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Impacts of removing Chinese privet from riparian forests on plant communities and tree growth five years later

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ABSTRACT

An invasive shrub, Chinese privet (*Ligustrum sinense* Lour.), was removed from heavily infested riparian forests in the Georgia Piedmont in 2005 by mulching machine or chainsaw felling. Subsequent herbicide treatment eliminated almost all privet by 2007. Recovery of plant communities, return of Chinese privet, and canopy tree growth were measured on removal plots and heavily invaded control plots in 2012 approximately five years after complete removal of privet. Plant communities were also measured on three 'desired future condition' plots which were never heavily infested with privet. These areas provided a goal condition for plant communities on removal plots. Approximately 7% of mulched plots and 3% of felling plots were re-infested by Chinese privet. In contrast, non-privet herbaceous plants covered 70% of mulched plots and 60% of felling plots compared to only 20% of untreated control plots and 70% in desired plots. Both mulched and felled plots had more plant species than the control plots, and mulched plots had more species than felled plots. Analysis of similarity (ANOSIM) and non-metric multidimensional scaling (NMS) ordination indicated that control, removal, and desired future condition plots had three distinct plant communities but the methods used to remove privet did not result in different communities. There was no difference in growth of canopy trees in removal and control plots five years after removal. Removing Chinese privet from riparian areas is beneficial to plant communities, promoting biodiversity and secondary succession while progressing toward a desired condition regardless of the method used to remove it.

1. Introduction

Chinese privet, *Ligustrum sinense* Lour. (Oleaceae), was first introduced as an ornamental plant in 1852 (Dirr, 1983). By the 1930s, it had escaped cultivation and was widely established in floodplains across the Southeastern U.S. (Small, 1933). It is now estimated to inhabit 1 million ha in the southeast (Miller et al., 2008). These estimates are misleading however, due to the underestimated area that privet inhabits in cities, towns, and along roadsides.

Privet is common in riparian areas, possibly because they are similar to its native habitat in China (Langeland and Burkes, 1998). These areas also appear to be susceptible to invasion (Stroh and Struckhoff, 2009), probably due to the same factors that contribute to the overall increased biodiversity that occurs in them (Planty-Tabacchi et al., 1996; Hood and Naiman, 2000). Favorable habitat combined with less herbivory on privet and greater fruit production, when compared to native shrubs of the same family (Morris et al., 2002), result in the formation of dense, single species shrub layers that reduce native herbaceous vegetation (Merriam and Feil, 2002; Hanula et al., 2009; Greene and Blossey, 2012).

Chinese privet is the primary cause of the decline in the abundance and diversity of native herbaceous plants and native tree seedlings in infested riparian areas (Merriam and Feil, 2002; Hanula et al., 2009; Greene and Blossey, 2012) and increasing levels of infestation result in declining abundance of canopy trees (Hanula et al., 2009). Few studies, however, have sought to determine how native flora might respond over time (>2 years) to complete removal of privet. Merriam and Feil (2002) measured plant communities with or without privet in adjacent areas where they found that areas with privet had 41% less herbaceous plants and 42% less herbaceous species. They also found 75% fewer woody stems in privet areas than in privet free areas. After removing privet, they saw a substantial increase in herbaceous plants one year later. Similarly, Hanula et al. (2009) found greater than 60% increase in overall plant cover two years after removing privet

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which was similar to plots that had historically no privet infestation.

An important aspect of privet infestation that has yet to be studied is its impact on tree growth. Recent work with hardwood tree species has focused on sapling growth and survival. Galbraith-Kent and Handel (2008) found that native sapling growth and survival was higher under a native canopy while Hartman and McCarthy (2004) found that when the invasive shrub amur honeysuckle (*Lonicera maacki*) was removed the growth and survival of saplings of six native hardwood species could be increased. Conversely, 63 weeks after transplanting native hardwood saplings under a privet canopy, Greene and Blossey (2012) saw minimal survival of the natives.

