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RESEARCH ARTICLE

Basal bark herbicide treatment of *Lonicera maackii* (Amur honeysuckle) is effective regardless of application timing, with limited nontarget effects on native plant diversity

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Abstract

1. Managers tasked with controlling invasive species require effective methods that are quick and easy to use without inflicting extensive nontarget damage, while also being compatible with other scheduled management responsibilities. *Lonicera maackii* (Amur honeysuckle) is a non-native shrub that has invaded eastern and midwestern North American deciduous forests, altering ecosystem function and reducing biodiversity.

2. This study explores prescribed fire and seasonal basal applications of triclopyr ester as control methods and examines the extent of nontarget damage. We used paired-split plots to implement basal bark treatments in different seasons within burned and unburned units, and we tracked individual *L. maackii* to determine mortality and hyperlocal impacts of management.

3. Basal bark treatments killed 98.4% of *L. maackii* across seasonal timings. Nontarget plant cover declined similarly for all herbicide application seasons, but there were some signs of recovery within 4 years, and the early- and late spring treatments were less affected overall. Species richness showed biologically small but statistically different declines across treatment times. Prescribed fire did not impact *L. maackii* mortality or interact with herbicide efficacy.

4. Basal bark applications of triclopyr are an effective means of *L. maackii* control regardless of application timing, which allows managers to implement it at their convenience to avoid interfering with other management tasks that have time constraints.

KEYWORDS
habitat management, invasive species, land management, triclopyr

1 | INTRODUCTION

Invasive plant species are widespread, impact the dynamics of the ecosystems they invade (Roy et al., 2023; Vitousek et al., 1997) and are the second leading threat to biodiversity next to habitat loss (Wilcove et al., 1998). Invasive plants have a plethora of direct and indirect impacts that alter ecosystem functioning (Fargen et al., 2015; Hopfensperger et al., 2017), community interactions (Dutra et al., 2011; Schmidt & Whelan, 1999), community structure and species composition (Christopher & Cameron, 2012; Hartman

& McCarthy, 2008). Depending on the success of the invader, local biodiversity can be reduced to a near monoculture (Olson & Whitson, 2002), underscoring the importance of effective control strategies.

Land managers tasked with extirpating invasive species face many challenges and spend billions of US dollars per year (Kimball et al., 2015), with an estimated US$46 billion spent on management from 1960 to 2020 (Fantle-Lepczyk et al., 2022), and financial and time constraints mean that managers are frequently unable to control all invasive species in a location. Managers may ignore heavily invaded areas to focus actions on smaller invasive populations located in higher quality habitats where local extirpation of the invasive plants is feasible and less expensive (Welch et al., 2014). Time constraints also include other management activities, such as the application of prescribed fire, grazing management or seed collection and planting. When these other activities must occur at a particular time of year, it limits the time available for invasive species control. Identifying temporal compatibility between invasive management and other responsibilities can allow managers to optimize schedules and improve overall management efficacy. For example, if invasive control methods can take place during winter, when managers' time is more flexible and vegetation is dormant, but still be effective, this would benefit overall management.

1.1 | *Lonicera maackii* impacts

Lonicera maackii (Rupr.) Herder, an invasive species to North America, is an upright deciduous shrub (up to 6 m tall), often with many arching branches growing from a base or meristematic burl (Czarapata, 2005; Luken, 1988). It produces leaves early in the spring, usually 1–2 weeks before native species leaf-out and retains them late into the fall (Czarapata, 2005; Luken & Thieret, 1996). *Lonicera maackii* becomes reproductively mature at 3–8 years of age (Deering & Vankat, 1999) and produces prodigious amounts of fruits whose seeds may remain viable in the soil for 2–3 years (Czarapata, 2005). It flourishes in sunny, upland, disturbed habitats such as along forest edges (Luken et al., 1995; Luken & Mattimiro, 1991) but can still tolerate and dominate more shady and mesic conditions (Czarapata, 2005; Swearingen et al., 2010). Combined with other invasive traits, including a long dispersal range (Gorchov et al., 2014; Gosper et al., 2005; Nyboer, 2007), multiple seed dispersal methods (Castellano & Gorchov, 2013; Gosper et al., 2005), allelopathy (Cipollini et al., 2008; Cipollini & Dorning, 2008), and the ability to readily resprout from roots after above-ground tissues have been killed or removed (Luken et al., 1991; Luken & Mattimiro, 1991), it has few natural controls like herbivory in its introduced North American range (Lieurance & Cipollini, 2012).

Lonicera maackii is pervasive across eastern and midwestern U.S. deciduous forests and is present in 34 states as of 2013 (CABI, 2018), significantly altering invaded woodland communities. It reduces light penetration below its dense canopy and limits soil nutrient availability (Luken & Thieret, 1996), which can reduce native plant richness (Collier et al., 2002; Gould & Gorchov, 2000), diminish basal and radial growth of saplings (Hartman & McCarthy, 2008) and increase native tree seedling mortality (Gorchov & Trisel, 2003). Its negative impacts also extend to consumers like birds (Ingold & Craycraft, 1983; Meiners, 2007; Schmidt & Whelan, 1999; Witmer, 1996). Because of these ecological disruptions, the removal of *L. maackii* is often a high priority for woodland managers when the maintenance of native biodiversity is a goal.

1.2 | Common eradication methods

Like many invasive woody plants, control methods of *L. maackii* include prescribed burning, hand or machine pulling, cutting alone, cutting followed by an herbicide treatment, foliar-applied herbicide treatments, herbicide injections and basal bark herbicide treatments. These treatments vary in both work and time requirements and in efficacy (Hartman, 2005, reviewed in Baker, 2019). For example, cutting followed by herbicide application (i.e. cut-and-treat) is a common and effective eradication method (Olson & Whitson, 2002; Reinartz, 1997) where plants are cut near the base and herbicide is applied to the remaining stump. Although this can avoid soil disruptions that accompany mechanical removal (Gayek, 2000), as with other cutting methods, the manual removal of cut stems is time and labour-intensive. Additionally, many treatments need to be repeated for multiple consecutive years to eliminate viable *L. maackii* seeds from the seed bank (Czarapata, 2005; Luken & Mattimiro, 1991), making the cost, effort and time to remove them critical factors in control efforts.

Prescribed fire has been used to control invasive species including *L. maackii* in fire-adapted communities, but its effects are not well researched. Fire has produced limited mortality on bush honeysuckle species for spring, summer and fall burns (Zouhar et al., 2008). Although fire can completely kill younger plants, it may only top-kill adults, killing the above-ground stems but not the roots (Czarapata, 2005). Adults vigorously resprout from intact roots, so repeated annual burning may be required to drain root resource reserves (Batcher & Stiles, 2000; Czarapata, 2005). Few studies have rigorously investigated the effects of prescribed fire on *L. maackii* after the initial post-fire season (Zouhar et al., 2008) or interactions between fire and other control methods.

