = 101–500, and high > 500 stems ha⁻¹.

^a Because of the complexity of the repeated measures nested split plot design and the amount of tree and sapling data, only plots within the burn units were used to test for herbicide and year effects. Refer to Materials and Methods 2.6 Statistical Analyses for specifics.

(Fig. 5). However these increases were short-lived. By 2014, germinant densities declined to below pretreatment levels across all treatments with the lowest densities in H + B plots. A significant B × H × Y ($p = 0.045$) was detected for germinant density (Table 3).

The density of small established seedlings (stems < 0.5 m tall) was highly variable within treatments, but did not increase on treated plots,

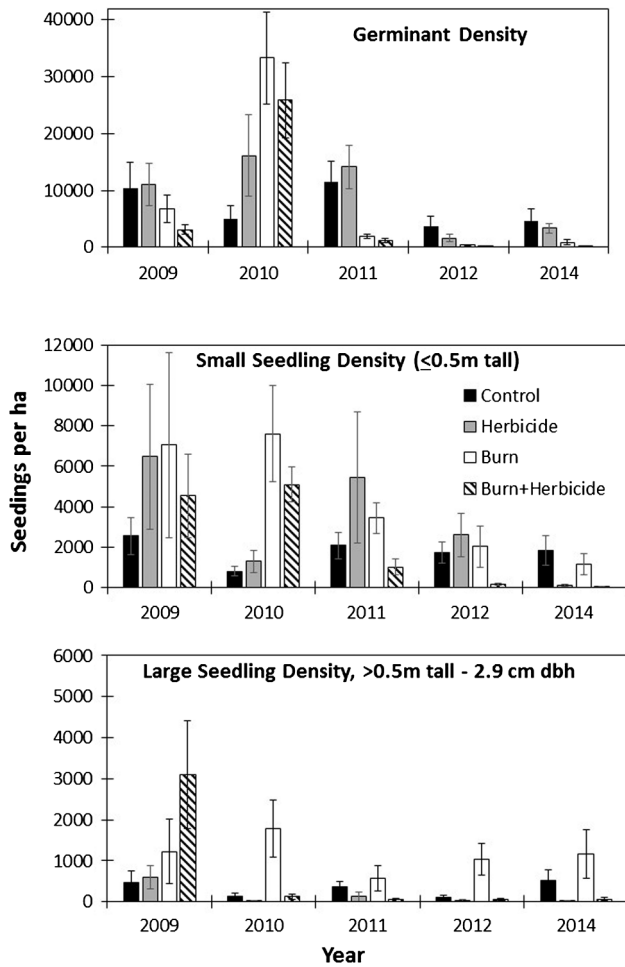


Fig. 5. Annual *Ailanthus* stem densities of germinants, small seedlings, and large seedlings. Only pre-treatment (2009) and post-treatment (2014) data were tested in statistical models. Because no significant Herbicide × Burn × Year effects were detected (Table 3), no mean comparison tests could be shown as in Fig. 3.

despite the large pulse of new germinants in 2010. Although not statistically tested in 2010 through 2012, small seedling densities in B and H + B plots remained unchanged while densities declined 19.8% in H plots immediately following the treatments (2010) (Fig. 5). Large seedling densities in B + H plots plummeted 95% immediately after the treatments and remained unchanged. In 2014, the statistically lowest densities of both small and large seedlings were observed in the H and H + B plots relative to H in 2009 and to control plots in 2009 and 2014. Significant B or B × H effects were not detected for either small or large *Ailanthus* seedlings in statistical models; only the H × Y (2009 and 2014) effect was significant (Table 3).

3.4. Light environment

Prior to the treatments, light levels within individual plots were not measured. However, light levels across Tar Hollow State Forest, averaged 3.6% full sunlight in 2009 (Rebbbeck et al. 2017). In 2011, approximately 18 months after burning, photosynthetically active radiation (PAR) was highest in the burn units (Fig. 2) but varied within treatment plots. The H + B plots averaged $30.9 \pm 6.2\%$ and ranged from 14.7 to 66.8% full sun; while the B-only plots averaged $13.3 \pm 3.8\%$ full sun and ranged from 1.6 to 27.7% full sun. Despite high rates of *Ailanthus* mortality, the herbicide only treatment had little initial impact on light levels; percent full sun only averaged $3.4 \pm 0.9\%$ in H plots compared to $2.3 \pm 0.4\%$ in the non-treated

