Small-Scale Amur Honeysuckle Removal and Passive Restoration May Not Create Long-Term Success

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Research Note

Small-Scale Amur Honeysuckle Removal and Passive Restoration May Not Create Long-Term Success

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ABSTRACT

Amur honeysuckle (*Lonicera maackii*), a woody shrub native to northeastern Asia, is a common invasive species in many urban, suburban, and rural environments in North America. Honeysuckle negatively impacts native plant communities, and prolonged removal efforts are required to slow its spread and prevent reestablishment. However, intensive management is not always feasible for homeowners and small landowners, so the potential for small-scale honeysuckle removal followed by a passive approach is highly desirable. To test the potential for small-scale honeysuckle removal to initiate observable native plant recruitment, we established a long-term, small-scale study within a well-established honeysuckle infestation located in a 12.1 ha suburban patch of oak-hickory forest in Shawnee County, Kansas. We annually cleared 10 plots of all honeysuckle and maintained 10 adjacent, uncleared plots for the duration of the study. Native plant numbers increased within 1 y in the removal plots, and this increase continued across years. However, the vast majority of individuals consisted of aggressive and early successional species, and no significant differences in the effective number of common or dominant native species were observed between 2017 and 2020. Our results suggest that small-scale suppression of honeysuckle in well-established honeysuckle populations will likely lead to recolonization by a small number of species that may remain dominant for several years. Therefore, in areas with well-established honeysuckle populations, small-scale management of honeysuckle growth will likely be insufficient to ensure that even a moderate sample of native species becomes established.

Index terms: Amur honeysuckle; invasive shrubs; *Lonicera maackii*; passive restoration

INTRODUCTION

Negative impacts of invasive plant species can be extensive (e.g., Gould and Gorchov 2000; Collier et al. 2002; Poulette and Arthur 2012; Little et al. 2021), and management is often necessary. Unfortunately, management options are often labor intensive and may be prohibitive for small landowners in urban, suburban, and exurban environments. An increasingly large quantity of forest and woodland habitat or adjacent properties are owned by small landowners, and their actions can significantly impact habitat quality (Krasny and Tidball 2012; Kilgore and Snyder 2016; Mayer 2019; Cavender-Bares et al. 2020). Thus, there is a need to consider management practices that are feasible for small landowners to perform and to determine whether such practices would create successful management outcomes.

Active restoration strategies, such as planting seedlings or saplings following invasive plant species removal, can often increase the success of restoration efforts (Forbes et al. 2021). However, this approach requires that small landowners select, purchase, and tend to the plantings if they are to maximize their success, which may create a hurdle to the adoption of restoration efforts (Rohr et al. 2018; Hopfensperger et al. 2019). In contrast, passive restoration requires the same amount of initial effort to reduce the target species' population, but the reestablishment of the native community is determined by natural succession within the cleared area. While restoration of the site may be slower, the reduced commitment of time and resources could allow land managers to restore larger areas rather than focusing intense efforts on relatively restricted areas within a site. However, success of this approach is often dependent on the degree of degradation at the site (Prach et al. 2020) and assumes that locally available native species can serve as a reliable propagule source via the seed bank, new seed input, or clonal growth. Areas where highly dominant invasive species have been well established for prolonged periods of time may therefore limit the likelihood of success, particularly when management is conducted at a small scale. In addition, given that many invasive plant species can reinvade removal areas (Hopfensperger et al. 2019), continued maintenance is likely required, necessitating that a reasonable success rate be observed to prevent perceived failure and maintain stakeholder interest (Zahawi et al. 2014; Arsénio et al. 2020; Höhl et al. 2020).

Amur honeysuckle, *Lonicera maackii* (Rupr), is a common invasive species throughout much of the United States east of the Rocky Mountains, particularly in urban–exurban habitats, where it can functionally prevent the growth and establishment of other plant species (Collier et al. 2002; White et al. 2014; McNeish and McEwan 2016; Chen and Matter 2017; Sena et al. 2021). Management often requires extensive mechanical removal and/or herbicide applications (Schulz et al. 2012), which may pose philosophical/practical constraints to their adoption by small landowners. Several studies have indicated that removal of honeysuckle can lead to a limited but noticeable increase in native plant species cover (e.g. Luken et al. 1997; Runkle et al. 2007; Boyce 2015; Shields et al. 2015), even when the scale of removal is relatively small (Luken et al. 1997) or *L. maackii* density is high (Boyce 2015). However, the potential for honeysuckle removal followed by a passive approach at a scale manageable by homeowners within areas where *L. maackii* infestations are well established is unknown.

