# Effects of Long-Term Manual Invasive Plant Removal on Forest Understory Composition

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RESEARCH ARTICLE

# Effects of Long-Term Manual Invasive Plant Removal on Forest Understory Composition

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**ABSTRACT**: Removal of invasive species is a common management goal to maintain native species composition and wildlife habitat. Due to the time and effort necessary to remove invasive species, it is important to clearly understand the benefits that will be gained through removal and what methods will best achieve those results. This study evaluated the response of native plant understory communities to the removal of invasive species that fell into a range of functional groups including perennial herbs (*Microstegium vimineum, Liriope muscari*), vines (*Lonicera japonica, Lygodium japonicum, Hedra helix*), shrubs (*Ligustrum sinense*), and trees (*Albizia julibrissin, Triadica sebifera*). Eight invasive plant species were removed from twenty-seven 1-m$^2$ plots for 8 y in an upland mixed hardwood–pine and riverine woodland within the Ocmulgee National Monument, Macon, Georgia. Species richness, herbaceous cover, and woody species number was measured 2 y before removal and each year during removal. Mechanical removal reduced invasive species richness, cover, and number, however all measures of native species diversity remained unchanged. Overall, common species remained common but there was some turnover in less common species over the 8 y. During the study period, the area experienced an exceptional drought and it is likely that native species recovery after invasive species removal was hindered by these extreme weather conditions. Invasive species may be a determinant of native species composition, but environmental factors like drought may be a more important determining factor.

*Index terms:* eradication, forest understory, invasive species, long-term research, mechanical control

## INTRODUCTION

Protected areas, such as parks and preserves, within human-dominated landscapes can provide safe havens for biodiversity. Although the threat of development might be reduced, these areas can still be vulnerable to invasive plant invasion. A total of 3756 unique nonnative plant species has been recorded in US National Parks; these plants cover 7.3 million hectares (Allen et al. 2009). Therefore, removal of invasive species is a common management goal to maintain native species composition and wildlife habitat, even in protected areas (Beard and App 2012).

Removal of invasive species not only plays a role in the protection of biodiversity, it can also help scientists better understand their function in suppressing natives and possibly altering the long-term successional trajectories of communities (Runkle et al. 2007; Aronson and Handel 2011). Due to the time and effort necessary to remove invasive species, it is important to clearly understand the benefits that will be gained through removal and what methods will best achieve those results (Abella 2014; Taggart et al. 2015).

Several means are available for removal of invasive plant species including physical removal, chemical removal, and biological control, each with their own cost in time and money and level of effectiveness. Physical removal offers the most straightforward approach. Physical removal removes most if not all parts of the plant and should not greatly impact nontarget plants (Flory and Clay 2009). By disturbing the soil and increasing sunlight, temperature, and nutrient leaching, removal could improve native plant germination (Biggerstaff and Beck 2007). Physical removal is particularly preferable when working with a volunteer base that does not have the expertise to use chemical or biological control (Freshwater 1991). The disadvantages include the high time commitment, which may limit this technique to small or satellite populations (Chapman et al. 2012). The same physical disturbance and increased light availability that can increase native plant germination can also promote recolonization by invasives or increase seed germination of invasives from the seed bank (D’Antonio and Meyerson 2002; Hulme and Bremner 2006; Flory and Clay 2009). If roots are left in the soil, the threat of recolonization from resprouting remains (Freshwater 1991).

In addition to the practical benefits of using hand removal, several studies have found that, when comparing this method to chemical methods, the overall result is a better rebounding of native species (Barto and Cipollini 2009). Removing by hand resulted in increased species richness and species diversity in similar forests whereas herbicide applications resulted in no change (Biggerstaff and Beck 2007). Seed addition was also more successful in hand-removal plots indicating the importance of the disturbance in seed germination (Biggerstaff and Beck 2007).

The purpose of this study was to measure the long-term effectiveness of repeated physical removal of multiple invasive plant species. Although there are several studies that look at the effectiveness of physical removal of invasive species (McCarthy 1997; Hulme and Bremner 2006; Vidra et al. 2007; Hanula et al. 2009; Jäger and Kowarik 2010; Beasley and McCarthy 2011; Chapman et al. 2012; Emery et al. 2013; MacDonald et al. 2013), few track the community beyond a year or two (Runkle et al. 2007; Hudson et al. 2014) or investigate the removal of multiple invasive species of different functional groups. This research investigates the effectiveness of mechanical removal in eradicating several invasive species including trees, shrubs, vines, and herbaceous plants and takes place over a 10-y time period.