Herbicide-focused methods can save both labour and time, particularly when vegetation removal is not necessary. Basal bark spray applications around the circumference of a plant's base and stems are similar to herbicide injectors in ease of use and may be faster, but they are more expensive and may have significant nontarget impacts (Holmes & Berry, 2009). Previous research shows basal bark treatments can be highly effective for *L. maackii* (Kleiman et al., 2018), and there may be additional benefits for achieving restoration goals. When native plants are under high herbivore pressure, using a herbicide treatment that leaves the *L. maackii* stems intact has increased native vegetative layer success by functionally fencing out the

herbivores (Cipollini et al., 2009). Also, unlike pulling, the remaining roots may prevent soil erosion until the vegetative layer can be reestablished (Luken et al., 1997).

In this study, we examine the efficacy and impacts of a combination of two control methods for *L. maackii*, prescribed fire and basal bark herbicide application. Managers may use combinations of invasive plant control techniques depending on available resources, community type, extent of invasion and goals for that site. Prescribed fire is often implemented to mimic historical disturbance regimes and benefit natural communities, but its concurrent use with cutting methods or with herbicide application methods for *L. maackii* is unstudied. Importantly, we also manipulate herbicide application timing to determine if this activity can be implemented at different times of the year without losing efficacy or increasing nontarget effects. Specifically, the purposes of this study are to (1) determine the optimal timing of triclopyr basal bark application to maximize *L. maackii* mortality, (2) determine whether basal bark application efficacy varies with prescribed fire and (3) measure potential negative nontarget effects on native and non-native understory plants.

2 | MATERIALS AND METHODS

2.1 | Study sites and design

We measured the effects of oil-based triclopyr basal bark treatments and prescribed fire on *L. maackii* across three locations in northern Illinois, USA: Franklin Creek State Natural Area (hereafter 'Franklin Creek'), Nachusa Grasslands and an adjacent private residence (Figure 1). Franklin Creek is a 400-hectare preserve located near Franklin Grove, Illinois, owned by the Illinois Department of Natural Resources. It consists largely of dry-mesic upland forest, woodlands and savannas with extensive mesophytic tree encroachment. Connected on the north side is Nachusa Grasslands, a 1200-ha preserve owned by The Nature Conservancy. It is mostly prairie but also has approximately 120 ha of upland woodlands and savanna. The private residence directly adjacent to Nachusa Grasslands also consists of upland woods. All work was permitted by and carried out under the approval of The Nature Conservancy and the Illinois Department of Natural Resources.

All areas have controlled *L. maackii* to some extent in the past. There have been mesophytic tree thinning efforts at Nachusa Grasslands and Franklin Creek, and both sites receive prescribed fire. Two plots were established at Franklin Creek, two plots at Nachusa Grasslands, and one plot straddled the border between Nachusa Grasslands and the adjacent private residence. Those five plots were split into burned and unburned subplots with 80 *L. maackii* individuals in each subplot. Individuals ranged from less than 1 m tall to more than 3 m tall. We randomly assigned each individual one of five herbicide treatments where herbicide was applied while *L. maackii* was entering dormancy, dormant, in late dormancy, and in early growing seasons or a control treatment that received no herbicide. A tree tag was attached to each individual and marked with tree paint for
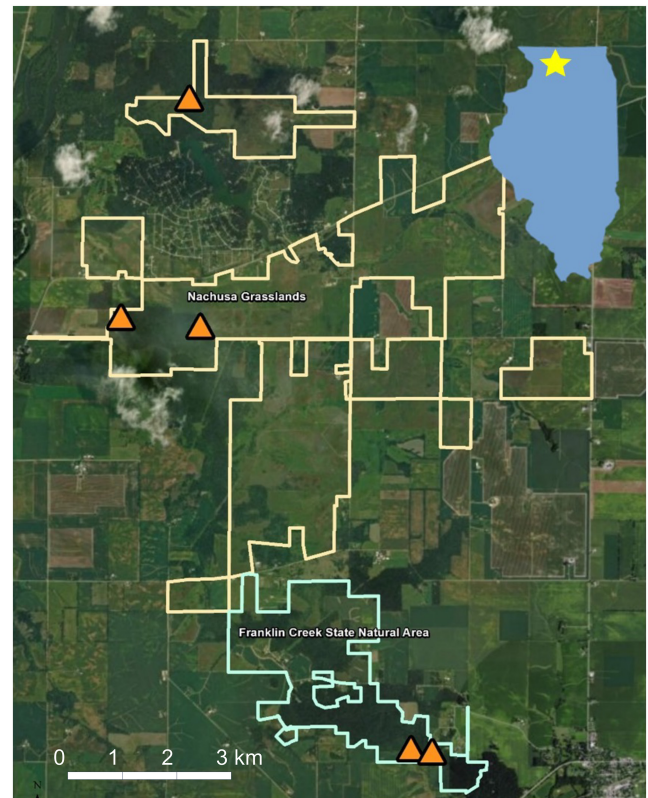

FIGURE 1 Map of *Lonicera maackii* Treatment Sites. Yellow outlined areas indicate Nachusa Grasslands, and blue outlined areas indicate Franklin Creek State Natural Area. Orange triangles indicate study site locations. Note: The northernmost subplot is on adjacent private land owner's property. Inset map indicates location within Illinois.

easy relocation. The experiment is a paired-split plot design, with 10 subplots ($n=5$ per fire treatment, paired) and herbicide application timing was replicated within each subplot ($n=16$ per treatment per subplot), for a total of 800 plants.

2.2 | Treatments and data collection

We applied a 10% triclopyr ester herbicide solution in oil (16.67% v/v Garlon 4 Ultra, Dow AgroSciences LLC, Indianapolis, IN) sprayed entirely around the circumference of each *L. maackii* stem using a backpack sprayer (Iris 15, Birchmeier, London, KY). The backpack cone spray tip was set to produce neither a stream nor a fine mist using low pressure. Herbicide was applied in a 15–30 cm band from ground level until all stems were fully coated, but the herbicide did not run off, as directed by the manufacturer. The herbicide application for entering dormancy application (hereafter 'fall') took place from November 26 to December 16, 2017. The dormant application (hereafter 'winter') was performed on January 28, 2018; late dormancy application (hereafter 'early spring') on March 10–11; and the early growing season (hereafter 'late spring') application on May 4–May 5 after prescribed fire was completed. The fall application had a wide range because hunting season limited accessibility, but

all individuals had some green leaves when treated. All applications across seasons were applied without snow cover, eliminating any potential water dilution impacts.

Paired burned and unburned plots were established based on prescribed fire activities between March 16 and April 22, 2018. Due to regular use of prescribed fire in all these sites, some unburned plots received fire in subsequent years. We tested for interactions and potential main effects of the 2018 fire status because we hypothesized that plants subjected to fire in the same year as the herbicide treatment may respond differently from unburned areas. We checked if the fire reached the base of all individuals within burned subplots when applying the early growing season herbicide treatment.