Here we were interested in the potential for *L. maackii* removal at a scale where annual efforts would be achievable for homeowners and small landowners and that could meet modest restoration goals in a well-established infestation. Specifically, we assessed the potential for small-scale removal to (1) reduce honeysuckle cover, (2) allow native tree seedlings to establish once honeysuckle is removed, and (3) allow the colonization of an understory community that is representative of native growth in the area.

METHODS

Study Site

The Karlyle Woods is a 12.14 ha (30 acre) oak-hickory forest tract located in a suburban area of the City of Topeka (Shawnee County, Kansas, USA; Latitude = 39.10°, Longitude = 95.70°). The site lies within a floodplain valley, with an elevation ranging from 270 to 300 m, and the climate corresponds to the humid continental, Köppen Dfa climate classification, with a USDA Plant Hardiness Zone 6. As part of a small survey, a 30 × 30 m grid was overlaid over a portion of the study area ($n = 37$ grid points), and *L. maackii* stems within a 10 m radius of each grid center were counted. The largest *L. maackii* stem was then removed and its age estimated using annual growth rings. Results from that survey indicated a mean density of 2.15 ± 0.2 stems per m^2, the mean age of the largest stems was 25.9 ± 0.89 y (max age 40 y). These results suggest that *L. maackii* has been present at the site for a minimum of 40 y and has been widespread at the site for at least 25 y. A few haphazard attempts to control the infestation began in the early 2000s, but the infestation has remained heavy with most areas never receiving any form of management.

Removal Treatments

In June 2016, 10 pairs of 113 m^2 (6 m diameter) circular plots were haphazardly established in the most highly infested areas at the site (20 plots). Plots within each pair were randomly assigned to one of two treatments, honeysuckle removal or control. The edges of the paired plots were within 15 m of each other and at least 30 m from other plot pairs. *L. maackii* density was high in all plots and was comparable between paired plots before the removal treatment was applied (Supplemental Table S1). In the removal plots, all honeysuckle stems were removed manually by clipping or sawing stems to within 5 cm of the ground. No herbicide treatments were used at any point in the study. All honeysuckle within 2 m of the edge of the plot was also clipped to minimize edge effects. Removed shrubs were cleared from the plots and placed in piles outside of the 2 m buffer. Plots were visited in mid-May to early June from 2017 to 2020 to conduct annual clipping in the removal plots, which represented a relatively small effort that could be easily maintained by homeowners and small landowners.

Following the initial honeysuckle removal in 2016, a photograph was taken at ground level facing up from the center of each plot. We used ImageJ (https://imagej.nih.gov/ij/) to estimate the proportion of unobstructed sky in these images as a crude proxy for relative canopy openness. The percent unobstructed sky in control plots was 10.07% ± 1.22% and for removal plots was 17.03% ± 1.57%. These results indicate that while overstory canopy cover was high in all plots, honeysuckle removal provided a substantial increase in direct sunlight reaching the forest floor even after full canopy leaf out.

Canopy Cover and Composition

Canopy cover in each plot was estimated using a modified dot count technique. A 3 m tall Jacob staff was moved in 10 cm increments along the length of a permanent transect, and the species identity of any leaves touching the staff was recorded at each point. To account for abundance of both ground cover and understory species, touches were recorded for points below 1 m and points above 2 m. Percent cover of each species was calculated as the total number of touches divided by the total number of points. Because the spontaneous establishment of native trees was a primary interest in this study, we completely scanned each plot for tree seedlings. The initial canopy estimates were performed mid to late June 2016. A second survey was completed in mid-September 2016. Although not formally analyzed, no perceivable changes in species cover had occurred in the plots. The June sampling period created logistical issues in the second year of the study so subsequent estimates were performed in early to mid-September (2017–2020).

Determining Seed Sources

Successful passive restoration requires a steady seed source of native tree species. The initial assessment of the species pool was limited to what was found in the plots. In 2019–2020, we documented the size (dbh) and species identity of all trees with a dbh ≥ 5 cm within 6 m of each plot to better assess the potential species pool at the site. *Lonicera maackii* was the only shrub present in or near the plots so any contribution by native shrubs would be incidental. Information is available in Supplemental Table S2.

Statistical Analyses

All analyses were conducted using R (R Core Team 2020).