While this work is important for forest regeneration after invasive removal, the impact of removing invasive plants on the growth of mature, canopy trees is unknown. Hartman and McCarthy (2007) compared mature hardwood tree growth on sites that were invaded by *L. maacki* or not and found tree growth was negatively impacted by infestation. If removing Chinese privet from heavily infested areas produced similar results it could provide economic incentive for doing so.

Previous study into the mechanism of privet infestation of un-colonized areas has primarily focused on landscape factors contributing to susceptibility of invasion (Merriam, 2003; Stroh and Struckhoff, 2009; Wang and Grant, 2012). Their findings are important for preventing the spread of Chinese privet into un-colonized areas, but provide no evidence on the rate at which privet might re-invade an area after complete removal (Gabler and Siemann, 2012). Panetta (2000) reported that seeds of Chinese privet are relatively short lived in the seed bank (1 year) and therefore privet must rely on dispersal from local infestations to facilitate reinvasion (Panetta and Sparkes, 2001). Birds and small mammals are probably the primary vehicle for the spread of Chinese privet (Gosper et al., 2005) yet the rate and pattern of dispersal has received little attention probably due to lack of areas to evaluate reinvasion.

Here we examine the status of the herbaceous plant community five years after removal of Chinese privet and how two methods of removal affected plant community response. Plant communities were compared among removal plots, untreated control plots, and plots with historically little or no privet. We also report on the growth of canopy trees five years after removal of Chinese privet to those in untreated control plots. In addition, we measured Chinese privet reinvasion of cleared areas and if this invasion was associated with proximity to other heavily infested areas.

2. Materials and methods

2.1. Study areas

This study was part of a long-term project investigating the effects of privet removal on plant and animal communities, so the study design and locations are described in detail by Hanula et al. (2009). Briefly, four study areas were chosen along the Oconee River in northeast Georgia (Fig. S1). Two of these areas, the Botanical Gardens of Georgia (N33° 54.046′, W083° 23.435′) and Sandy Creek Nature Center (N33° 59.167′, W083° 22.865′), are located near Athens, Georgia in Clarke County. The other two areas, Watson Springs Forest (N33° 41.908′, W083° 17.695′) and Scull Shoals Experimental Area (N33° 46.132′, W083° 16.897′), are located in Greene County and in more continuously forested areas. The canopy of these study areas are dominated by green ash (*Fraxinus pennsylvanica*), sweet-gum (*Liquidambar styraciflua*), water oak (*Quercus nigra*), willow oak (*Quercus phellos*), box elder (*Acer negundo*), and loblolly pine (*Pinus taeda*). Stand conditions and overstory tree composition were measured in 2007 on five 0.04 ha subplots per treatment plot (Hanula et al., 2009) and are provided as supplementary data (Tables S1–S3). In addition, three areas of the Oconee National Forest with historically little or no privet invasion were chosen as "desired future condition" plots and were included as a reference to what treatment areas might look like long-term without privet (see Hudson, 2013 for complete list of plant species). All three sites are located at least 10 m from a river or stream.

2.2. Privet removal

The treatments consisted of heavily infested untreated controls (approximately 34% herbaceous privet cover and 62% privet shrub cover) and two methods of Chinese privet removal applied October 2005 on 2 ha plots. Privet removal was done by either mechanical mulching or hand-felling. Specifics of removal can be found in Hanula et al. (2009). Briefly, a mechanical Gyrotrac® mulching machine was used to grind up privet to ground level and created the treatment plots hereafter referred to as "mulched". Mulched residue was left in the plots. At the same time in nearby similar sized plots, crews with chainsaws and machetes hand-felled privet and left the debris in the plots (these are referred to as "hand-felling" plots). Stumps in both treatment plots were sprayed with either 30% triclopyr (Garlon® 4) or 30% glyphosate (Foresters®) herbicide to prevent re-sprouting. The herbicide was selected by the location's manager.