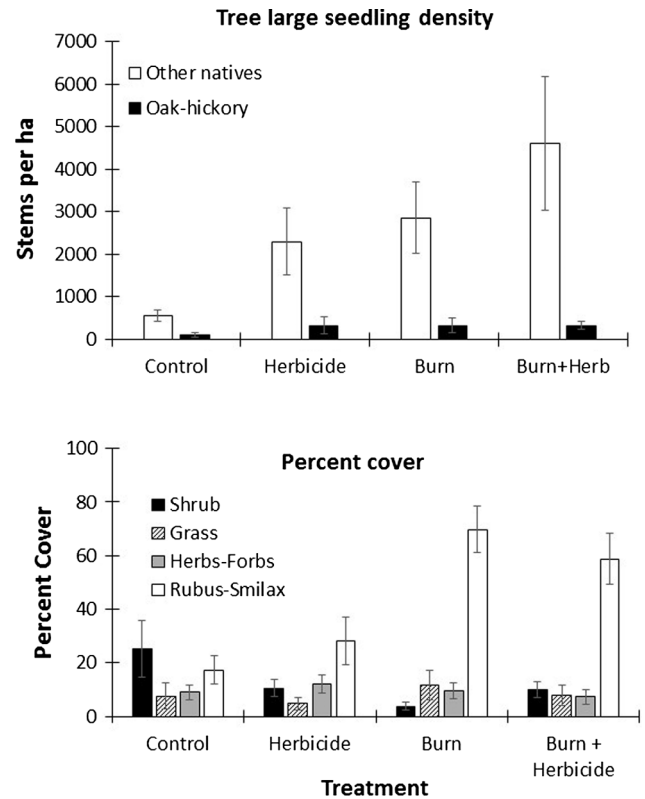


Fig. 6. Summary of non-*Ailanthus* vegetation composition and density four years after prescribed fire and herbicide treatments. Density of large native tree seedlings, represented by stems ≥ 50 cm tall up to 2.9 cm dbh and present cover of shrubs, grasses, herbs-forbs and *Rubus-Smilax* present within the 2-m × 8-m belt transects where *Ailanthus* germinant and seedling counts were done. No pre-treatment data were collected. Tree regeneration densities were higher in all treatment plots relative to control plots.

control plots. Burn ($0 < 0.001$), H ($p = 0.023$) and H × B ($p = 0.042$) had significant effects on percent full sun in the second post-burn growing season (2011).

3.5. Post-treatment non-*Ailanthus* vegetation composition

In August 2014, native tree regeneration and vegetative cover were assessed within all plots. Densities of large native tree seedlings (50 cm height to 2.9 cm dbh) were more abundant on the treatment units, but did not include oak-hickory (Fig. 6). Although no pretreatment data were collected, these 2014 data suggest that the density of large seedlings of native tree species, excluding oak and hickory, increased substantially in all treatments relative to control plots. The observed percent cover of grasses, herb-forbs and shrub (excluding *Rubus-Smilax*) individually suggest little differences among treatment plots. However, due to much greater *Rubus-Smilax* cover on burned plots, total percent cover in B and H + B plots averaged 79% compared with 39% in H and C plots (Fig. 6).

4. Discussion

4.1. Effectiveness of stem-injection herbicide + prescribed fire

On the H + B plots, a stem-injection herbicide treatment followed by fire was effective in killing > 99% of *Ailanthus* trees and saplings. More importantly, this effect was sustained for four years post-burn and the H + B treatment did not stimulate persistent seedling establishment, despite abundant germination and high light levels (31% of full sunlight). *Ailanthus* can be difficult to control with a single herbicide