The diversity of invasive species at the study site represents a range of growth habits that allow for a unique comparison of hand removal effectiveness. The eight invasive species present include *Albizia julibrissin*, *Hedra helix*, *Ligustrum sinense*, *Liriope muscari*, *Lonicera japonica*, *Lygodium japonicum*, *Microstegium vimineum*, and *Triadica sebifera*. *Microstegium vimineum* and *Liriope muscari* represent the only two herbaceous species. *Microstegium vimineum* (Trin.) A. Camus (Japanese stilt grass) is an annual grass that that is shade tolerant and a prolific seed producer with a persistent seed bank (Gibson et al. 2002). *Liriope muscari* (Decne.) L.H. Bailey (big blue lilyturf) is an evergreen grass-like perennial that forms clumping mounds and spreads by rhizomes (Franz 2008). The most common growth type is vines including *Lonicera japonica*, *Lygodium japonicum*, and *Hedra helix*. *Lonicera japonica* Thunb. (Japanese honeysuckle) is a woody trailing vine that densely covers the ground surface (Schierenbeck 2004). *Lygodium japonicum* (Thunb.) Sw. is a species of climbing fern common in moist woods and riparian habitats that can smother understory vegetation (Lott et al. 2003). *Hedra helix* L. (English ivy) is woody vine that not only forms a very dense ground cover, but also climbs and weakens trees (Thomas 1980). *Ligustrum sinense* Lour. (Chinese privet) is an evergreen shrub that forms impenetrable monocultural stands and shades out herbaceous plants (Maddox et al. 2010). Although overall uncommon, there are two invasive trees present in the site. *Albizia julibrissin* Durazz (mimosa) and *Triadica sebifera* (L.) (Chinese tallow) are prolific seed producers that readily germinate (Dirr 2009). All nonnative species found in the field site are native to Asia and are shade tolerant enabling them to invade relatively intact forests (Gibson et al. 2002; Franz 2008; Dirr 2009).

## MATERIALS AND METHODS

The Ocmulgee National Monument is located along the Ocmulgee River in central Georgia (Bibb County) at the convergence of the piedmont and coastal plain. The park is within the city of Macon, an urban center with a population of close to 90,000 (US Census Bureau 2015). The park protects significant archeological sites including nine earthen mounds associated with two Native American Mississippian cultures (Hally 1994). Within the 283.9-ha park, I used a 5-ha portion of upland mixed hardwood–pine and riverine woodland bounded in the north by a park road, in the west by a trail, and to the south and east by Walnut Creek, a tributary of the Ocmulgee River. The canopy of the forest is dominated by mixed hardwoods, including *Liquidambar styraciflua*, *Carya* spp., *Magnolia grandiflora*, and *Quercus* spp., and pine (*Pinus tadea)* (Zomlefer et al. 2013).

Within this forest, 27 permanent plots were marked with plastic survey stakes along four parallel transects stretching from the road to Walnut Creek. Transects were 30 m apart and plots along the transect were also 30 m apart. Starting in May 2006, I surveyed the herbaceous and woody understory vegetation within a 1-m$^2$ quadrat at each plot marker. Each quadrat was aligned on a compass direction to enable resampling at the same location each year. The percent cover for herbaceous species and vines was visually estimated and the number of each woody species (small shrubs and woody seedlings) was recorded. Given the small size of the plot, all woody species were either small shrubs or seedlings. Each plot was surveyed in late May for 10 years. The floras of Radford et al. (1968) and Weakley (2011) were primary sources for plant identification.

Starting in 2008 and continuing each year after during annual surveys of permanent plots, all invasive species in each plot were collected by hand, being careful to remove as much root from the soil as possible. Each species per plot was separately bagged, dried, and weighed.

Importance values were calculated for each species each year. For herbaceous plants, the importance value was calculated as the relative cover and relative frequency for each species each year. For small shrub and tree seedlings, relative stem density and relative frequency were used to calculate importance value. Plants that were both common and abundant had an importance value close to 1. The dependent variables of native and invasive species number (frequency count), herbaceous cover (% area), and stem number were analyzed by one-way ANOVAs with one repeated-measures factor with year of monitoring as the within-subjects factor (independent variable) using JMP (Version Pro 11, SAS Institute Inc.). The repeated-measure ANOVA allowed for each year to be compared to initial conditions (year 2006). Based on a significant departure from sphericity and a Greenhouse–Geisser epsilon value below 0.75 for each data set, multivariate *F* tests were used to determine whether measures of diversity changed over time when compared to initial conditions in 2006 (Lehman et al. 2013).

## RESULTS

A total of 48 species was recorded in the 27 permanent plots over 10 y of sampling (Table 1). In any one year, there was an average of 22.6 (± 1.53) species recorded. The most common native species throughout the plots were perennial vines including *Parthenocissus quinquefolia*, *Smilax* sp., *Toxicodendron radicans*, and *Vitis rotundifolia.* These vines were consistently found in the same plots each year whereas most other plants were more variable (i.e., pres-

**Table 1. Importance values for each species each year. An importance value close to 1 indicates that the species was common and abundant in the 27 permanent plots sampled (* indicates invasive species; gray fill indicates years following initial removal of invasive species in 2008; underlined importance values indicate highest value for each year in each functional group).**