One year later, in December 2006, privet sprouts and seedlings were treated with a foliar application of 2% glyphosate using back pack sprayers or mist blowers. By the next summer (2007), less than 1% the plots were covered by privet in the shrub or herbaceous layer (Hanula et al., 2009).

2.3. Measuring plant communities

The plant communities in the herbaceous and shrub layer were measured using the line-point intercept method (Godínez-Alvarez et al., 2009; Outcalt and Brockway, 2010) in July 2012. Presence or absence of plants and shrubs and the species present were recorded at points every 1.5 m along three transects that spanned the length of each plot. Transects were located equidistant from each other and the plot boundary and were the same as those used previously (Hanula et al., 2009). Percent plant cover was determined by dividing the number of points with a plant by the total points sampled per plot.

2.4. Measuring tree growth

Tree growth from 2006 to 2011 was measured in treatment and control plots to determine if trees grew faster where privet was removed. Trees >4 cm DBH that were located in five 0.4 ha subplots, designated at the beginning of the study (Hanula et al., 2009), were cored with a Mattson® increment corer with a 5.15 mm core diameter to a depth sufficient to include at least 10 years of growth. Not all tree species occurred in all of the plots so we examined cores of red oaks (primarily water and willow oaks), pines, and green ash which were common to all plots. Other common species such as sweet-gum, box elder, sycamore (*Plantanus occidentalis*), river birch (*Betula nigra*), and red maple (*Acer rubrum*) were present on all plots but were not measured because growth rings were not detectable. In total, 142 cores of oaks, pines, and green ash were X-rayed so growth rings could be observed more easily.

2.5. Chinese privet reinvasion

Both removal methods created distinct edges between heavily privet infested and privet free areas. These characteristics provided a unique opportunity to determine if the spread or "reinvasion" after five years were related to proximity to the edge or if dispersal over a larger distance occurred. Only the removal plots at the Botanical Gardens, Watson Springs, and Scull Shoals sites were used because of their uniform size, shape, and relationship to the river. To assess reinvasion, we selected the edges along the river and along the opposite side of the plots from the river because these edges were consistently bordered by large privet shrubs. If privet invasion occurs from the edges, then plots nearer these edges should be more heavily reinvested than those near the middle. Chinese privet stems and stem height were measured in 28.3 m^2 circular subplots. Thirteen subplots per transect were used which were located at 10 m intervals along three transects each beginning 10 m from the river and ending 10 m from the opposite boundary farthest away from the river (Fig. 1). Transects ran perpendicular from the edge of the removal plots nearest the river and were located equidistant from one another and 70 m from edge boundaries that ran perpendicular to the river.

2.6. Experimental design and statistical analysis

The study was designed as a complete block experiment with locations as blocks. Treatments were not randomly assigned due to limited access of the mulching machine, but plots within locations had similar levels of invasion of Chinese privet so plot homogeneity was achieved within each location and, therefore, random allocation of treatments was not deemed to be essential.

Percent herbaceous plant cover and percent shrub cover were calculated by dividing the total number of points at which a non-privet plant or shrub was present by the total number of points per plot (the sum of the points in three transects per treatment at each location). Percent privet seedlings in the herbaceous layer (<2 m tall) and percent privet shrub cover (>2 m) were determined in the same manner. The effects of the three treatments on percent non-privet herbaceous plant and shrub cover, non-privet plant and shrub species richness, percent privet in the herbaceous and shrub layer, and tree growth were subjected to analysis of variance (ANOVA) using the general linear model (GLM) of SAS (SAS, 2000). Shannon diversity and evenness of herbaceous and shrub layers were also calculated and compared between treatments. Desired future condition plots were not included in any ANOVA due to their lack of association with a block. The Shapiro-Wilks test was used to determine if data were normally distributed. Percent privet in the herbaceous and shrub layer was not normally distributed. Herbaceous privet cover data were normally distributed following $\log_{10}(x+1)$ transformation, but transformation did not result in normal distribution of privet shrub cover. This dataset was non-parametrically ranked for analysis using the rank procedure of SAS. Mean separation was achieved with the Ryan–Einot–Gabriel–Welch Quotient (REGWQ) multiple comparison test.