treatment or when combined with manual cutting; follow-up monitoring and treatment to minimize basal and root sprouting is often required (Burch and Zedaker, 2003; DiTomaso and Kyser, 2007). However implementing a one-time stem injection of imazapyr followed by a prescribed burn was extremely effective and could be very efficient on an operational scale, as long as the *Ailanthus* infestation is not too extensive and fuels are adequate for prescribed fire. Mechanical treatments (single or multiple) prior to prescribed burns or herbicide treatments have been proposed for control of some non-native woody invasives in the northeastern U.S. (Richburg, 2005) and Canada (Meloche and Murphy, 2006), by timing the treatments with a given species' phenology to sufficiently deplete carbohydrate reserves. However, these methods have not proven to be effective on woody shrubs without follow-up treatments (Richburg, 2005). Pile et al. (2017a) demonstrated this with the invasive Chinese tallow tree (*Triadica sebifera*) and found that using early growing season mastication to reduce root carbohydrate reserves and stimulate sprouting followed by growing season prescribed fire and herbicide treatments was effective in reducing its stem densities. Because *Ailanthus* has an extended root system harboring numerous dormant buds, we discourage the use of mechanical treatments alone. This can lead to heavy root sprouting which requires extensive follow-up treatment (Kowarik and Sämel, 2007). It is crucial to combine any cutting with either herbicide or prescribed fire.

4.2. Fire effects

Our current work represents the first multi-year empirical study to characterize the direct effects of prescribed fire on the prolific seed producer and clonal sprouter, *Ailanthus*, a problematic non-native invasive tree in eastern U.S. and European forests (Kowarik and Sämel, 2007; Kasson et al., 2013). Immediately after the burn, we did observe a dramatic increase in germination but not in sprouting. However those increased germinant densities did not persist four years post-burn and thus do not support the predicted increase in *Ailanthus* abundance, based on the Lewis study (2007) as well as the USDA Forest Service Fire Effects Information System (FEIS) which suggests that given the species' prolific sprouting capacity, fire would be likely to promote *Ailanthus* populations (USDA Forest Service, 2018). Lewis (2007) also reported that saplings were easily top-killed by an early spring dormant-season fire but resprouted to preburn density levels after 23 weeks. Unfortunately, as in many fire effects studies (Gucker et al., 2012), vegetation responses were only monitored for one season.

Our objective was to characterize the impacts of "typical" operational dormant season burning on *Ailanthus* populations within mixed oak forests. In Central Appalachia forests, these burns occur in early spring from mid-March through mid-April on public lands (Hutchinson et al., 2005; Bowden, 2009; Iverson et al., 2017). Fall dormant season burns can be more difficult to implement operationally due to more uncertain weather conditions (Brose and Van Lear, 1998) and are typically less effective in controlling target species because below-ground carbohydrate reserves are at their highest levels (Hodgkins, 1958). At the time of the prescribed burns (April 5 and 13, 2010), leaf emergence of all woody species was earlier than unusual. We estimated a cumulative growing degree days (GDD) of 242 during the April 13 prescribed burn, which is 20 days ahead of the 10-year average GDD ((2000–2010) (The Ohio State University Growing Degree Days Calculator, last accessed May 10, 2018, <https://ohiograpeweb.cfaes.ohio-state.edu/weather/gdd-your-area>)). Burning during leaf expansion can be more effective in reducing stem densities of species such as red maple and yellow-poplar with diminished root carbohydrates relative to oaks and hickories (Brose and Van Lear, 1998). However, *Ailanthus* has high-temperature requirements for bud break, and leaf out is later than most other tree species including oaks (Kowarik and Sämel, 2007), suggesting it may be impacted less by spring fires than other species, further exacerbating the challenges of controlling this problematic species. However in our study, *Ailanthus* was beginning to leaf-out during the

prescribed burns which suggests that these stems may have been more susceptible to fire than in more typical springs when *Ailanthus* leaf emergence occurs in early May (Rebbeck, personal obs.). Although we did document high levels of resprouting in the year following the burn, stem densities of saplings and large seedlings were not significantly higher than pre-treatment levels after four years. However, it is possible that a single prescribed fire (with no follow-up treatment) may result in greater dominance of *Ailanthus* as suggested by the increased abundance of seedlings and saplings observed in 2014. This supports anecdotal observations that a single fire may increase *Ailanthus* but the impacts of repeated fires is not known. It is plausible that once a functional frequent fire return interval is re-established, *Ailanthus* populations could be reduced, however this requires further study.