Removal and Control Plot Comparisons: The effect of treatments on the total number of native individuals and number of species was analyzed using linear mixed models (LMM), where clipping was modeled as a fixed effect and year and plot as random effects. LMMs were conducted using the *lme4* package (Bates et al. 2015). Due to heteroscedasticity,

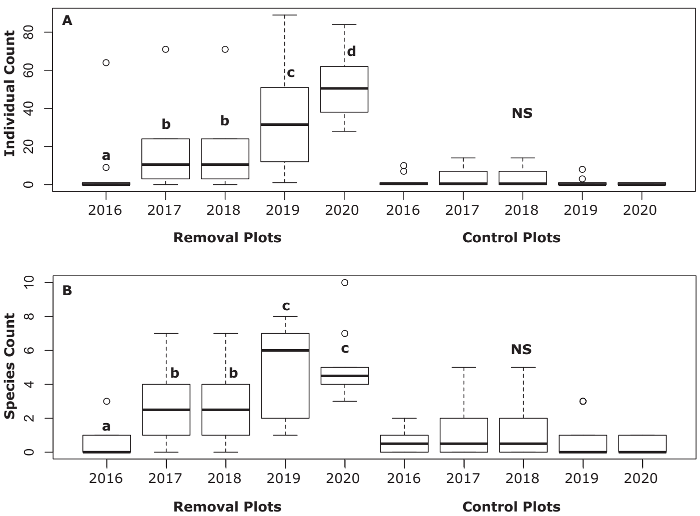

Figure 1.—Abundance (A; upper plot) and species richness (B; lower plot) of non–*L. maackii* stems in plots. In each figure, the bars represent the median, and the boxes represent the interquartile range with whiskers extending an additional 1.5*interquartile range or the min/max points if shorter. Open circles represent mild outliers. Within plot type (removal or control), years followed by the same letter are not significantly different at the $P < 0.05$ level. Note that low numbers of non–*L. maackii* individuals in the removal plots prevented between-year comparisons for those plots.

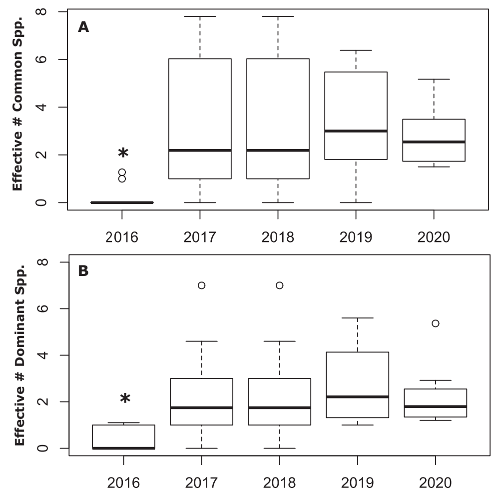

Figure 2.—Asymptotic estimates of the effective number of common species and the effective number of dominant species in *L. maackii* removal plots. (A; upper plot) The effective number of common species, estimated as the exponentiated Shannon index, and (B; lower plot) the effective number of dominant species (estimated as the inverse Simpson index). In each figure, the bars represent the median and the boxes represent the interquartile range with whiskers extending an additional 1.5*interquartile range or the min/max points if shorter. Open circles represent mild outliers, and asterisks indicate extreme outliers.

significance for the difference between treatments was estimated using permutation likelihood ratio tests using the *predictmeans* package (Dongwen et al. 2018). Total number of native individuals and species count were square-root transformed to meet distributional assumptions of residuals.

Diversity Estimates within Removal Plots: Due to the extremely low numbers of individuals other than *L. maackii* in the control plots (Figure 1), diversity estimates and between-year comparisons were not feasible. Therefore, diversity estimates were only determined for the removal plots. We used the *iNEXT* package (Hsieh et al. 2022) to calculate asymptotic estimates of Hill numbers of native plant species. Hill numbers have been rediscovered in ecology in the last two decades and have become increasingly used as they offer a more unified approach, which include Shannon and Simpson indices as special cases (Chao et al. 2014; Roswell et al. 2021). In addition, Hill numbers have the useful function of being expressed at the same scale and in units of species, in particular the effective number of species (equal to species richness), the effective number of common species (equal to Shannon diversity index exponentiated), and the effective number of dominant species (equal to the inverse of Simpson index). Individual rarefaction curves for Hill numbers were also conducted using the *iNEXT* package as estimates of sample coverage.