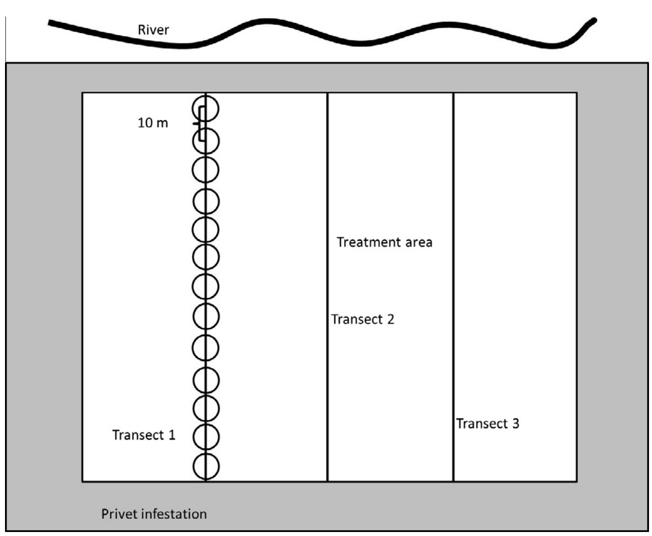

Fig. 1. Diagram showing distribution of sample plots for measuring reinvasion of Chinese privet in relation to the edge of plots and the river at the Oconee National Forest, Botanical Gardens, and Watson Springs Experimental Forest sites. The white area denotes the combined mulched and hand-felling plots and the grey denotes surrounding privet infestation. Circular subplots are 9 m in diameter (28.3 m^2) and were located along three transects that spanned the length of the combined treatment area. Centers of circular subplots were 10 m apart. Centers of first and last subplots were also 10 m from plot boundary.

Analysis of similarity (ANOSIM) was used to determine if herbaceous plant communities found at each location, treatment, and desired future condition plot were significantly dissimilar. The PAST program (Hammer et al., 2001) was used to perform ANOSIM using the Bray–Curtis distance measure. PC-ORD (McCune and Mefford, 1999) was used to perform non-metric multidimensional scaling (NMS) ordinations of herbaceous plant communities on treatment and desired future condition plots using the "slow and steady" autopilot feature. An additional ANOSIM and ordination were conducted to compare herbaceous plant communities measured in this study to those measured in 2007, two years after removal of privet (Hanula et al., 2009). PC-ORD was also used to perform an indicator species analysis to detect if certain plant species were indicative of desired, removal, or control plots. Due to the similarities between mulched and hand-felling plots, plant data from these two treatments were combined into one group termed "removal" for indicator species analysis. Indicator species from 2012 were compared to those of 2007 to determine if the indicator value of species changed over this period.

A two-way ANOVA was used to analyze privet seedling and sapling density with location and distance from the river edge as independent variables. The data were normally distributed after log transformation. Means were separated using the REGWQ multiple comparison test.

3. Results

3.1. Plant communities

Privet seedlings in the herbaceous layer ($F_{2,6} = 78.34$, $P < 0.0001$) and privet shrub cover ($F_{2,6} = 16.14$, $P = 0.0039$) differed among treatments. Approximately 7% of the mulched plots were covered by privet seedlings, which was significantly higher than the 3% in the hand-felling plots (Fig. 2). Privet seedling cover in both removal plots was also significantly lower than the 34% in control plots. The amount of privet seedlings in felling plots was very similar to the desired future condition plots (2%). Privet shrub cover in the removal plots was similar regardless of removal method and both were still very low compared to the control plots (Fig. 2).