| Species | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|---|---|---|---|---|
| **Woody plants** | | | | | | | | | | |
| *Acer rubrum* | 0.27 | 0 | 0.14 | 0.16 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Albizia julibrissin* * | 0.08 | 0 | 0 | 0.07 | 0 | 0 | 0.06 | 0.12 | 0 | 0 |
| *Carya* sp. | 0 | 0 | 0 | 0 | 0 | 0 | 0.07 | 0 | 0 | 0 |
| *Celtis laevigata* | 0 | 0.06 | 0.29 | 0.14 | 0.29 | 0.23 | 0.17 | 0.2 | 0.29 | 0.05 |
| *Euonymus americanus* | 0 | 0 | 0 | 0.2 | 0 | 0 | 0.06 | 0 | 0 | 0.05 |
| *Ligustrum sinense* * | <u>0.82</u> | <u>0.66</u> | <u>0.63</u> | <u>0.85</u> | 0.57 | <u>0.79</u> | <u>0.91</u> | 0.08 | <u>0.54</u> | 0.37 |
| *Liquidambar styraciflua* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.07 |
| *Pinus taeda* | 0.1 | 0.06 | 0.21 | 0.05 | <u>0.97</u> | 0.13 | 0.06 | 0.08 | 0 | <u>0.61</u> |
| *Prunus caroliniana* | 0 | 0.06 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 | 0 |
| *Quercus* sp. | 0.4 | 0.6 | 0.32 | 0.24 | 0.23 | 0 | 0.11 | 0.16 | 0.29 | 0.11 |
| *Triadica sebiferum* * | 0.05 | 0 | 0.11 | 0 | 0 | 0.09 | 0.13 | 0.08 | 0 | 0 |
| *Sassafras albidum* | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0.08 | 0 | 0.16 |
| *Ulmus alata* | 0.14 | 0.32 | 0 | 0 | 0.09 | 0.07 | 0 | 0 | 0 | 0 |
| Unknown 1 | 0 | 0.07 | 0 | 0.11 | 0 | 0.07 | 0 | <u>0.51</u> | 0 | 0.14 |
| Unknown 2 | 0 | 0 | 0.07 | 0.05 | 0 | 0 | 0 | 0.08 | 0 | 0.38 |
| **Herbs and ferns** | | | | | | | | | | |
| *Chimaphila maculata* | 0 | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 |
| *Erechtites hieraciifolia* | 0 | 0 | 0 | 0.08 | 0.12 | 0 | 0 | 0 | 0 | 0 |
| *Galium* sp. | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | <u>0.06</u> | 0 |
| *Impatiens capensis* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.04 |
| *Liriope muscari* * | 0 | 0 | 0 | 0.08 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Polygonum* sp. | 0 | 0 | 0 | 0.09 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stellaria media* | 0 | 0 | <u>0.11</u> | <u>0.14</u> | 0.09 | <u>0.05</u> | 0 | 0 | 0.05 | 0 |
| *Viola* sp. | <u>0.04</u> | <u>0.04</u> | 0.04 | 0.04 | <u>0.13</u> | <u>0.05</u> | 0 | <u>0.04</u> | <u>0.06</u> | <u>0.1</u> |
| Unknown 1 | <u>0.04</u> | 0 | 0.08 | 0 | 0.04 | 0.04 | 0 | 0 | 0 | 0.04 |
| Unknown 2 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unknown 3 | 0 | 0 | 0 | 0.09 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unknown 4 | 0 | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 |
| Unknown 5 | 0 | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 |
| **Grasses** | | | | | | | | | | |
| *Anthraxon* sp. | 0.04 | 0 | 0.04 | <u>0.13</u> | 0.05 | 0.05 | 0.07 | 0.04 | 0.05 | <u>0.15</u> |
| *M. vimineum* * | <u>0.17</u> | 0.04 | <u>0.2</u> | 0.1 | <u>0.28</u> | <u>0.29</u> | 0 | 0 | 0.1 | 0.09 |
| *Poa cuspidata* | 0 | 0 | 0.09 | <u>0.13</u> | 0.19 | 0 | 0 | 0 | 0 | 0 |
| *Dicanthiclium* sp.1 | 0.05 | 0 | 0 | 0 | 0.04 | 0 | 0.07 | 0.06 | 0 | 0.05 |
| *Dicanthiclium* sp.2 | 0.04 | 0.1 | 0 | 0.05 | 0.05 | 0.05 | <u>0.13</u> | <u>0.23</u> | 0 | 0 |
| Unknown grass 1 | 0.08 | <u>0.2</u> | 0.04 | 0.04 | 0 | 0.19 | 0.07 | 0.18 | <u>0.14</u> | 0.13 |
| Unknown grass 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Unknown grass 3 | 0 | 0 | 0 | 0 | 0.04 | 0 | 0 | 0 | 0 | 0 |

*Continued*

**Table 1. (Cont'd.)**

| Species | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|---|---|---|---|---|
| **Vines** | | | | | | | | | | |
| *Berchemia scandens* | 0 | 0 | 0.08 | 0.09 | 0.08 | 0.09 | 0 | 0 | 0 | 0.05 |
| *Bignonia capreolata* | 0.12 | 0.13 | 0.13 | 0.1 | 0.16 | 0.15 | 0.18 | 0.14 | 0.16 | 0.16 |
| *Campsis radicans* | 0.14 | 0.08 | 0.04 | 0.11 | 0.06 | 0.06 | 0.07 | 0 | 0.09 | 0.07 |
| *Hedera helix* * | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.04 |
| *Lonicera japonica* * | 0.71 | 0.79 | 0.68 | 0.37 | 0.25 | 0.13 | 0 | 0 | 0 | 0 |
| *Lygodium japonicum* * | 0.04 | 0.04 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Mikania scandens* | 0 | 0 | 0 | 0 | 0.04 | 0.05 | 0 | 0 | 0 | 0 |
| *P. quinquefolia* | 0.6 | 0.59 | 0.75 | 0.7 | 0.59 | 0.63 | 0.35 | 0.54 | 0.54 | 0.69 |
| *Passiflora lutea* | 0.04 | 0.08 | 0.08 | 0.09 | 0.1 | 0 | 0.05 | 0 | 0 | 0 |
| *Rubus argutus* | 0.23 | 0.22 | 0.09 | 0.22 | 0.17 | 0.09 | 0.19 | 0.17 | 0.13 | 0.14 |
| *Smilax* sp. | 0.41 | 0.4 | 0.4 | 0.38 | 0.48 | 0.35 | 0.46 | 0.36 | 0.05 | 0.48 |
| *Toxicodendron radicans* | 0.73 | 0.64 | 0.76 | 0.76 | 0.71 | 0.6 | 0.66 | 0.33 | 0.44 | 0.65 |
| *Vitis rotundifolia* | 0.91 | 0.92 | 0.8 | 0.74 | 0.53 | 0.48 | 0.58 | 0.67 | 0.57 | 0.71 |