Between-years comparisons of Hill number estimates were conducted as LMMs considering year as a fixed effect and plots as random effects. LMMs were conducted using the *lme4* package and the *lmerTest* package (Kuznetsova et al. 2017) to calculate Satterthwaite approximations of *P*-values. Significance estimates of pairwise comparisons were conducted as Tukey adjusted multiple comparisons using the *emmeans* package (Lenth et al. 2017).

RESULTS AND DISCUSSION

The success of conservation relies on effective treatments that reduce pest species and increase the establishment of target species. However, these treatments must also be realistic in terms of time and effort required by property owners and land managers. The initial removal treatments required one week of clipping, sawing, and removal by two workers (80 person-hours total). Annual maintenance of the removal plots has been completed in a single morning by two workers, with nearly half of the time spent hiking between plots. Landowners removing *L. maackii* from more contiguous areas could complete this work with considerably less effort.

As has been observed in other *L. maackii* removal projects (Cipollini et al. 2009; Schulz et al. 2012; Hopfensperger et al. 2019), our small-scale *L. maackii* removal efforts resulted in modest increases in both the number (LRT: $\chi^2 = 59.261$, df $= 1$, Perm-$P < 0.001$) and the richness (LRT: $\chi^2 = 38.79$, df $= 1$, Perm-$P < 0.001$) of native plant species (Figure 1). This result was not surprising given the exceedingly low number of individual plants, other than *L. maackii*, present in control plots across all years (Figure 1). However, the increase in native plant species diversity was fairly low in our removal plots throughout the study.

Within removal plots, we detected a significant yearly increase in the total number of non–*L. maackii* individual plants between 2016 and 2020 ($F_{4,36} = 30.6$, $P < 0.001$; Figure 1A). However,