We assessed the health status of treated L. *maackii* plants in mid-May 2018 (1 week after the late spring herbicide treatment) and again between September 25 and October 5, 2018, as checking mortality immediately after application may not accurately reflect plant mortality due to herbicide, and some managers claim there is a delayed herbicide effect (B. Kleiman, pers. comm.). Plants without any green leaves were marked as dead, and those with leaves were marked as alive.

To examine longer term nontarget effects of the herbicide treatments on the surrounding vegetation communities, we randomly selected four L. *maackii* from each herbicide treatment within four subplots, for a total of 80 survey locations in spring 2018. Initially, two burned and two unburned subplots were selected for nontarget effects monitoring, but we removed fire from subsequent analyses because after the initial 2018 fire treatments, normal management resumed, and there was no further differentiation between burned and unburned subplots.

We evaluated plant species richness and percent cover between May 22 and June 2 annually for 4 years (2018–2021). We estimated percent cover of each species using a 1 m^2 quadrat centred on the L. *maackii* root burl so the percent cover including thatch and bare ground totalled 100%. Separate cover measurements were taken for each species based on whether it was healthy or showed signs of wilting or senescence where it was considered damaged. For analyses, we grouped damaged individuals with healthy individuals because both categorizations were still alive, and we removed dead individuals from the analyses. Some individuals that were difficult to identify were only identified to genus, but most plants were identified to species using a regional flora guide (Wilhelm & Rericha, 2017).

2.3 | Statistical analysis

We analysed L. *maackii* mortality using a binomial generalized linear mixed effects model and nontarget species richness and percent cover using linear mixed models constructed using maximum likelihood. The mortality model used herbicide treatment, prescribed fire at the individual plant (sampling point) level, and the herbicide-prescribed fire interaction as fixed factors and nested random

effects to account for the split-plot design (fire subplot within site), controlling for any variation in site differences that would also impact fire intensity. Since L. *maackii* height may impact susceptibility to herbicide or fire, we initially included plant height class (0–1 m, 1–2 m, 2–3 m and >3 m) as a covariate in the model, but we excluded it from the final model as it had no impact on L. *maackii* mortality either as an interaction or fixed factor.

Both species richness and percent cover models used herbicide treatment and an interaction of sampling year and plant native status as fixed effects at the individual sampling point level, with site as a random effect. For the percent cover model, we log-transformed mean native and non-native percent cover for analysis to meet model assumptions and normalize model residuals, and evaluated model fit by extracting and plotting residuals. For both models, sampling year was analysed as a factor and not continuous variable, as we did not hypothesize that plant recovery would follow a linear trajectory.

We assessed mortality and nontarget models using Type II ANOVAs ($\alpha = 0.05$). All models were constructed in the lmer4 package (Bates et al., 2015), using the glmer() and lmer() functions, and assessed using the Anova() function in the car package (Fox & Weisberg, 2011) in R (R Core Team and Korpela, 2013) and the getgof() function from the modelsummary package (Arel-Bundock, 2022), with factors only considered significant if $p < 0.05$. If factors were significant, we analysed pairwise comparisons using the emmeans() function in the emmeans package, with a Tukey HSD adjustment (Lenth, 2023). Three L. *maackii* could not be relocated when collecting mortality information due to a fallen tree or fire disruption and were not included in mortality models, resulting in a sample size of 797 plants.

3 | RESULTS

3.1 | Mortality of L. *maackii*

Mortality was significantly impacted by herbicide treatment ($\chi^2 = 80.01$, $p < 0.001$), with plants in the control group experiencing 2.5% mortality and all herbicide treatment seasons resulting in ≥96.9% mortality. Seasonal herbicide treatments were not statistically different from one another (Figure 2), resulting in an average basal bark mortality of 98.4% across all sites and burn plots. During the spring 2018 check of treated honeysuckle, 74.7% of treated plants began to produce leaves, but the final mortality for all herbicide treatments by September 2018 was ≥96.9%, indicating a delayed herbicide effect. Mortality was not significantly different between burned and unburned plots ($\chi^2 = 0.11$, $p = 0.74$), nor was the interaction between burn plot and herbicide treatment season significantly different ($\chi^2 = 1.36$, $p = 0.85$). The fixed effects explained 93% of the variance in the model, and the variance explained by the random effect (fire subplot nested within site) was small (<2%), indicating that site-specific factors affecting burn intensity had little effect on mortality.

3.2 | Nontarget vegetation impacts

Herbicide treatment significantly impacted species richness ($\chi^2 = 17.03$, $p = 0.002$), with greater species richness in the control group than fall, winter and late spring treatments, but not greater than the early spring treatment (Figure 3). Herbicide treatment groups had on average 2 fewer species than the control group, with the early spring treatment having 1.4 fewer species and the other treatments having 1.9–2.3 fewer species. Sampling year did not significantly impact overall species richness ($\chi^2 = 4.61$, $p = 0.20$). Native species richness was significantly higher than non-native species richness ($\chi^2 = 1108.70$, $p < 0.001$), and neither native nor non-native

species richness changed differently over time (i.e. no significant interaction of sampling year and native status; $\chi^2 = 0.81$, $p = 0.85$). Native species richness ranged from 7.6 to 9.2 species, and non-native species richness ranged from 1.6 to 1.9 species. The fixed effects in the model explained 55% of the variance in the data, and the random effect of site explained an additional 17% of the variance (for a total $R^2 = 72\%$).

Over the entire four-year period, herbicide treatment time significantly impacted average percent cover per species ($\chi^2 = 22.36$, $p < 0.001$), with cover in the control group higher than in the fall and winter treatments, but not early or late spring treatments (Figure 4). Herbicide treatment resulted in an overall percent cover reduction of 15% across all 4 years, with individual treatment reductions ranging from 11% (early spring) to 21% (winter).