ent one year and absent the next). These species had importance values of 0.60 and above most years. No herbaceous species or grass had an importance value higher than 0.29. Even after years of removal, *Ligustrum sinense* maintained high importance values (generally above 0.6), often higher than any other woody species. The most significant year to year change came from the presence and absence of tree seedlings. In 2010, *Pinus* seedlings had an importance value of 0.97, but dropped to 0.13 next year. Although there are some changes in the community after invasive species removal, in general, common species maintain a similar frequency before and after removal. New species may appear after invasive removal, but none become common or maintain frequency over the years. The total number of species from the beginning of the study (24) was the same after 8 y of invasive species removal. Of those initial 24 species, 16 (or 67%) were found in plots in 2015.

In 2006, there were six invasive species present in the 27 permanent plots: *Albizia julibrissin*, *Ligustrum sinense*, *Lonicera japonica*, *Lygodium japonicum*, *Microstegium vimineum*, and *Triadica sebifera* (Table 1). In that year, 17 plots were invaded, four by more than one species. In 2007, *Triadica sebifera* and *Albizia julibrissin* were no longer present and 18 plots were invaded, four by more than one species. However, which plots were invaded shifted slightly from one year to the next. Two plots that were invaded in 2006 were not invaded in 2007. Additionally, two plots became newly invaded in 2007. In 2008, when removal of invasives began, 16 plots were invaded and four species were present (Table 2). *Lygodium japonicum* was no longer present and did not reappear during the study while *T. sebifera* reemerged. In 2009, a year after first removal, *T. sebifera* was no longer present, but the other five invasive species were present with only a small decrease in the number of plots invaded. The five species present included the only appearance of *Liriope muscari*. *L. sinense* was found in one more plot than before removal. In 2010, the presence of *L. sinense* and *L. japonica* decreased, but the presence of *M. vimineum* increased from two plots to four. In 2011, *T. sebifera* returned, *L.*

**Table 2. The total dry weight (grams) of invasive species collected in sampling plots.**

| | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 | 2015 |
|---|---|---|---|---|---|---|---|---|
| *Albizia julibrissin* | 0 | 0.08 | 0 | 0 | 0.018 | 0.053 | 0 | 0 |
| *Hedera helix* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.076 |
| *Ligustrum sinense* | 163.04 | 4.28 | 1.04 | 3.18 | 1.23 | 2.21 | 0.494 | 0.815 |
| *Liriope muscari* | 0 | 0.23 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Lonicera japonica* | 98.31 | 7.78 | 1.56 | 2.31 | 0 | 0 | 0 | 0 |
| *Microstegium vimineum* | 2.48 | 6.9 | 12.89 | 33.42 | 0 | 0 | 0.516 | 1.03 |
| *Triadica sebifera* | 0.12 | 0 | 0 | 0.444 | 0 | 0.071 | 0 | 0 |
| Total | 263.95 | 19.27 | 15.49 | 39.35 | 1.25 | 2.33 | 1.01 | 1.92 |

*sinense* showed a slight increase in number of plots present, while *M. vimineum* stayed the same and *L. japonica* continued to decrease. However, in 2012, after 4 y of removal, there was a dramatic decrease in all invasives except *L. sinense* as well as the appearance of a new invasive tree seedling, *Albizia julibrissin*. The number of invaded plots stayed low in 2013, but *T. sebifera* returned. In 2014, seedlings of *L. sinense* continued to be found in low numbers while *M. vimineum* reemerged in two plots. In the final year of the study, only three plots continued to be invaded by *M. vimineum* and *Ligustrum sinense.* One of those three also contained *Hedera helix*, the first appearance of this invasive woody vine. All invaded plots were within close proximity to each other and were disturbed by flooding of the creek that bounds the study area to the south and east.

By looking at the total weights of the invasive species, a more complete picture emerges (Table 2). In 2008, *L. sinense* and *L. japonica* had the highest initial weights. These weights dropped dramatically in 2009 indicating only the presence of seedlings and regrowth from roots after initial removal. The weight for *L. sinense* fluctuated based on the number and size of seedlings present. *L. japonica* continued to be a small presence in 2010 and 2011 before no longer being present in the sample plots. *M. vimineum* fluctuated in terms of presence and weight throughout the years and seems the least influenced by eradication measures. Both *A. julibrissin* and *T. sebifera* showed up only occasionally as seedlings and in association with plots that have adult plants nearby. A single seedling of *Hedera helix* appeared in one plot in the last year of sampling.