Although not statistically significant, *Ailanthus* tree and sapling densities in burn-only plots had declined by more than 50% four years post-burn. Given that these fires were classified as medium (52% of plots) to high (24% of plots) severity, we had expected more dramatic and sustained increases in *Ailanthus* stem densities over time due to sprouting. Fire severity was not significantly correlated with either maximum probe temperature or heat index (temperature over 50 °C summed every 2 s over duration; Iverson et al., 2004). In addition, no correlation was found between the mortal status of *Ailanthus* stems ≥ 3 cm dbh and heat index, only the mortal status of small stems (< 3 cm dbh) was correlated with heat index. This suggests that fire tolerance may be related to increased bark thickness as stem diameter increases. Although *Ailanthus* has not been studied directly, research in the southern Appalachians of NC suggests that yellow-poplar stems > 2.4 cm dbh are tolerant to prescribed surface fires (Beck, 1990; Gucker, 2007). These findings do corroborate our earlier work which identified that time since the last timber harvest, not recent prescribed fire, as the major driver of *Ailanthus* presence and density (Rebbeck et al., 2017).

Only in the final year of post-burn monitoring (2014), did *Ailanthus* sapling (3–9.9 cm dbh) density begin to "recover" in the burn-only plots, when compared to post-burn densities in 2010 through 2012. It should be noted that this increase in 2014 was highly variable; one plot had a very high sapling density (1146 stems ha^{-1}) compared to the mean of all other B-only plots (96 stems ha^{-1}). It suggests that surviving sprouts (root sprouts and basal sprouts) were relatively slow to reestablish compared to rapid increases typically observed following harvesting disturbances (Call and Nilsen, 2003; Carter and Fredericksen, 2007). This may be the result of the relatively modest increase in understory light (13% of full sunlight) on the burn only plots. Also, had this burn happened during a more "typical" spring, it is possible that more stems would have been more vigorous given *Ailanthus*' delayed phenology. However, it does illustrate the potential uncertainty of how native and non-native species will respond to management activities in future climate conditions because of the initiation of earlier growing seasons. Additionally, the typical fire weather window may also shift.

In the initial year after the prescribed fire, *Ailanthus* germinant density increased dramatically (4.7 times in B-only and 5.7 in B + H), but rapidly fell to pre-treatment levels the following year. In subsequent years of the study, seedlings did not persist suggesting they are short-lived following fire. The pulse of *Ailanthus* germinants was likely due to exposure of mineral soil after the fires consumed the leaf litter and stimulated germination of the seed bank. Rebbeck and Jolliff (2018) report that viable *Ailanthus* seeds can persist for more than 5 years in soil, which suggests that soil seed banks within forest stands occupied by seed-producing female trees could be very substantial. Guthrie et al. (2016) found decreased germination of *Ailanthus* seeds incubated on the soil surface in burned as compared with unburned plots. However, they found germination of *Ailanthus* seeds buried at either 3 or 6 cm soil depth were unaffected by fire, relative to controls. The ephemeral response of *Ailanthus* germinants to fire is similar to the native shade-intolerant species, yellow-poplar, which also displays a pulse of short-lived germinants after fire if the canopy remains mostly closed

(Hutchinson et al., 2005). High seedling mortality of *Ailanthus* under closed canopy conditions has been previously reported (Kowarik, 1995; Kota et al., 2007). Small seedling (< 0.5 m tall) density, represented by a mixture of sprouts and true seedlings, changed little in the initial year following the burn. Within the B-only and H + B plots, light levels averaged 13% and 31% full sun, respectively, which are well above closed canopy mixed oak stands (4% full sun) (Hutchinson et al., 2005) but considerably lower than the ~53% full sunlight found in oak shelterwood stands with 50% residual basal area (Hutchinson et al., 2016). At that same time, the cover of other vegetation increased, capturing the newly available light resource. Similar to the shade-intolerant yellow-poplar, these newly established *Ailanthus* seedlings with limited root systems likely could not persist and compete for light with faster growing stems of sprout-origin which were able to utilize larger root reserves and overtop slower growing seedlings. Although we did not measure percent cover of tree seedlings, it was likely high, given the abundance of large seedlings on treated plots, it is plausible that on sites with less abundant understory vegetation, *Ailanthus* germinants and seedlings could have persisted and become established. In addition, percent cover of *Rubus-Smilax* was much higher in the B and H + B plots relative to the control and H plots, suggesting these species were stimulated by the burn treatments. Clearly, longer-term studies are needed to test effectiveness of multiple burns on *Ailanthus* sprouting as well as other woody competing vegetation such as native *Rubus* spp., the non-native wineberry (*R. phoenicolasius*), and *Smilax* spp., which were abundant in the B and B + H plots. A similar post-burn response has been observed for the non-native invasive princess tree, *Paulownia tomentosa*. Kuppinger et al. (2010) reported that *Paulownia* seedling establishment was widespread and abundant after wildfire in the southern Appalachians, but these new seedlings were unable to compete over time. Six years post-fire, *Paulownia* was only able to persist on xeric and exposed topographic positions where residual vegetation cover was limited.