Sampling year significantly impacted percent cover per species ($\chi^2 = 17.21$, $p = <0.001$). Following herbicide treatments in spring 2018, herbicide-treated plots had on average 20.2% less total non-target cover, with a mean of 31.1% percent cover per plot for all treatment groups and 51.3% per plot in the control group. Across all 4 years, native species percent cover was not significantly higher than non-native percent cover ($\chi^2 = 1.19$, $p = 0.29$), and neither native nor non-native species percent cover changed differently over time (the sampling year × native status interaction was not significant, $\chi^2 = 2.02$, $p = 0.57$). The fixed effects in the model explained 6.5% of the variance in the data, and the random effect of site explained an additional 7.7% of the variance (for a total $R^2 = 14.2\%$).
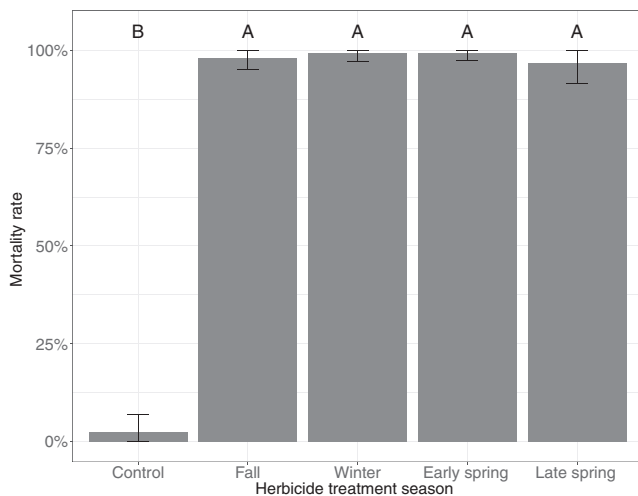

FIGURE 2 *Lonicera maackii* mortality across control and herbicide treatment groups. Bars are mean percent mortality of treatments, and error bars indicate standard deviation of the mean across all plots. Letters A and B indicate differences between groups.

4 | DISCUSSION

Basal bark treatments across all seasons were highly effective at killing *L. maackii*, whereas prescribed fire was ineffective by itself
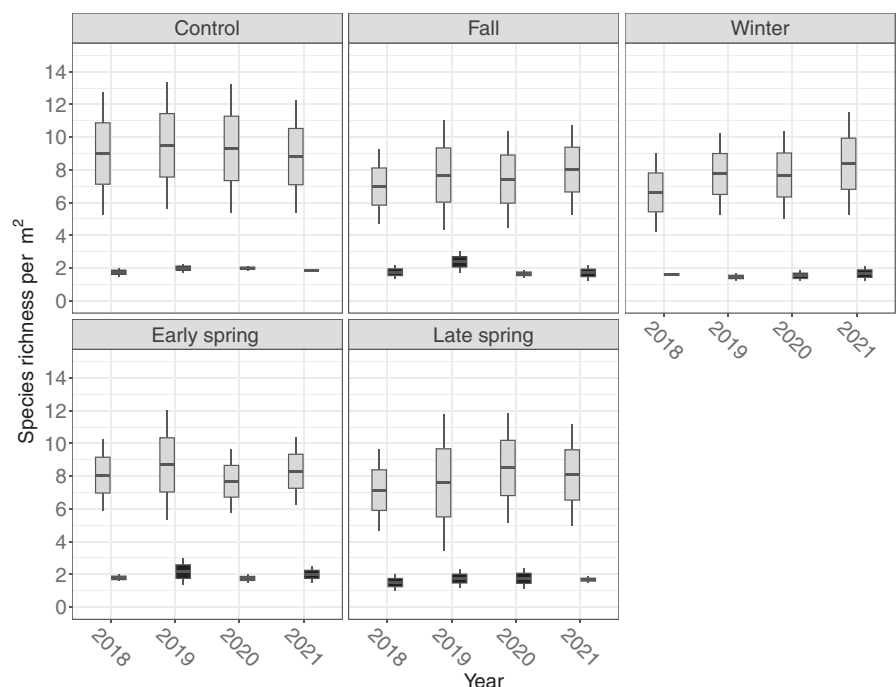

FIGURE 3 Average native (grey) and non-native (black) species richness per m² from 2018 to 2021 from all sampling locations ($n = 80$). Bold centre bars indicate the median. Hinges indicate the middle 50% of the data, and whiskers include data within 1.5 × the interquartile range or distance from the 1st to the 3rd quartiles.
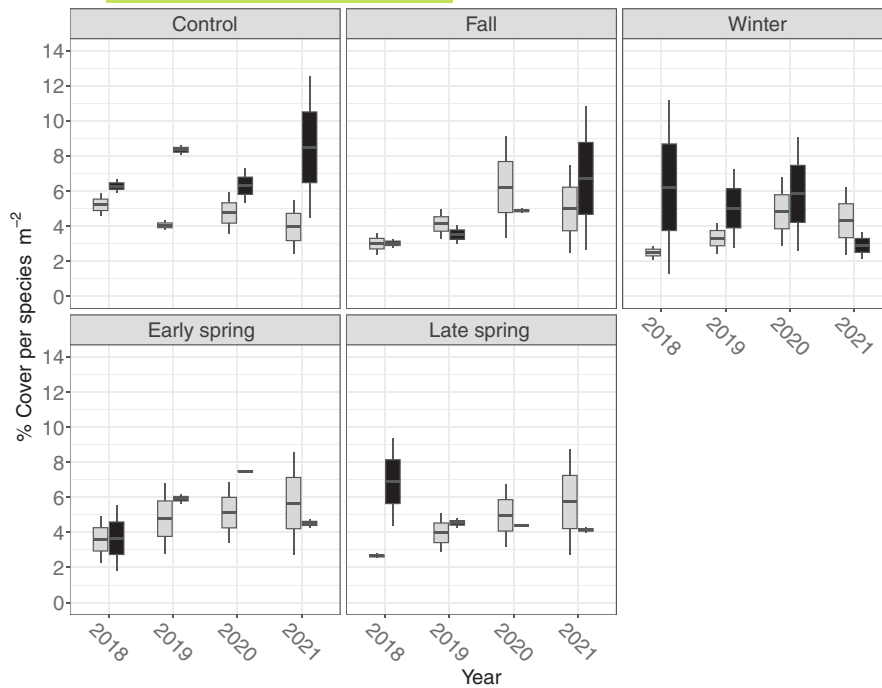
FIGURE 4 Average native (grey) and non-native (black) percent cover per species, per m² (not total percent cover per m²) from 2018 to 2021, from all sampling locations (n=80). Bold centre bars indicate the median. Hinges indicate the middle 50% of the data, and whiskers include data within 1.5×the interquartile range or distance from the 1st to the 3rd quartiles.

and did not alter the impacts of basal bark treatments. There was a minor herbicide impact on the native plant community, resulting in reduced species richness and percent cover by about 15% for some treatments, although richness and percent cover of non-native species did not increase in any overall pattern in the 3 years following treatment. There was some evidence that treating *L. maackii* with herbicide in its late dormancy period in early spring (March 10–11 in this study) may have slightly reduced impacts on native plant communities compared to other treatment times, with the smallest loss of native species richness and cover. Overall, herbicide treatment resulted in an average loss of two species in the immediately surrounding plant community (native and non-native). Taken together, these results indicate that managers can use basal bark application to control *L. maackii*, regardless of site fire status or application time outside of the active growing season, and they can expect results that meet management objectives of invasive control without extensive nontarget effects.