Initially, there was a positive trend in native species richness as invasive species were removed with the highest average, 3.52 (± 0.386) native species per plot, being observed in 2010, 2 y after initial removal (Figure 1). However, that trend started to reverse starting in 2011 as the number of native species dropped each year with 2012, 2013, and 2014 being significantly lower than the initial native species number in 2006. The lowest number of natives was recorded in 2014 (1.59 ± 0.263) even though the number of invasive species was also at its lowest (0.11 ± 0.061). In 2015, the number of natives rebounded to initial levels with 3.33 ± 0.456 species per plot. As expected, the number of invasive species per plot decreased with each successive removal, although it took 2 y for a significant decrease. Invasive species were still found in three plots after 8 y of removal, but at a significantly lower level than at the beginning of the study.

Native herbaceous percent cover showed a downward trend with cover being significantly lower than initial in 2008 (year of invasive removal) and 2011–2012 (Figure 2). This included a 72% drop in one common native, *Vitus rotundifolia*. As expected, the cover of invasives decreased as they were removed, however, it took 4 y of removal before cover values were significantly below initial surveys. While invasive cover remained low (with only *M. vimineum* remaining), native cover increased with 2015 reaching total cover values near 2006, before invasives were removed.

Due to the small size of the sampling plots, woody abundance is highly influenced by the presence and absence of seedlings. By counting number, but not indicating size, it appears that woody plants (native and invasive) greatly increase after removal (Figure 3). Instead, what this data captures is the emergence of seedlings after removal of large *L. sinense* individuals. The fact that they are seedlings is also reflected in the small amount of biomass those years (Table 2). In 2010, a significantly higher number of native seedlings were dominated by *Pinus taeda*.

Although this study set out to record changes in community composition after invasive species removal, other factors can play a role when conducting an experiment in field conditions. Over the 10 y of the study, Macon, Georgia, experienced several years of drought. Starting in May of 2011 and lasting until March of 2013, most, if not all, of Bibb County was in a severe drought. Between April 2012 and February 2013, Bibb County fell into the category of exceptional drought, the US Drought Monitor's worst classification for extended periods of low rainfall and dried out soil (NDMC 2015). These periods are evident in the monthly total rainfall records for Macon (Figure 4). These periods of low rainfall correspond to several changes in community composition including a significant decrease in woody species abundance, native herbaceous cover, and native species number.