4.3. Herbicide effectiveness

Our findings provide natural resource managers with an effective herbicide prescription to kill *Ailanthus* stems (3–34 cm dbh) by utilizing a late summer to early fall hack-n-squirt application of 1.9% imazapyr (6% aqueous solution of Arsenal®, 53% a.i.) (Kochenderfer et al., 2012). This treatment was extremely effective in killing *Ailanthus* trees and saplings and showed minimal sprouting observed within treatment areas (plots and buffer areas) during a mortal status survey done five years after herbicide treatments. Since imazapyr has residual soil activity and translocation through root grafting has been reported (DiTomaso and Kyser, 2007; Lewis and McCarthy, 2008), treatment areas were monitored for two years with no observable symptoms on non-target vegetation within 5 m of treated *Ailanthus* stems. DiTomaso and Kyser (2007) found that late growing season timing of the herbicide application was critical to eliminate *Ailanthus* root and basal sprouting as the chemical is translocated belowground along with the seasonal movement of carbohydrates and nutrients.

4.4. Survival of untreated *Ailanthus*

We had an opportunity to monitor survival of *Ailanthus* stems in the untreated controls over the course of the five-year study, and it was not surprising given its clonal nature, that rates of mortality varied with stem size. By 2014, *Ailanthus* tree (≥ 10 cm dbh) density within the untreated control plots had declined slightly, by 6%, while sapling (3–9.9 cm dbh) and large seedling (> 1.4 m tall to 3 cm dbh) densities declined substantially, by 67% and 71%, respectively. This suggests that once *Ailanthus* stems reach dominant or codominant canopy status, survival improves greatly, and that smaller stems are ephemeral but serve as a dynamic vegetative bank ready to exploit resources created by either small (single tree canopy gaps) or large (timber harvesting)

disturbances (Knapp and Canham, 2000; Carter and Fredericksen, 2007; Rebeck et al., 2017). However, it is likely that other factors may have an impact on *Ailanthus* populations.

Although *Ailanthus* has limited susceptibility to insect and disease pests in China, North America and Europe (Ding et al. 2006), it can succumb to *Ailanthus* webworm, *Atteva punctella*, an insect native to Central America and southern Florida. This insect is widely distributed in eastern, central and southern USA (Powell et al., 1973). Its larvae feed on *Ailanthus* foliage and stems which can cause serious damage to seedlings, suckers, and saplings (Ding et al., 2006). We did observe webworm feeding on these smaller stem sizes in this current study as well as at other sites throughout Ohio. In addition, we often see significant dieback on suppressed *Ailanthus* stems in shady understories.

4.5. Native tree regeneration

We observed much higher densities of large (> 50 cm tall) native tree seedlings across all treatments relative to controls in 2014. Species included both shade-tolerant and shade-intolerant species such as red and sugar maple, ash (*Fraxinus americana*), yellow-poplar, sassafras (*Sassafras albidum*), paw-paw (*Asimina triloba*), black locust (*Nyssa sylvatica*), and Eastern red bud (*Cercis canadensis*). However, large oak and hickory seedlings occurred at low densities across all treatments. Research has shown that the development of abundant large oak regeneration in response to fire is dependent on either multiple fires or growing-season fire combined with gap formation or partial harvest on sites that have adequate densities of small oak seedlings to start with (Brose and Van Lear 1998, Iverson et al. 2017). Future long-term studies are needed to determine if multiple burns can effectively reduce sprouting of both undesirable native and non-native vegetation while simultaneously promoting desirable regeneration such as oak and hickory.