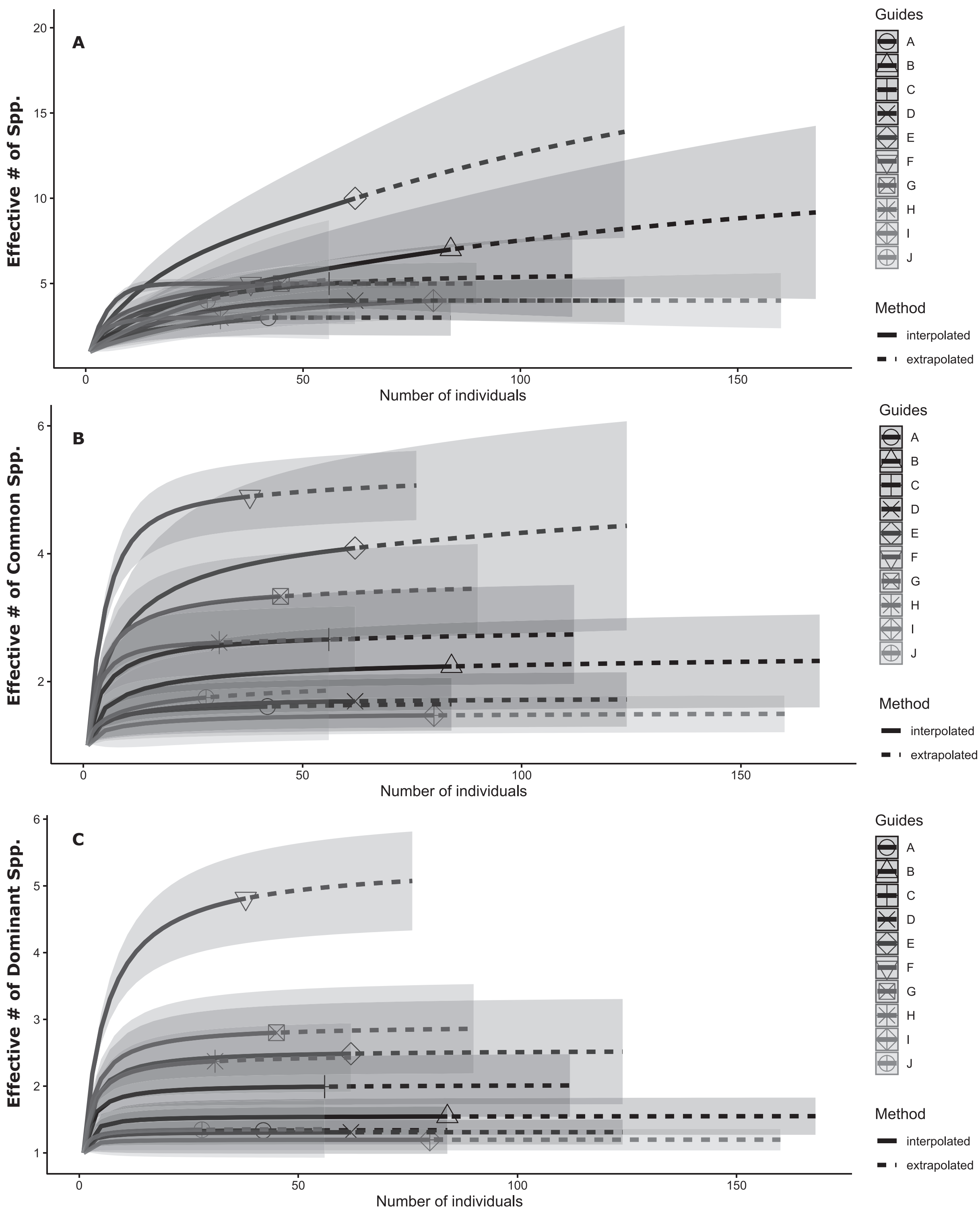

Figure 3.—Rarefaction curves for (A; upper plot) species richness, (B; middle plot) the effective number of common species, estimated as the exponentiated Shannon index, and (C; lower plot) the effective number of dominant species, estimated as the inverse of Simpson index, in removal plots where *L. maackii* was clipped yearly from 2016 to 2020. In each figure, the lines represent separate estimates for each of the 10 plots (A–J), and the gray shaded areas represent 95% confidence intervals.

while the number of species increased in that time period, we did not see an increase in the number of species in the final year ($F_{4,36} = 19.47$, $P < 0.001$; Figure 1B). There was also no increase in the effective number of common species (exponentiated Shannon's index; $F_{3,27} = 0.83$, $P = 0.487$; Figure 2A) or dominant species (inverse Simpson's index; $F_{3,27} = 1.67$, $P = 0.198$; Figure 2B) between 2017 and 2020. Due to the extremely low number of non–*L. maackii* individuals present in 2016, the effective number of common or dominant species was very low. This trend continued through 2020 (Figure 2), indicating very low diversity for all years. Furthermore, we did not observe any canopy tree seedlings in any of our removal plots, despite the presence of neighboring trees (Supplemental Table S2) and evidence of two common tree species in the seed bank (Supplemental Table S3). These results suggest that simply removing *L. maackii* from areas where it has become well established is unlikely to create small patches that may serve as new sources of native plants for future restoration efforts or significantly aid in the regeneration of dominant tree species at this site.

Although the number of individual plants increased, each plot contained only a small number of species. Rarefaction curves (Figure 3) indicate that these differences are unlikely to be due to low sampling effects, but rather to the dominance of a very small number of species in each plot (primarily one or two species). While the identity of dominant individuals between removal plots varied, almost all these species were early successionals, aggressive native species, or invasive species (Supplemental Table S4). Further, the overall diversity in our plots was far lower than observed in other studies (e.g., Luken et al. 1997; Runkle et al. 2007; Boyce 2015; Shields et al. 2015). Of particular concern, four of the ten removal plots were dominated by either *Asimina triloba* or *Carex* spp., two species known to slow the establishment of other native plant species (Baumer and Runkle 2010). In Michigan, *Carex* spp. have been noted to form dense mats that prevent the establishment of saplings in jack pine (*Pinus banksiana*) stands following disturbance (Abrams et al. 1985) and in riparian habitats following mortality of ash trees (*Fraxinus* spp.) due to the emerald ash borer (Engelken et al. 2020), functionally forming sedge meadows with open canopies. *Asimina triloba* has similarly been observed to interfere with tree regeneration (Baumer and Runkle 2010), particularly in areas where deer are common.