## DISCUSSION

The long-term study of native plant recov-

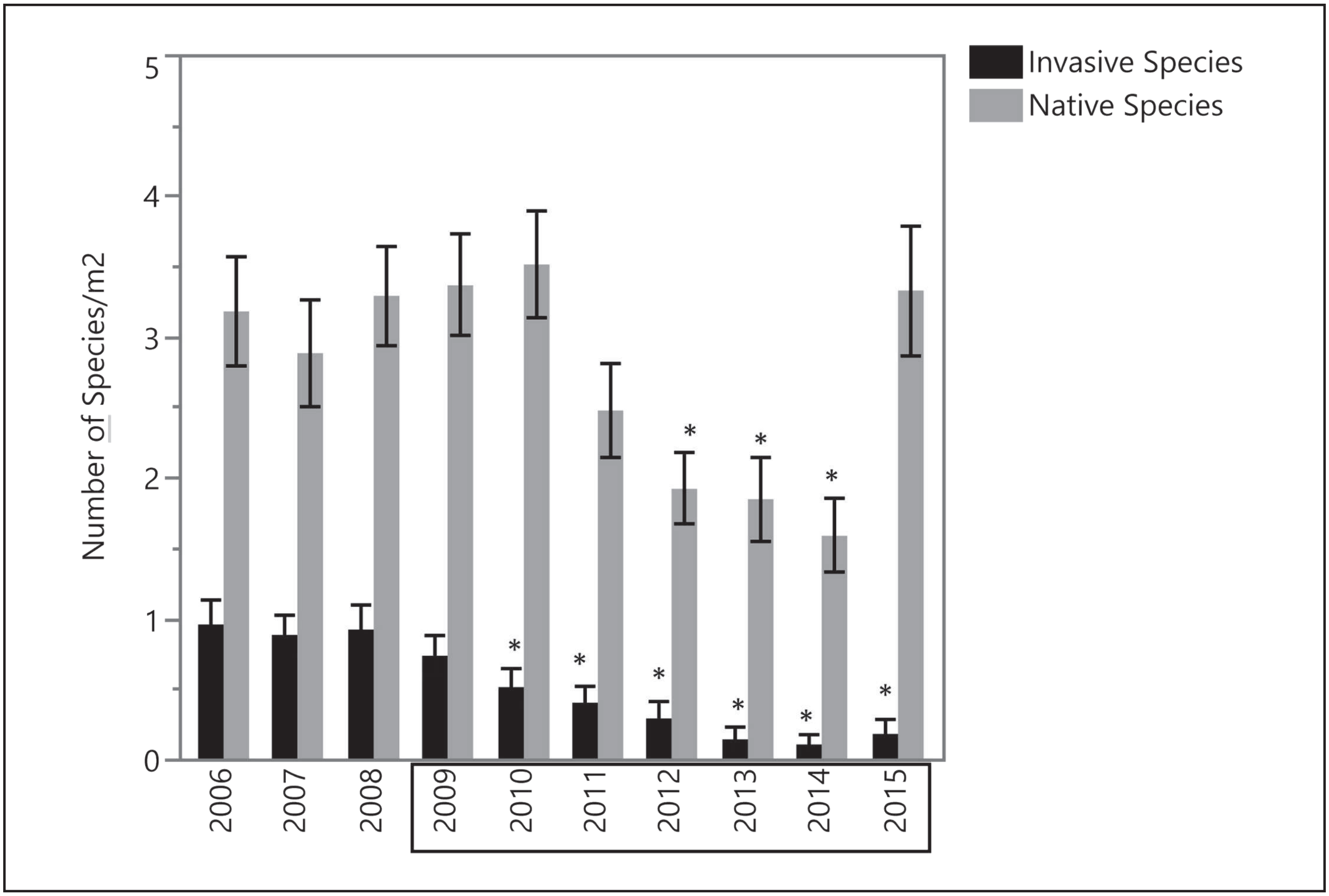


**Figure 1. Average number of native and invasive species per plot. Results analyzed using an ANOVA repeated-measures design revealed a significant difference between initial conditions (2006) and 2010 ($F_{1,26}$ = 4.872; $p$ = 0.0369), 2011 ($F_{1,26}$ = 11.6071; $p$ = 0.0021), 2012 ($F_{1,26}$ = 14.1818; $p$ = 0.0009), 2013 ($F_{1,26}$ = 17.8750; $p$ = 0.0003), 2014 ($F_{1,26}$ = 29.2638; $p$ < 0.0001), and 2015 ($F_{1,26}$ = 15.9250; $p$ = 0.0005) for invasive species number. For native species, there was only a significant change in number of species in 2012 ($F_{1,26}$ = 9.4994; $p$ = 0.0048), 2013 ($F_{1,26}$ = 9.0435; $p$ = 0.0058), and 2014 ($F_{1,26}$ = 15.0231; $p$ = 0.0006), likely a result of extreme drought conditions. In 2015, the number of native species returned to initial levels.**

ery and potential reinvasion after invasive plant removal is essential in understanding the effectiveness of removal protocols and justification of control efforts. Eight years of removal of invasive species from plots within a forest ecosystem resulted in very low occurrence of invasive species. Average cover of herbaceous invasives was reduced by 90.5% and biomass of invasives was reduced by 99.3%. The extent of invasion was also reduced with an 83.3% reduction in number of plots with an invasive species present and only three of eight recorded invasive species presence during the final survey. However, even with yearly effort, complete eradication was not achieved.

One of the detrimental effects of nonnative plant species is that they decrease diversity by suppressing or replacing native species (Flory and Clay 2009; Heida et al. 2009; Aronson and Handel 2011). In this study, when comparing the initial to the final survey, every measure of native species diversity remained unchanged. The average number of native species per plot was statistically the same as before removal (2006: 3.2; 2015: 3.3), as was the total number of native species recorded (2006: 17; 2015: 18). Native herbaceous cover and number or woody seedlings per plot also show little change at the end of the study (Figures 2 and 3). No new species became common as invasives were removed (Table 1). However, there was turnover in the identity of the species present in the beginning and end of the study. Of 24 species at the end of the study, eight were not present at the beginning of the study. Most of these new species were tree seedlings. Runkel et al. (2007) saw a similar increase in number of woody seedlings 8 y after the removal of *Lonicera maackii*. Although many of these seedlings do not survive year to year, their presence could have an important impact on forest regeneration.

Several studies have shown that increases in native species richness and abundance after invasive species removal does not become apparent until several years after removal (Luken et al. 1997; Runkle et al. 2007). There are several other possible reasons for the lack of expected recovery of the native community. First, invasive species do not necessarily affect survival of native plants, but instead simply limit