4.1 | Management impacts on *L. maackii*

Basal application of triclopyr from fall to late spring effectively killed 98.4% of all herbicide-treated *L. maackii* (Figure 2). These results are similar to a previous study at a nearby site, which found 100% mortality from basal bark applications (Kleiman et al., 2018). However, other studies have reported reduced effectiveness, such as ≤40% mortality (Rathfon & Ruble, 2007) or effectiveness as low as ≤2% mortality for fall treatments and ≤35% mortality for spring treatments (Riley, 2013). Both of these studies reported mortality by visually estimating treatment plot mortality and did not assess the individual plant mortality as here and in Kleiman et al. (2018). Mortality assessments at the larger plot scale do not consider new

recruitment from the seed bank or incomplete herbicide application from operator error and can lead researchers to conclude limited mortality from more targeted treatments. Plot-based mortality assessments would, therefore, favour less targeted treatments that do not require the operator to keep track of individuals that have not yet been treated or favour treatments that damage the seed bank and adjacent native plants. The extremely low mortality reported by Riley (2013) may also be due to pre-emptive mortality determination that did not allow sufficient time for the herbicide to be transported throughout the plant or for the leaves to drop. In our study, our apparent 'mortality rate' at the time of our spring 2018 leaf-out check was only 25.3% but had increased to 98.4% by the fall of that year. Therefore, allowing sufficient time before evaluating mortality (4–10 months in our study) is likely important when determining herbicide effectiveness.

Some herbicides' effectiveness can differ by season, and often there is greater mortality when application occurs during seasons when herbicide is actively transported throughout the plant (Fuchs & Geiger, 2005). This may explain the reduced efficacy of foliar application of triclopyr on invasive shrubs with summer and fall treatments, especially under drought stress, compared to more effective spring treatments when the herbicide actively moves through the phloem (Lanini, 1992; Lanini & Radosevich, 1982; Riley, 2013). However, we show that basal bark application of triclopyr is not impacted by the dormancy status of *L. maackii*, with consistent high-mortality treatment effects (≥96.9% mortality) for each treatment season.

Prescribed fire was not effective at killing *L. maackii* nor did it limit the efficacy of basal bark treatments. Previous studies provide mixed mortality results, reporting both a lack of effect like we found here (Zouhar et al., 2008), and evidence of effective control (Batcher & Stiles, 2000; Nyboer, 2007; Saxton et al., 2016). *Lonicera maackii*

leaf litter does not carry fire well due to its relatively small leaves (Engber & Morgan Varner, 2012), resulting in limited fire intensity that is likely not hot enough nor resides long enough to kill the roots, allowing *L. maackii* to readily resprout after being top-killed (Klein & McClintock, 1992). However, fire intensity surrounding *L. maackii* has not been studied and is likely widely variable across locations, depending on fuel type, fuel load and distribution of burnable fuels beneath the shrubs. Both the type of burnable fuels and the continuity of fuels (how connected the fuels are) would impact the effectiveness of prescribed fire as a *L. maackii* control method (Batcher & Stiles, 2000). When *L. maackii* are sparsely distributed with nearly continuous well-burning fuels between plants, fire may top-kill the plants. However, fire is a disturbance that may facilitate *L. maackii* invasion by opening the canopy and enhancing nitrogen availability (Luken et al., 1995; Taylor & Midgley, 2018; Zouhar et al., 2008). In the context of fire-adapted woodland and savanna management, which relies on fire disturbance to maintain an open canopy (Bowles et al., 2003; McEwan et al., 2007), it is not recommended to remove fire from the ecosystem. Rather, it is recommended that managers be more vigilant in recently burned areas for new recruitment of *L. maackii* and other non-natives that may necessitate treatment and removal (Guthrie et al., 2016). The method used in this study may be a time- and resource-effective way of doing this by allowing managers to quickly treat newly recruited *L. maackii* individuals and not have to spend more time on mechanical removal.

4.2 | Management impacts on nontarget vegetation

To our knowledge, the damage to nontarget vegetation caused by different seasonal basal bark applications of triclopyr has not yet been examined. Immediately following treatment (in 2018), plants in all triclopyr treatment seasons suffered similar damage. This differs from other herbicides such as glyphosate which produces less nontarget damage during winter and spring treatments due to its rapid uptake during photosynthesis and rapid degradation in the environment (Love & Anderson, 2009; Merriam, 1999). However, over the course of the entire study, only fall and winter basal bark treatments resulted in an overall reduction of nontarget percent cover. Most of the damage was not from herbicide directly contacting nontarget vegetation, because the winter and early spring seasons had no vegetative cover present and photodegradation would still occur for any unabsorbed above-ground herbicide (Tu et al., 2001, 2003). Residual herbicide in the soil could be responsible despite triclopyr's 30-day half-life in soil, because soil microbes that degrade the herbicide may be less metabolically active during cold, dormant months (Classen et al., 2015; Onwuka, 2018; Tu et al., 2001, 2003). A companion study to this one examined the impacts of basal bark treatments on soil microbial communities and found weak or no biological responses (James et al., 2022); a decline in arbuscular mycorrhizal fungi colonization of sprayed *L. maackii* might even be expected to benefit surrounding plants if the fungi shift colonization and nutrient

benefits to surrounding plants. A release of allelopathic chemicals (Stinson et al., 2006) from *L. maackii* roots upon death also is possible, but the potential mechanisms of nontarget damage require more research.

Eight of the 10 most abundant species in plant surveys (summed live cover across all treatments, including the control) were native, with *L. maackii* being the most abundant non-native species, followed by *Alliaria petiolata*. The most abundant native species included *Antenoron virginianum*, *Circaea canadensis*, *Osmorhiza* spp., *Sanincula odorata*, *Parthenocissus quinquefolia*, *Viola* spp. and *Carex* spp. The most abundant native species, jumpseed (*A. virginianum*), increased in average plot-level cover from 6% in 2018 to 14% in 2021, and the most abundant non-native, nontarget species, garlic mustard (*A. petiolata*), increased from 1% to 6% average cover from 2018 to 2021, respectively. Land managers may need to be vigilant for local incursions of garlic mustard or other non-natives following honeysuckle treatment, as removing live honeysuckle canopy cover may encourage increased spread or germination from the seed bank (Frank et al., 2018).

Despite the statistical differences in richness and percent cover, the biological difference (i.e. the effect size) was minimal, with a mean difference of approximately two species lost directly around *L. maackii*, and a mean percent cover per species difference of fifteen percent. Managers could avoid basal spraying of triclopyr in very high-quality areas that contain sensitive species in the fall, winter and later in the spring. However, the benefits of *L. maackii* removal likely outweigh the temporary negative impacts of basal bark removal treatments because there was some evidence of native cover increases in plots following basal bark treatments, but not in control plots without basal bark treatments.