The negative impacts of *L. maackii* on native communities are well established (Collier et al. 2002; White et al. 2014; McNeish and McEwan 2016; Chen and Matter 2017; Sena et al. 2021). Removal efforts are clearly necessary and have been fairly successful in other studies (Hopfensperger et al. 2019). However, if our aim is to restore habitats where *L. maackii* has impacted the community for an appreciable amount of time and to maintain community participation, active restoration may be necessary. Luken et al. (1997) indicated that active management would be required following initial restoration efforts, particularly to prevent the reestablishment of *L. maackii* and other invasives. In this study, *L. maackii* was actively removed every season for 5 y (2016–2020). These efforts were easy to maintain and achieved marked reductions in *L. maackii* canopy cover, but we observed a considerably lower species diversity than by Luken et al. (1997) and no canopy tree species regeneration. In another 5 y study, Dolan and Brown (2019) found an increase in diversity but no increase in floristic quality index values following honeysuckle removal in urban riparian forests. Their removal efforts also relied on a large, community effort rather than the relatively modest effort that could feasibly be performed by a small landowner. In addition, the low native plant diversity and high proportion of "weedy species" observed in this study are likely to be perceived as failures by small landowners, leading to a decreased likelihood of participation (Zahawi et al. 2014; Arsénio et al. 2020; Höhl et al. 2020). Characteristics of the site used in this study may limit the scope of our conclusions. The infestation at our site was very well established, with *L. maackii* occupying the site for at least 40 y. Sporadic removal of *L. maackii* in the past has also created a mosaic of patches across a landscape that includes steep slopes and areas of intermittent flooding. Our results indicate that active restoration beyond continuous suppression of invasives will likely be necessary in areas where *L. maackii* has become well established. However, our success in reducing *L. maackii* in localized patches suggest that a passive restoration approach may be successful at sites with relatively contiguous populations.

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LITERATURE CITED

Abrams, M.D., D.G. Sprugel, and D.I. Dickmann. 1985. Multiple successional pathways on recently disturbed jack pine sites in Michigan. Forest Ecology and Management 10(1–2):31–48.

Arsénio, P., P.M. Rodríguez-González, I. Bernez, F. S. Dias, M.N. Bugalho, and S. Dufour. 2020. Riparian vegetation restoration: Does social perception reflect ecological value? River Research and Applications 36:907–920.

Bates, D., M. Mächler, B. Bolker, and S. Walker. 2015. Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67(1):1–48. <https://doi.org/10.18637/jss.v067.i01>

Baumer, M., and J.R. Runkle. 2010. Tree seedling establishment under the native shrub, *Asimina triloba*. Castanea 75:421–432.

Boyce, R.I. 2015. Recovery of native plant communities in southwest Ohio after *Lonicera maackii* removal. Journal of the Torrey Botanical Society 142:193–204.

Cavender-Bares, J., J.P. Cubino, W.D. Pearse, S.E. Hobbie, A.J. Lange, S. Knapp, and K.C. Nelson. 2020. Horticultural availability and homeowner preferences drive plant diversity and composition in urban yards. Ecological Applications 30(4):e02082.

Chao, A., N.J. Gotelli, T.C. Hsieh, E.L. Sander, K.H. Ma, R.K. Colwell, and A.M. Ellison. 2014. Rarefaction and extrapolation with Hill numbers: A framework for sampling and estimation in species diversity studies. Ecological Monographs 84(10):45–67.

Chen, H., and S.F. Matter. 2017. Quantification of changes in light and temperature associated with invasive Amur honeysuckle (*Lonicera maackii*). American Midland Naturalist 177:143–152.

Cipollini, K., E. Ames, and D. Cipollini. 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 2:45–54.

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

Dolan, R.W., and K.H. Brown. 2019. Five-year response of spontaneous vegetation to removal of invasive Amur bush honeysuckle along an urban creek. Urban Naturalist 26:1–17.

Dongwen, L., S. Ganesh, and J. Koolaard. 2018. Predicted means for linear and semiparametric models. <https://cran.r-project.org/package=predictmeans>

Engelken, P.J., M.E. Benbow, and D.G. McCullough. 2020. Legacy effects of emerald ash borer on riparian forest vegetation and structure. Forest Ecology and Management 457:117684.

Forbes, A.S., R.B. Allen, J.W. Herbert, K. Kohiti, W.B. Shaw, and L. Taurua. 2021. Determining the balance between active and passive indigenous forest restoration after exotic conifer plantation clear-fell. Forest Ecology and Management 479:118621.

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

Höhl, M., V. Ahimbisibwe, J.A. Stanturf, P. Elsasser, M. Kleine, and A. Bolte. 2020. Forest landscape restoration—What generates failure and success? Forests 11(9):938.

Hopfensperger, K.N., R.L. Boyce, and D. Schenk. 2019. Potential reinvasion of *Lonicera maackii* after urban riparian forest regeneration. Ecological Restoration 37:25–33.