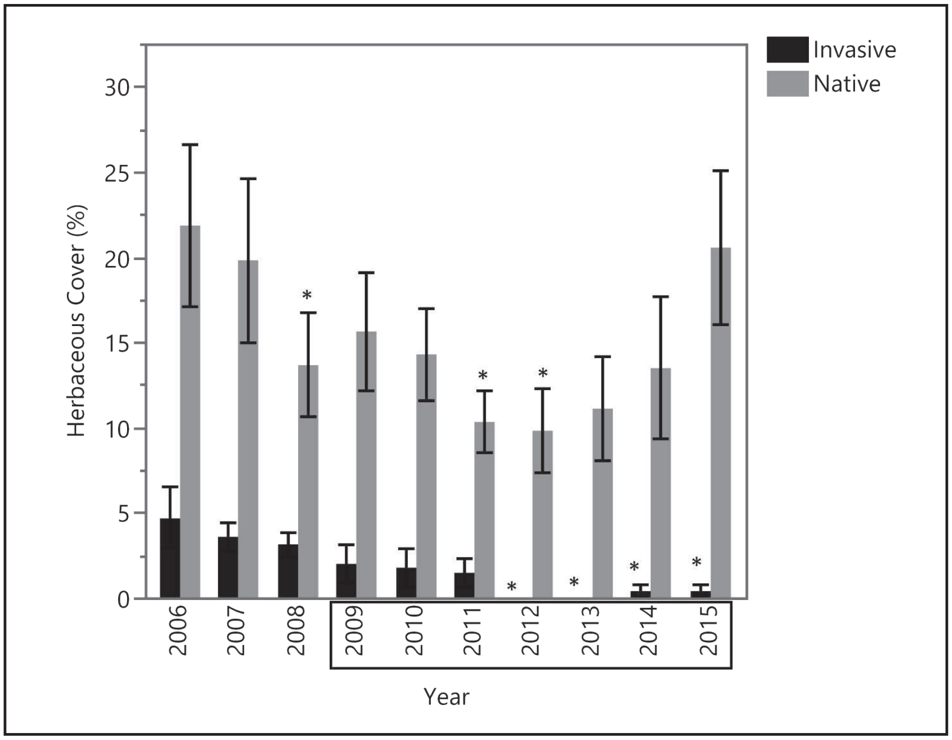


**Figure 2. Average herbaceous native and invasive species percent cover per plot. Herbaceous cover dropped significantly in 2008 ($F_{1,26}$ = 9.1040; $p$ = 0.0056), then again in 2011 ($F_{1,26}$ = 6.3564; $p$ = 0.0182) and 2012 ($F_{1,26}$ = 5.8479; $p$ = 0.0229). Invasive cover dropped each year following removal, but did not reach significantly lower levels until 2012 ($F_{1,26}$ = 6.6215; $p$ = 0.0161) when invasive cover reached zero for 2 y.**

their growth and fecundity by reducing resources (Miller and Gorchov 2004) or changing ecosystem properties (Ehrenfeld 2003; Corbin and D'Antonio 2012). It is possible that this study did not capture those subtler changes. Second, lack of change in native species abundance and diversity could be an indicator that invasives were filling niches unfilled by natives (Simberloff 1981). Third, though invasive species were common in the experimental plots (Table 1), native species were generally more common. Therefore, the invasive species present in this forest could be under a threshold that would inhibit growth of natives.

Although these other interpretations are possible, it is more likely that the return of natives was hindered by the onset of a significant drought during the study period. From 2011 to 2013, rainfall levels were at historic lows. Those same years saw the lowest levels of species diversity, herbaceous cover, and seedling number. Drought can initiate significant changes in community structure and composition (Tilman and El Haddi 1992; Yurkonis and Meiners 2006). Uncommon species were no longer present during the drought and some common species decreased considerably. For example, *Vitus rotundifolia* decreased by 72% while *Toxicodendron radicans* decreased by 65.4%. No new species established during the drought. Some invasives saw a greater drop in occurrence during the drought than with removal alone. In 2013, *L. sinense* had one of the lowest importance values of all woody species and *M. vimineum* was not growing in any plots. Normal rainfall returned in the spring of 2013, however tree seedlings saw their lowest abundance in 2014, which could indicate a lack of production during the drought. It was not until 2015 that the plant community showed signs of recovery. Invasives did not exhibit a rapid recovery from drought; the incidence of invasives remained quite low while natives increased in number and cover. This result agrees with research conducted in mesocosms that showed repeated drought can give natives an advantage over invasives (Meisner et al. 2013). Only one invasive, *A. julibrissin*, took advantage of the drought, but numbers remained low and did not reoccur after normal rainfall conditions returned. It remains to be seen whether recovery from the drought will continue and result in greater diversity in the absence of invasive species. Recovery could also be further slowed by lack of available native propagules (Vidra et al. 2007). Ocmulgee National Monument is surrounded by urban development and dispersal of new species into this forest fragment could be rare.

Each invasive species reacted somewhat differently to being removed. *Lonicera japonica* continued to resprout 3 y after the initial removal, with biomass decreasing until it reached zero. Once respouting was brought under control, there was no reoccurrence of this species indicating that hand removal could be an effective means of control if sprouts from roots are removed. Limited pollination of *Lonicera* flowers results in small seed set that reduces the probability of reinvasion from seed (Larson et al. 2002). The invasive trees, *Albizia julibrissin* and *Triadica sebifera*, relied on seeds of mature individuals within the vicinity of the research plot to reinvade. Therefore, if seed producers are removed, the likelihood of eradication is high. With *L. sinense*, the incidence of resprouting from roots and germination of seeds were observed, which meant that even with continued removal, the likelihood of reinvasion was high if seedlings were left to develop from the seed bank. The species that showed the least response to control measures was *M. vimineum*. Its presence or absence was more related to weather conditions than eradication measures. It was heavily impacted by the drought and was not found in any plots for 2 y, but returned when precipitation increased. Gibson et al. (2002) showed that flowering was restricted in *M. vimineum* during times of moisture stress, but since *M. vimineum* maintains a persistent soil seed bank, the plant was able to recover after the drought and maintain a presence.