5 | CONCLUSIONS

Basal bark application of triclopyr is an effective means of killing *L. maackii* regardless of when managers choose to apply herbicide. This study did not find that fire is an effective control method, but fire did not inhibit triclopyr effects and should be implemented to maintain fire-adapted communities. There is localized damage to understory vegetation around the *L. maackii* base following application during some seasons, but the extent of nontarget damage from basal bark treatments is minimal compared to the long-term direct negative impacts of *L. maackii* on native plant communities. By using quick and easy targeted treatments like basal bark application that are effective regardless of application season, managers can promote restoration goals of reestablishing native plant communities by locally extirpating *L. maackii* in its invaded range and focusing on recently invaded areas where ecosystems have not yet been severely altered.

AUTHOR CONTRIBUTIONS
Katharine F. E. Hogan designed analyses, analysed data and wrote the manuscript. Kaleb Baker conceived and designed the study, collected the data, analysed the data and wrote the manuscript.

CONFLICT OF INTEREST STATEMENT

Elizabeth Bach is employed by The Nature Conservancy, which owns and manages the land on which work was conducted. Elizabeth Bach is an Associate Editor of *Ecological Solutions and Evidence*, but took no part in the peer review and decision-making processes for this paper. At the time of submission Dr. Holly Jones was Katharine Hogan's supervisor and Senior Editor of *Ecological Solutions and Evidence* and took no part in the peer review and decision-making processes for this paper.

PEER REVIEW

The peer review history for this article is available at https://www.webofscience.com/api/gateway/wos/peer-review/10.1002/2688-8319.12332.

DATA AVAILABILITY STATEMENT

The data (Hogan, 2024) analysed in this publication are publicly available on Zenodo (https://zenodo.org/records/10999338).

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REFERENCES

Arel-Bundock, V. (2022). Modelsummary: Data and model summaries in R. *Journal of Statistical Software*, *103*(1), 1–23.

Baker, K. (2019). *Controlling Lonicera maackii (Amur honeysuckle): Basal bark and prescribed fire efficacy and impacts*. Northern Illinois University. https://huskiecommons.lib.niu.edu/allgraduate-thesesdissertations/1961/

Batcher, M. S., & Stiles, S. A. (2000). *Element stewardship abstract for Lonicera maackii (Rupr.) Maxim (Amur honeysuckle), Lonicera morrowii A. Gray (Morrow's honeysuckle), Lonicera tatarica L. (Tatarian honeysuckle), Lonicera x bella Zabel (Bell's honeysuckle), the bush honeysuckles*. The Nature Conservancy.

Bates, D., Mächler, M., Bolker, B., & Walker, S. (2015). Fitting linear mixed-effects models using Lme4. *Journal of Statistical Software*, *67*(1), 1–48. https://doi.org/10.18637/jss.v067.i01

Bowles, M., Jones, M., Dunn, C., McBride, J., Bushey, C., & Moran, R. (2003). Twenty-year woody vegetation changes in northern flatwoods and mesic forest at Ryerson Conservation Area, Lake County, Illinois. *Erigenia*, *19*(September), 31–51.

CABI. (2018). Lonicera maackii (Amur honeysuckle). In *Invasive species compendium*. CAB International. https://www.cabi.org/isc/datasheet/31192

Castellano, S. M., & Gorchov, D. L. (2013). White-tailed deer (*Odocoileus virginianus*) disperse seeds of the invasive shrub, Amur honeysuckle (*Lonicera maackii*). *Natural Areas Journal*, *33*(1), 78–80. https://doi.org/10.3375/043.033.0109

Christopher, C. C., & Cameron, G. N. (2012). Effects of invasive Amur honeysuckle (*Lonicera maackii*) and white-tailed deer (*Odocoileus virginianus*) on litter-dwelling arthropod communities. *The American Midland Naturalist*, *167*(2), 256–272. https://doi.org/10.1674/0003-0031-167.2.256

Cipollini, D., & Dorning, M. (2008). Direct and indirect effects of conditioned soils and tissue extracts of the invasive shrub, *Lonicera maackii*, on target plant performance. *Castanea*, *73*(3), 166–176.

Cipollini, D., Stevenson, R., Enright, S., Eyles, A., & Bonello, P. (2008). Phenolic metabolites in leaves of the invasive shrub, *Lonicera maackii*, and their potential phytotoxic and anti-herbivore effects. *Journal of Chemical Ecology*, *34*(2), 144–152. https://doi.org/10.1007/s10886-008-9426-2

Cipollini, K., Ames, E., & Cipollini, D. (2009). Amur honeysuckle (*Lonicera maackii*) management method impacts restoration of understory plants in the presence of white-tailed deer (*Odocoileus virginiana*). *Invasive Plant Science and Management*, *2*(1), 45–54. https://doi.org/10.1614/IPSM-08-108.1

Classen, A. T., Sundqvist, M. K., Henning, J. A., Newman, G. S., Moore, J. A. M., Cregger, M. A., Moorhead, L. C., & Patterson, C. M. (2015). Direct and indirect effects of climate change on soil microbial and soil microbial-plant interactions: What lies ahead? *Ecosphere*, *6*(8), 130. https://doi.org/10.1890/ES15-00217.1

Collier, M. H., Vankat, J. L., & Hughes, M. R. (2002). Diminished plant richness and abundance below *Lonicera maackii*, an invasive shrub. *American Midland Naturalist*, *147*(1), 60–71.

Czarapata, E. J. (2005). *Invasive plants of the upper midwest: An illustrated guide to their identification and control* (pp. 32–35). The University of Wisconsin Press.

Deering, R. H., & Vankat, J. L. (1999). Forest colonization and developmental growth of the invasive shrub *Lonicera maackii*. *American Midland Naturalist*, *141*(1), 43–50.

Dutra, H. P., Barnett, K., Reinhardt, J. R., Marquis, R. J., & Orrock, J. L. (2011). Invasive plant species alters consumer behavior by providing refuge from predation. *Oecologia*, *166*(3), 649–657. https://doi.org/10.1007/s00442-010-1895-7

Engber, E. A., & Morgan Varner, J. (2012). Patterns of flammability of the California oaks: The role of leaf traits. *Canadian Journal of Forest Research*, *42*(11), 1965–1975. https://doi.org/10.1139/x2012-138

Fantle-Lepczyk, J. E., Haubrock, P. J., Kramer, A. M., Cuthbert, R. N., Turbelin, A. J., Crystal-Ornelas, R., Diagne, C., & Courchamp, F. (2022). Economic costs of biological invasions in the United States. *Science of the Total Environment*, *806*, 151318.

Fargen, C., Emery, S. M., & Carreiro, M. M. (2015). Influence of *Lonicera maackii* invasion on leaf litter decomposition and macroinvertebrate communities in an urban stream. *Natural Areas Journal*, *35*(3), 392–403. https://doi.org/10.3375/043.035.0303

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9 of 10

Fox, J., & Weisberg, S. (2011). *An {R} companion to applied regression* (2nd ed.). Sage. http://socserv.socsci.mcmaster.ca/jfox/Books/Companion

Frank, G. S., Saunders, M. R., & Jenkins, M. A. (2018). Short-term vegetation responses to invasive shrub control techniques for Amur honeysuckle (*Lonicera maackii* [Rupr.] herder). *Forests*, 9, 607. https://doi.org/10.3390/f9100607

Fuchs, M. A., & Geiger, D. R. (2005). Assessing herbicidal damage in Amur honeysuckle, *Lonicera maackii*, stem tissue. In J. Cardina (Ed.), *Ohio invasive plant research conference: Bridging the gap between land management and research proceedings* (pp. 95–100). Ohio Agricultural Research and Development Center.