Non-privet herbaceous plant cover was 3–4 times higher on removal plots than on the control plots ($F_{2,6} = 28.48$, $P = 0.0009$). The two methods of removal resulted in similar levels of plant cover. Plant cover on removal plots was also very similar to that of desired future condition plots (Fig. 2). Non-privet shrub cover did not differ among treatments ($F_{2,6} = 4.76$, $P = 0.0578$).

Herbaceous plant species richness also differed among treatments ($F_{2,6} = 18.04$, $P = 0.0029$). Mulched plots had the highest number of species with 32 species/plot which was significantly higher than felling (26 species/plot) and control plots (16 species/plot) but comparable to the desired plots (30 species/plot). Hand-felling plots had more species than control plots (Fig. 3). Common native species present in the herbaceous layer were box

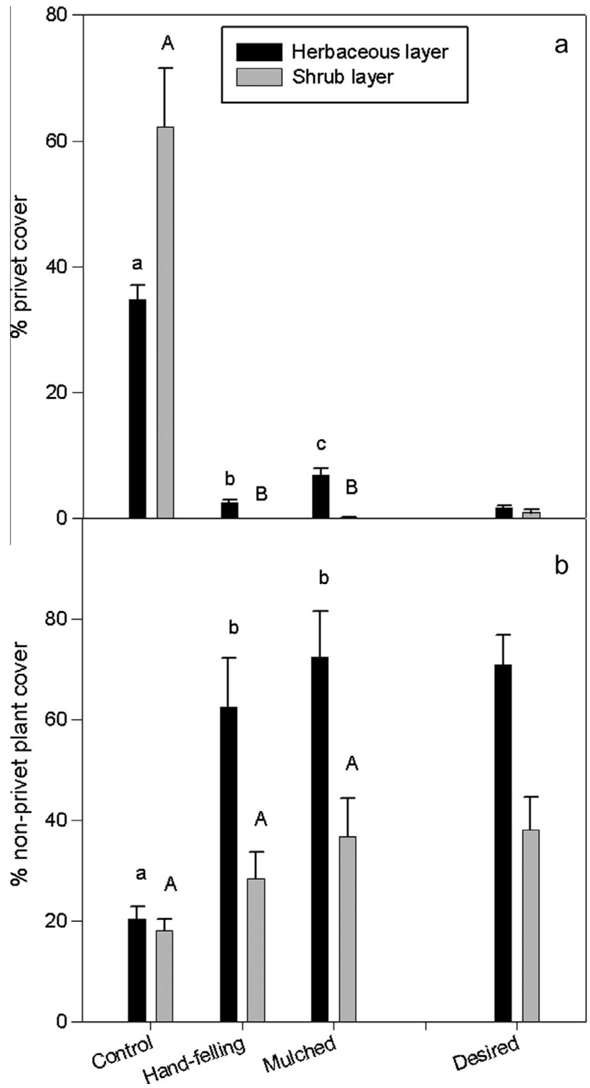

Fig. 2. Percent privet (a) and non-privet (b) plant cover in the shrub and herbaceous layers in control, hand-felling, mulched, and desired future condition plots in 2012. Bars with the same letters are not significantly different ($\alpha = 0.05$, REGWQ). Results from desired plots are included for comparison but the data were not included in analyses.