In addition to differences between species, the location of the research plot also influenced probability of reinvasion. All plots that continued to be invaded well

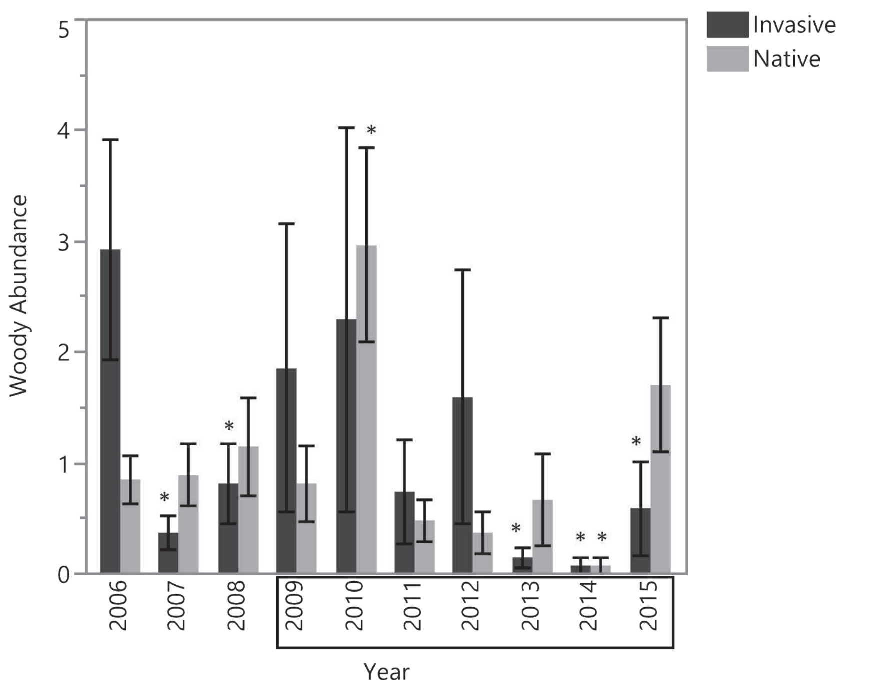


**Figure 3. Average number of native and invasive woody individuals per plot. Number of invasive individuals deceased significantly in 2007 and 2008 (before removal, $F_{1,26}$ = 7.0678, $p$ = 0.0132 and $F_{1,26}$ = 5.6610, $p$ = 0.0250) due to a drop in Ligustrum sinense seedlings. The numbers of invasives were also low in 2013, 2014, and 2015 ($F_{1,26}$ = 7.5793, $p$ = 0.0106, $F_{1,26}$ = 8.1399, $p$ = 0.0084, $F_{1,26}$ = 6.1054, $p$ = 0.0204). Native individuals increased in 2010 ($F_{1,26}$ = 6.7043, $p$ = 0.0155) and then dropped to a significant low after the drought in 2014 ($F_{1,26}$ = 10.4426, $p$ = 0.0033). Overall, numbers of woody individuals is erratic due to yearly differences in seedling abundance.**

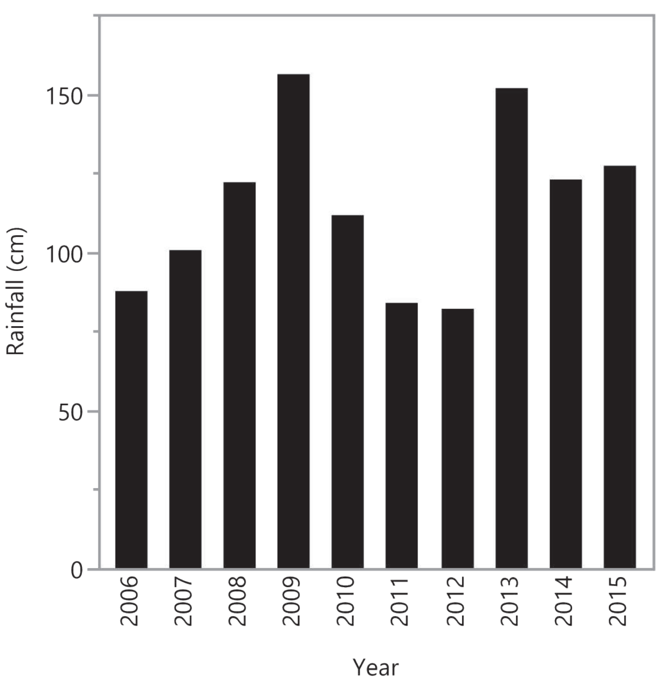


**Figure 4. Rain totals for Macon, Georgia, during the study period (NDMC 2015).**

after the initial removal of invasives were located near Walnut Creek where overbank flooding occurs frequently. This position results in disturbance to these plots from flooding while also carrying invasive propagules into the plots from nearby heavily infested forested areas. Proximity to a body of water has been shown to be a highly predictive value for likelihood of invasion (Planty-Tabacchi et al. 1996; Wang and Grant 2012). Therefore, when eradicating invasive species, position in the landscape and exposure to disturbance must be considered.

This study suggests physical removal can achieve significantly lower levels of invasive species, particularly with continued follow-up removal. Although this method is impractical to control invasive species covering large areas, this technique is valuable for parks and other natural areas that often use volunteers to help control invasive plants. Manual removal requires little training and can significantly reduce invasive species richness and abundance over time. This method is also applicable in sensitive habitats, such as wetlands or in the presence of an endangered species, where broad use of herbicides would be inappropriate (Chapman et al. 2012). Given the upward trend of native species richness following the drought, this study may support many others that showed invasive removal, particularly by manual methods, can support a recovery of native communities. The results also highlight the importance of long-term research in understanding the multiple factors, such as drought, that determine change in community composition.

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