5. Conclusions and management implications

This study has contributed to our understanding of the direct impacts of fire on *Ailanthus* demography and the limited role fire plays in the expansion of this problematic invasive tree within mixed oak forests of the Central Appalachian region. We documented that one early spring prescribed fire did not significantly increase *Ailanthus* seedling and sprout densities. After four years, sapling densities were beginning to recover on the burn only treatment, but were still below pre-treatment densities, suggesting the need for continued monitoring to determine if a follow-up burn is necessary.

We have demonstrated for the first time that a stem-injection herbicide treatment with imazapyr followed by single moderately intense fire was extremely effective in killing > 99% of *Ailanthus* trees up to 34 cm dbh. This is very significant given *Ailanthus*'s high propensity for suckering and root sprouting. That being said, the herbicide only treatment was also quite effective, killing 83% of *Ailanthus* stems up to 26 cm dbh. We also demonstrated that *Ailanthus* germinants and seedlings did not persist four years after all active treatments, despite increased canopy openness on the burned and herbicide + burned plots. This result may be due to intense competition from sprouts of faster-growing woody species, including native tree species and *Rubus* spp. and *Smilax* spp. Further research on the long term effects of multiple fires, both dormant and early-growing season, combined with major disturbances such as timber harvesting on *Ailanthus* and other woody native and non-native species, are still needed.

Based on the findings presented, we propose that natural resource managers incorporate late summer to early fall stem-injection imazapyr treatments of *Ailanthus* stems (≥ 3 cm dbh) prior to either a prescribed fire or timber harvest to control future expansion of this problematic invasive tree.