Gayek, A. (2000). *Effect of habitat variation on rate and success of colonization of two invasive shrubs, Lonicera maackii and Ligustrum vulgare, in a forested Glen*. Ohio State University.

Gorchov, D. L., Castellano, S. M., & Noe, D. A. (2014). Long-distance dispersal and diffusion in the invasion of *Lonicera maackii*. *Invasive Plant Science and Management*, 7(3), 464–472. https://doi.org/10.1614/IPSM-D-13-00105.1

Gorchov, D. L., & Trisel, D. E. (2003). Competitive effects of the invasive shrub, *Lonicera maackii* (Rupr.) herder (Caprifoliaceae), on the growth and survival of native tree seedlings. *Plant Ecology*, 166(1), 13–24. https://doi.org/10.1023/A:1023208215796

Gosper, C. R., Stansbury, C. D., & Vivian-Smith, G. (2005). Seed dispersal of fleshy-fruited invasive plants by birds: Contributing factors and management options. *Diversity and Distributions*, 11(6), 549–558. https://doi.org/10.1111/j.1366-9516.2005.00195.x

Gould, A. M. A., & Gorchov, D. L. (2000). Effects of the exotic invasive shrub *Lonicera maackii* on the survival and fecundity of three species of native annuals. *American Midland Naturalist*, 144(1), 36–50.

Guthrie, S. G., Crandall, R. M., & Knight, T. M. (2016). Fire indirectly benefits fitness in two invasive species. *Biological Invasions*, 18(5), 1265–1273. https://doi.org/10.1007/s10530-016-1064-y

Hartman, K. M. (2005). *The impacts, Invasibility, and restoration ecology of an invasive shrub, Amur honeysuckle (Lonicera maackii)*. Ohio University. https://etd.ohiolink.edu/pg_10?0::NO:10:P10_ACCESSION_NUM:ohiou1133734744.

Hartman, K. M., & McCarthy, B. C. (2008). Changes in forest structure and species composition following invasion by a non-indigenous shrub, Amur honeysuckle (*Lonicera maackii*). *The Journal of the Torrey Botanical Society*, 135(2), 245–259. https://doi.org/10.3159/07-RA-036.1

Hogan, K. (2024). Data from: Basal bark herbicide treatment of Lonicera maackii (Amur honeysuckle) is effective regardless of application timing, with limited nontarget effects on native plant diversity (v1.0) [data set]. Zenodo https://doi.org/10.5281/zenodo.10999338

Holmes, K. A., & Berry, A. M. (2009). Evaluation of off-target effects due to basal bark treatment for control of invasive fig trees (*Ficus Carica*). *Invasive Plant Science and Management*, 2(4), 345–351. https://doi.org/10.1614/IPSM-09-029.1

Hopfensperger, K. N., Boyce, R. L., & Schenk, D. (2017). Removing invasive *Lonicera maackii* and seeding native plants alters riparian ecosystem function. *Ecological Restoration*, 35(4), 320–327.

Ingold, J. L., & Craycraft, M. J. (1983). Avian Frugivory on honeysuckle (Lonicera) in southwestern Ohio in fall. *Ohio Journal of Science*, 83(5), 256–258.

James, J. J., Bach, E. M., Baker, K., Barber, N. A., Buck, R., Shahrtash, M., & Brown, S. P. (2022). Herbicide control of the invasive Amur honeysuckle (*Lonicera maackii*) does not alter soil microbial communities or activity. *Ecological Solutions and Evidence*, 3(3), e12157. https://doi.org/10.1002/2688-8319.12157

Kimball, S., Lulow, M., Sorenson, Q., Balazs, K., Fang, Y.-C., Davis, S. J., O'Connell, M., & Huxman, T. E. (2015). Cost-effective ecological restoration. *Restoration Ecology*, 23(6), 800–810. https://doi.org/10.1111/rec.12261

Kleiman, L. R., Kleiman, B. P., & Kleiman, S. (2018). Successful control of *Lonicera maackii* (Amur honeysuckle) with basal bark herbicide. *Ecological Restoration*, 36(4), 267–269.

Klein, V. M., & McClintock, T. (1992). Effect of burning on a dry oak forest infested with woody exotics. In *Spirit of the Land, Our Prairie Legacy: Proceedings 13th North American Prairie Conference* (pp. 207–213). Department of Parks and Recreation.

Lanini, W. T. (1992). Forest chemicals: Mode of action. In *Forest Vegetation Management Conference Proceedings* (Vol. 11, p. 118). Forest Vegetation Management Conference.

Lanini, W. T., & Radosevich, S. R. (1982). Herbicide effectiveness in response to season of application and shrub physiology. *Weed Science*, 30, 467–475.

Lenth, R. V. (2023). Emmeans: Estimated marginal means, aka least-square means. R.

Lieurance, D., & Cipollini, D. (2012). Damage levels from arthropod herbivores on *Lonicera maackii* suggest enemy release in its introduced range. *Biological Invasions*, 14(4), 863–873. https://doi.org/10.1007/s10530-011-0123-7

Love, J. P., & Anderson, J. T. (2009). Seasonal effects of four control methods on the invasive Morrow's honeysuckle (*Lonicera morrowii*) and initial responses of understory plants in a southwestern Pennsylvania old field. *Restoration Ecology*, 17(4), 549–559. https://doi.org/10.1111/j.1526-100X.2008.00421.x

Luken, J. O. (1988). Population structure and biomass allocation of the naturalized shrub *Lonicera maackii* (Rupr.) maxim. in forest and open habitats. *The American Midland Naturalist*, 119(2), 258–267. https://doi.org/10.2307/2425809

Luken, J. O., Hinton, A. C., & Baker, D. G. (1991). Assessment of frequent cutting as a plant-community management technique in power-line corridors. *Environmental Management*, 15(3), 381–388. https://doi.org/10.1007/BF02393884

Luken, J. O., Kuddes, L. M., & Tholemeier, T. C. (1997). Response of understory species to gap formation and soil disturbance in *Lonicera maackii* thickets. *Restoration Ecology*, 5(3), 229–235. https://doi.org/10.1046/j.1526-100X.1997.09727.x

Luken, J. O., & Mattimiro, D. T. (1991). Habitat-specific resilience of the invasive shrub Amur honeysuckle (*Lonicera maackii*) during repeated clipping. *Ecological Applications*, 1(1), 104–109. https://doi.org/10.2307/1941852

Luken, J. O., & Thieret, J. W. (1996). Amur honeysuckle, its fall from grace. *BioScience*, 46(1), 18–24.