Hsieh, T.C., H. Ma, and A. Chao. 2022. Interpolation and extrapolation for species diversity. <http://chao.stat.nthu.edu.tw/wordpress/software_download/>

Kilgore, M.A., and S.A. Snyder. 2016. Exploring the relationship between parcelization metrics and natural resource managers' perceptions of forest land parcelization intensity. Landscape and Urban Planning 149:44–48.

Krasny, M.E., and K.G. Tidball. 2012. Civic ecology: A pathway for Earth stewardship in cities. Frontiers in Ecology and the Environment 10(5):267–273.

Kuznetsova, A., P.B. Brockhoff, and R.H.B. Christensen. 2017. lmerTest package: Tests in linear mixed effects models. Journal of Statistical Software 82(13):1–26. <https://doi.org/10.18637/jss.v082.i13>

Lenth, R.V., B. Bolker, P. Buerkner, I. Giné-Vázquez, M. Herve, M. Jung, J. Love, F. Miguez, H. Riebl, and H. Singmann. 2017. Estimated marginal means, aka least-squares means. <https://cran.r-project.org/package=emmeans>

Little, M.N., K.W. Custer, E.B. Borth, J.I. Chapman, M.J. Kukla, C. Kuminecz, M.E. Maloney, M.J. Woods, and R.W. McEwan. 2021. The influence of riparian invasion by the terrestrial shrub *Lonicera maackii* on aquatic macroinvertebrates in temperate forest headwater streams. Biological Invasions 23:25–35.

Luken, J.O., L.M. Kuddes, and T.C. Tholmeier. 1997. Response of understory species to gap formation and soil disturbance in *Lonicera maackii* thickets. Restoration Ecology 5:229–235.

Mayer, A.L. 2019. Family forest owners and landscape-scale interactions: A review. Landscape and Urban Planning 188:4–18.

McNeish, R.E., and R.W. McEwan. 2016. A review of the invasion ecology of Amur honeysuckle (*Lonicera maackii*, Caprifoliaceae) a case study of ecological impacts at multiple scales. Journal of the Torrey Botanical Society 143:367–385.

Poulette, M.M., and M.A. Arthur. 2012. The impact of the invasive shrub *Lonicera maackii* on the decomposition dynamics of a native plant community. Ecological Applications 22:412–424.

Prach K., L. Šebelíková, K. Řehounková, and R. del Moral. 2020. Possibilities and limitations of passive restoration of heavily disturbed sites. Landscape Research 45:247–253.

R Core Team. 2020. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Rohr, J.R., E.S. Bernhardt, M.W. Cadotte, and W.H. Clements. 2018. The ecology and economics of restoration: When what, where, and how to restore ecosystems. Ecology and Society 23(2):15.

Roswell, M., J. Dushoff, and R. Winfree. 2021. A conceptual guide to measuring species diversity. Oikos 130:321–338.

Runkle, J.R., A. Disalvo, Y. Graham-Gibson, and M. Dorning. 2007. Vegetation release 8 years after removal of *Lonicera maackii* in west-central Ohio. Ohio Journal of Science 107:125–129.

Schulz, K.E., J. Wright, and S. Ashbaker. 2012. Comparison of invasive shrub honeysuckle eradication tactics for amateurs: Stump treatment versus regrowth spraying of *Lonicera maackii*. Restoration Ecology 20:788–793.

Sena, K., L. Gauger, T. Johnson, F. Shirkey, W. Caldbeck, J. Hammock, A. Kim, B. Mazza, I. Pethtel, and W. Leuenberger. 2021. Honeysuckle (*Lonicera maackii*) presence is associated with reduced diversity and richness of flowering spring flora in central Kentucky. Natural Areas Journal 41:249–257.

Shields, J.M., M.R. Saunders, K.D. Gibson, P.A. Zollner, J.B. Dunning, and M.A. Jenkins. 2015. Short-term response of native flora to the removal of non-native shrubs in mixed-hardwood forests of Indiana, USA. Forests 6:1878–1896.

White, R.J., M.M. Carrerio, and W.C. Zipperer. 2014. Woody plant communities along urban, suburban, and rural streams in Louisville, Kentucky, USA. Urban Ecosystems 17:1061–1094.

Zahawi, R.A., J.L. Reid, and K.D. Holl. 2014. Hidden costs of passive restoration. Restoration Ecology 22:284–287.