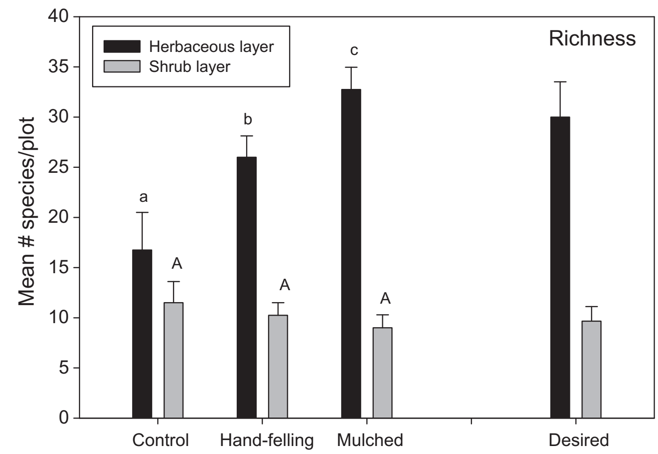

Fig. 3. Species richness of the herbaceous and shrub layers in control, hand-felling, mulched, and desired future condition plots in 2012. Bars with the same letters are not significantly different ($\alpha = 0.05$, REGWQ). Results from desired plots are included for comparison but the data were not included in analyses.

elder saplings, sedges (*Carex* sp.), muscadine (*Vitis rotundifolia*), nettles (*Urtica* sp.), pokeweed (*Phytolacca americana*), and Virginia creeper (*Parthenocissus quinquefolia*) (See Hudson, 2013 for complete list). In contrast to herbaceous plants, the number of non-privet shrub species was similar across all plots ($F_{2,6} = 0.56$, $P = 0.5979$). Common native species present in the shrub layer were box elder, eastern hop hornbeam (*Ostrya virginiana*), and American hornbeam (*Carpinus caroliniana*).

Herbaceous plant diversity ($F_{2,6} = 21.30$, $P = 0.0019$) and evenness ($F_{2,6} = 18.60$, $P = 0.0027$) also differed among treatments. Removal plots had higher diversity of herbaceous plants and greater evenness than control plots but the two methods of removal resulted in similar plant diversity and evenness (Fig. 4). Diversity and evenness of shrubs were similar among treatments ($F_{2,6} = 1.75$, $P = 0.2513$ for diversity, $F_{2,6} = 2.83$, $P = 0.1365$ for evenness).

ANOSIM showed that only plant communities on mulched and hand-felling plots were similar (Table 1). Communities on control plots and desired plots were dissimilar from each other and both removal treatments. A total of 101 species were used for NMS

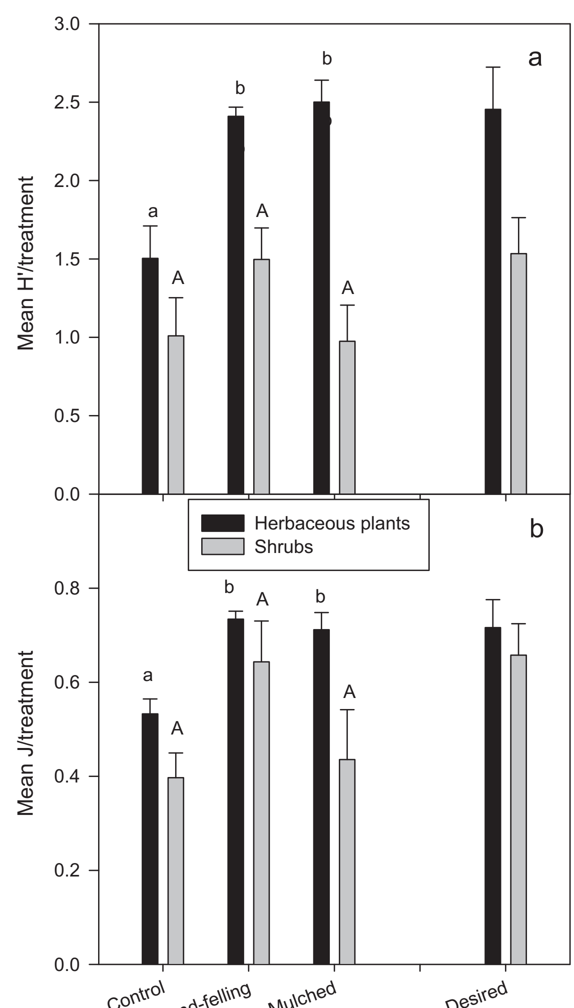

Fig. 4. Shannon diversity (A) and evenness (B) of the herbaceous and shrub layers in control, hand-felling, mulched, and desired future condition plots in 2012. Bars with the same letters are not significantly different ($\alpha = 0.05$, REGWQ). Results from desired plots are included for comparison but the data were not included in analyses.