Luken, J. O., Tholemeier, T. C., Kunkel, B. A., & Kuddes, L. M. (1995). Branch architecture plasticity of Amur honeysuckle (*Lonicera maackii* (Rupr.) herder): Initial response in extreme light environments. *Bulletin of the Torrey Botanical Club*, 122(3), 190–195. https://doi.org/10.2307/2996083

McEwan, R. W., Hutchinson, T. F., Long, R. P., Robert Ford, D., & McCarthy, B. C. (2007). Temporal and spatial patterns in fire occurrence during the establishment of mixed-oak forests in eastern North America. *Journal of Vegetation Science*, 18(5), 655–664. https://doi.org/10.1111/j.1654-1103.2007.tb02579.x

Meiners, S. J. (2007). Apparent competition: An impact of exotic shrub invasion on tree regeneration. *Biological Invasions*, 9(7), 849–855. https://doi.org/10.1007/s10530-006-9086-5

Merriam, R. (1999). Winter spraying with glyphosate kills honeysuckle, does no damage to Groundlayer plants (North Carolina). *Ecological Restoration*, 17(1/2), 87.

Nyboer, R. (2007). *Review of Vegetation Management Guideline Bush Honeysuckles: Tartarian, Marrow's, Belle, and Amur Honeusuckle (Lonicera tatarica L., L. Morrowwii Gray, L. x bella Zabel, and L. maackii (Rupr.) Maxim.), by Bob Edgin*. Illinois Nature Preserves Commission. https://www.dnr.illinois.gov/inpc/documents/vmg/vmg%20bush%20honeysuckle%20revised%202007%20-%20copy.pdf

Olson, R. A., & Whitson, T. D. (2002). Restoring structure in late-successional sagebrush communities by thinning with Tebuthiuron. *Restoration Ecology, 10*(1), 146–155. https://doi.org/10.1046/j.1526-100X.2002.10116.x

Onwuka, B. M. (2018). Effects of soil temperature on some soil properties and plant growth. *Advances in Plants & Agriculture Research, 8*(1), 34–37. https://doi.org/10.15406/apar.2018.08.00288

R Core Team, & Korpela, M. (2013). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. http://www.R-project.org/

Rathfon, R., & Ruble, K. (2007). Herbicide treatments for controlling invasive bush honeysuckle in a mature hardwood forest in West-Central Indiana. In *Proceedings, 15th Central Hardwood Forest Conference. USDA Forest Service E-Gen. Technical Report SRS-101* (pp. 187–197). U.S. Department of Agriculture Forest Service Southern Research Station.

Reinartz, J. A. (1997). Controlling glossy buckthorn (*Rhamnus frangula* L.) with winter herbicide treatments of cut stumps. *Natural Areas Journal, 17*(1), 38–41.

Riley, S. A. (2013). *Biology and control of Amur honeysuckle.* University of Missouri-Columbia.

Roy, H. E., Pauchard, A., Stoett, P., Renard Truong, T., Bacher, S., Galil, B. S., Hulme, P. E., Ikeda, T., Sankaran, K., McGeoch, M. A., Meyerson, L. A., Nuñez, M. A., Ordonez, A., Rahlao, S. J., Schwindt, E., Seebens, H., Sheppard, A. W., & Vandvik, V. (2023). *Summary for policymakers of the thematic assessment report on invasive alien species and their control of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services*. IPBES Secretariat. https://doi.org/10.5281/zenodo.7430692

Saxton, M., Kleiman, B., Walk, J., & Hagen, S. (2016). *Illinois fire needs assessment*. Illinois Prescribed Fire Council. https://www.illinoisprescribedfirecouncil.org/uploads/1/0/5/8/105892833/il_fire_needs_assessment.pdf

Schmidt, K. A., & Whelan, C. J. (1999). Effects of exotic Lonicera and Rhamnus on songbird Nest predation. *Conservation Biology, 13*(6), 1502–1506.

Stinson, K. A., Campbell, S. A., Powell, J. R., Wolfe, B. E., Callaway, R. M., Thelen, G. C., Hallett, S. G., Prati, D., & Klironomos, J. N. (2006). Invasive plant suppresses the growth of native tree seedlings by disrupting belowground mutualisms. *PLoS Biology, 4*(5), e140. https://doi.org/10.1371/journal.pbio.0040140

Swearingen, J., Slattery, B., Reshetiloff, K., & Zwicker, S. (2010). *Plant invaders of mid-Atlantic natural areas: Revised & updated—with more species and expanded control guidance*. National Park Service and US Fish and Wildlife Service. http://www.nyis.info/user_uploads/files/Plant%20Invaders%20of%20Mid-Atlantic.pdf

Taylor, Q. A., & Midgley, M. G. (2018). Prescription side effects: Long-term, high-frequency controlled burning enhances nitrogen availability in an Illinois Oak-dominated Forest. *Forest Ecology and Management, 411*(March), 82–89. https://doi.org/10.1016/j.foreco.2017.12.041

Tu, M., Hurd, C., Robinson, R., & Randall, J. M. (2003). K: Triclopyr. In *Weed control methods handbook, 7k.1-7k.8*. The Nature Conservancy. https://www.invasive.org/gist/products/handbook/20.triclopyr.pdf

Tu, M., Hurd, C., & Randall, J. M. (2001). Weed control methods handbook: Tools & techniques for use in natural areas. p. 219.

Vitousek, P. M., Mooney, H. A., Lubchenco, J., & Melillo, J. M. (1997). Human domination of earth's ecosystems. *Science, 277*(5325), 494–499. https://doi.org/10.1126/science.277.5325.494

Welch, B. A., Geissler, P. H., & Latham, P. (2014). *Early detection of invasive plants—Principles and practices: U.S. Geological Survey Scientific Investigations Report 2012–5162*. Scientific Investigations Report. https://doi.org/10.3133/sir20125162

Wilcove, D. S., Rothstein, D., Dubow, J., Phillips, A., & Losos, E. (1998). Quantifying threats to imperiled species in the United States. *BioScience, 48*(8), 607–615. https://doi.org/10.2307/1313420

Wilhelm, G., & Rericha, L. (2017). *Flora of the Chicago region: A floristic and ecological synthesis*. Indiana Academy of Science.

Witmer, M. C. (1996). Consequences of an alien shrub on the plumage coloration and ecology of cedar waxwings. *The Auk, 113*(4), 735–743. https://doi.org/10.2307/4088853

Zouhar, K., Smith, J. K., & Sutherland, S. (2008). *Effects of fire on nonnative invasive plants and Invasibility of wildland ecosystems* (pp. 7–32). USDA Forest Service Gen. Tech. Rep. RMRS-GTR-42-vol. 6.

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