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References

- Beck, Donald E., 1990. Liriodendron tulipifera L. yellow poplar. In: Burns, Russell M., Honkala, Barbara H. (Eds.), *Silvic of North America*. Volume 2. Hardwoods. Agric. Handb. 654. U.S. Department of Agriculture, Forest Service, Washington, DC, pp. 406–416.
- Bowden, M., 2009. Building a state prescribed fire program: experiences and lessons learned in Ohio. Gen. Tech. Rep. NRS-P-46 In: Hutchinson, Todd F. (Ed.), *Proceedings of the 3rd Fire in Eastern Oak Forests Conference*; 2008 May 20–22; Carbondale, IL. U.S. Department of Agriculture, Forest Service, Northern Research Station, Newtown Square, PA, pp. 118–125.
- Braun, L.E., 1950. *Deciduous Forest of Eastern North America*. The Blackburn Press, Caldwell, NJ 596p.
- Brooks, M.L., D'Antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., Di Tomaso, J.M., Hobbs, Richard J., Pellant, M., Pyke, D., 2004. Effects of invasive alien plants on fire regimes. *BioScience* 54, 677–688.
- Brose, P.H., Van Lear, D.H., 1998. Responses of hardwood advance regeneration to seasonal prescribed fires in oak-dominated shelterwood stands. *Can. J. For. Res.* 28, 331–339.
- Brown, M.B., Forsythe, A.B., 1974. Robust tests for the equality of variances. *J. Am. Stat. Assoc.* 69, 364–367.
- Burch, P.L., Zedaker, S.M., 2003. Removing the invasive tree *Ailanthus altissima* and restoring natural cover. *J. Arboricult.* 29, 18–24.
- Call, L.J., Nilsen, E.T., 2003. Analysis of spatial patterns and spatial association between the invasive tree-of-heaven (*Ailanthus altissima*) and the native black locust (*Robinia pseudoacacia*). *Am. Midl. Nat.* 150, 1–14.
- Carter, W.K., Fredericksen, T.S., 2007. Tree seedling and sapling density and deer browsing incidence on recently logged and mature non-industrial private forestlands in Virginia, USA. *For. Ecol. Manage.* 242, 671–677.
- Dibble, A.C., Zouhar, K., Smith, J.K., 2008. Fire and nonnative invasive plants in the northeast bioregion. Gen. Tech. Rep. RMRS-GTR-42-vol. 6 In: Zouhar, K., Smith, J.K., Sutherland, S., Brooks, M. (Eds.), *Wildland Fire in Ecosystems: Fire and Nonnative Invasive Plants*. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT, pp. 61–89.
- Ding, J., Wu, Y., Zheng, H., Fu, W., Reardon, R., Liu, M., 2006. Assessing potential biological control of the invasive plant, tree-of-heaven, *Ailanthus altissima*. *Biocontrol Sci. Technol.* 16 (6), 547–566. <https://doi.org/10.1080/09583150500531909>.
- DiTomaso, J.M., Kyser, G.B., 2007. Control of *Ailanthus altissima* using stem herbicide application techniques. *Arboricult. Urban Forest.* 33, 55–63.
- Gucker, Corey L., 2007. Research Project Summary: Early postfire effects of a prescribed fire in the southern Appalachians of North Carolina. Fire Effects Information System. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer) (accessed 2018.12.04). www.fs.fed.us/database/feis/research_project_summaries/Elliott99/all.html.
- Gucker, C.L., Zouhar, K., Smith, J., Kapler, Stone, K.R., 2012. Characteristics of information available on fire and invasive plants in the eastern United States. *Fire Ecol.* 8, 57–81.
- Guthrie, S.G., Crandall, R.M., Knight, T.M., 2016. Fire indirectly benefits fitness in two invasive species. *Biol. Invasions* 18, 1265–1273.
- Hodgkins, E.J., 1958. Effects of fire on undergrowth vegetation in upland southern pine forests. *Ecology* 39, 36–46.
- Hu, S.Y., 1979. *Ailanthus*. *Arnoldia* 39, 29–50.
- Hutchinson, Todd F., Rebbeck, Joanne, Stout, Susan L., 2016. The devil is in the small dense saplings: a midstory herbicide treatment has limited effects on short-term regeneration outcomes in oak shelterwood stands. *For. Ecol. Manage.* 372, 189–198. <https://doi.org/10.1016/j.foreco.2016.04.016>.
- Hutchinson, T.F., Rebbeck, J., Long, R.P., 2004. Abundant establishment of *Ailanthus altissima* (Tree-of-Heaven) after restoration treatments in an upland oak forest. Gen. Tech. Rep. NE-316 In: Yaussy, D.A., Hix, D.M., Long, R.P., Goebel, P.C. (Eds.), *Proceedings, 14th Central Hardwood Forest Conference*. USDA Forest Service, Northeastern Research Station, Newtown Square, OH, pp. 514.
- Hutchinson, T.F., Sutherland, E.K., Yaussy, D.A., 2005. Effects of repeated prescribed fires on the structure, composition, and regeneration of mixed-oak forests in Ohio. *For. Ecol. Manage.* 218, 210–228.
- Iverson, L.R., Yaussy, D.A., Rebbeck, J., Hutchinson, T.F., Long, R.P., Prasad, A.M., 2004. A comparison of thermocouples and temperature paints to monitor spatial and temporal characteristics of landscape-scale prescribed fires. *Int. J. Wildl. Fire* 13, 311–322.
- Iverson, Louis R., Hutchinson, Todd F., Peters, Matthew P., Yaussy, Daniel A., 2017. Long-term response of oak-hickory regeneration to partial harvest and repeated fires: influence of light and moisture. *Ecosphere* 8 (1), e01642. <https://doi.org/10.1002/ecs2.1642>.
- Kasson, M.T., Davis, M.D., Davis, D.D., 2013. The invasive *Ailanthus altissima* in Pennsylvania: a case study elucidating species introduction, migration, invasion, and growth patterns in the Northeastern US. *Northeastern Monogr.* 20, 1–60.
- Knapp, L.B., Canham, C.D., 2000. Invasion of an old growth forest in New York by *Ailanthus altissima*: sapling growth and recruitment in canopy gaps. *J. Torrey Bot. Soc.* 127, 307–315.
- Kochenderfer, J.D., Kochenderfer, J.N., Miller, G.W., 2012. Manual herbicide application methods for managing vegetation in Appalachian hardwood forests. Gen. Tech. Rep. NRS 96. U.S. Department of Agriculture Forest Service, Northern Research Station, Newtown Square, PA, pp. 59p.
- Kota, N.L., Landenberger, R.E., McGraw, J.B., 2007. Germination and early growth of *Ailanthus* and tulip poplar in three levels of forest disturbance. *Biol. Invasions* 9, 197–211.
- Kowarik, I., 1995. Clonal growth in *Ailanthus altissima* on a natural site in West Virginia. *J. Veg. Sci.* 6, 853–856.
- Kowarik, I., Säuml, I., 2007. Biological flora of Central Europe: *Ailanthus altissima* (Mill.) Swingle. *Perspect. Plant Ecol. Evol. Syst.* 8, 207–237.
- Kuppinger, D.M., Jenkins, M.A., White, P.S., 2010. Predicting the post-fire establishment and persistence of an invasive tree species across a complex landscape. *Biol. Invasions* 12, 3473–3484.
- Lewis, K.C., 2007. Control techniques and management implications for the invasive *Ailanthus altissima* (tree-of-heaven). Ohio University, Athens, Ohio Thesis.
- Lewis, K., McCarthy, B., 2008. Nontarget tree mortality after tree-of-heaven (*Ailanthus altissima*) injection with imazapyr. *North. J. Appl. For.* 25, 66–72.
- Littell, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., Schabenberger, O., 2006. *SAS 1 for Mixed Models*, second ed. SAS Institute Inc., Cary, NC.
- Maringer, J., Wohlgemuth, T., Neff, C., Pezzatti, G.B., Conedera, M., 2012. Post-fire spread of alien plant species in a mixed broad-leaved forest of the Insubric region. *Flora* 207, 19–29.
- McAvoy, T.J., Snyder, A.L., Johnson, N., Salom, S.M., Kok, L.T., 2012. Road survey of the invasive tree-of-heaven (*Ailanthus altissima*) in Virginia. *Invasive Plant Sci. Manage.* 5, 506–512.
- Meloche, C., Murphy, S.D., 2006. Managing tree-of-heaven (*Ailanthus altissima*) in parks and protected areas: a case study of Rondeau Provincial Park (Ontario, Canada). *Environ. Manage.* 37, 764–772.
- Pile, L.S., Wang, G.G., Waldrop, T.A., Walker, J.L., Bridges, W.C., Layton, P.A., 2017a. Managing an established tree invader: developing control methods for Chinese tallow (*Triadica sebifera*) in maritime forests. *J. Forest.* 115, 522–529. <https://doi.org/10.5849/JOF-2016-022R2>.
- Pile, L.S., Wang, G.G., Knapp, B.O., Walker, J.L., Stambaugh, M.C., 2017b. Chinese tallow (*Triadica sebifera*) invasion in maritime forests: the role of anthropogenic disturbance and its management implication. *For. Ecol. Manage.* 398, 10–24.
- Pomp, J.A., 2008. Disturbances, Prescribed Fire, and Invasion by Exotic Plants in a Xeric Mixed-oak and Oak-pine Dominated Area of the Ridge and Valley in Eastern West Virginia. West Virginia University, Morgantown, WV 173 p. Thesis.
- Powell, J.A., Comestock, J.A., Harbison, C.F., 1973. Biology geographical distribution and status of *Atteva exquisita* (Lepidoptera: Yponomeutidae). *Trans. San Diego Soc. Natural Hist.* 17, 175–186.
- Rebbeck, J., Jolliff, J., 2018. How long do seeds of the invasive tree, *Ailanthus altissima* remain viable? *For. Ecol. Manage.* 429, 175–179. <https://doi.org/10.1016/j.foreco.2018.07.001>.
- Rebbeck, J., Hutchinson, T., Iverson, L., Peters, M., Yaussy, D., Fox, T., 2017. Distribution and demographics of *Ailanthus altissima* in an oak forest landscape managed with timber harvesting and prescribed fire. *For. Ecol. Manage.* 401, 233–241. <https://doi.org/10.1016/j.foreco.2017.06.050>.
- Richburg, J., 2005. Timing treatments to the phenology of root carbohydrate reserves to control woody invasive plants. University of Massachusetts Amherst, Amherst, MA pp. 162, Ph.D. Dissertation.
- Webster, C.R., Jenkins, M.A., Jose, S., 2006. Woody invaders and the challenges they pose to forest ecosystems in the eastern United States. *J. Forest.* 104, 366–374.
- Zouhar, K., Smith, J.K., Sutherland, S., 2008. Effects of fire on nonnative invasive plants and invasibility of wildland ecosystems. Gen. Tech. Rep. RMRS-GTR-42-vol. 6 In: Zouhar, K., Smith, J.K., Sutherland, S., Brooks, M. (Eds.), *Wildland Fire in Ecosystems: Fire and Nonnative Invasive Plants*. USDA Forest Service, Rocky Mountain Research Station, Ogden, UT, pp. 7–32.