Table 1
Analysis of similarity (ANOSIM) of 2012 herbaceous plant communities in control, hand-felling, mulched, and desired future condition plots using the Bray–Curtis distance measure.

	ANOSIM *P*-values		
	Hand-felling	Control	Mulched
Desired	0.0299*	0.0234*	0.0287*
Hand-felling		0.0299*	0.8842
Mulched		0.0286*	

* $P < 0.05$.

ordinations of the herbaceous plant communities in 2012 which resulted in a two dimensional solution with a final stress of 9.90 (Fig. 5). Plant communities of control plots were grouped closely. Hand-felling plots and mulched plots were grouped together and formed a separate group from desired and control plots.

ANOSIM of herbaceous plant communities from 2007 and 2012 showed that hand-felling and mulched plots in 2007 were similar to hand-felling plots in 2012 (Table 2). Desired plots from both years were similar to each other as were control plots. Mulched plots in 2012 were similar to desired plots in 2007 and hand-felling plots in 2012 but dissimilar from all other treatments. NMS ordination resulted in a two dimensional solution with a final stress of 13.98 using 113 species (Fig. 6). Desired plots from 2007 and 2012 were grouped together by location. Plant communities on removal plots in 2007 and 2012 were grouped together although two plots from 2012 were near two of the desired future condition plots.

There were nine species with significant value as indicators of the desired plots in 2007 (Table 3). Two, crossvine (*Bignonia capreolata*) and Carolina silverbell (*Halesia carolina*), occurred exclusively in the desired plots in 2007 and 94% of panic grasses (*Dicanthelium* sp.) occurred in them. Fireweed (*Erichtites hieracifolia*) and pokeweed were the best indicators of removal plots in 2007 while three other species also had significant indicator value. Chinese privet was the only indicator species for control plots.

In 2012, desired plots had 10 species with significant indicator values. Of those, crossvine (92%) still had a high indicator value but it was no longer exclusively found in desired plots. Wild rye grass (*Elymus virginicus*) had the highest indicator value (95%) for desired plots in 2012. Other indicator species of desired plots in 2012 were river oats (*Chasmanthium latifolium*), panic grasses, and American hornbeam. Four species were indicators of removal plots in 2012. Pokeweed was found exclusively in removal plots while nettle and *Carex* sp. also had high indicator value for removal plots. Like 2007, Chinese privet was the only indicator species for controls.

3.2. Tree growth

Removal of Chinese privet did not result in detectable changes in growth of trees ($F_{2,6} = 0.10$, $P = 0.9064$).

3.3. Privet reinvasion

Privet seedling and sapling density was highest 10 m from the river edge but not significantly higher than densities at 20, 30, 40, 70, 110, and 130 ($F_{2,12} = 4.01$, $P < 0.0001$). Privet densities 20 m from the river were higher than those at 50, 60, 80, 90, 100, and 120 (Fig. 7).

4. Discussion

Long-term monitoring of native plant recovery and potential reinvasion after invasive plant removal is crucial in determining removal efficacy and justification of future control efforts (Kettenring and Adams, 2011). In 2007, Hanula et al. (2009) found an average of 1% cover of privet seedlings in the herbaceous layer of these removal plots. Five years later in 2012, we found that privet cover had increased to 3% in the hand-felling plots and 7% in mulched plots (Fig. 2). The detrimental effects of Chinese privet infestation, however, are mostly attributed to shade produced from shrub cover, resulting in decreased biodiversity and limiting recreation (Sparks et al., 1996; McKinney and Goodell, 2010). Five years after the shrubs were removed no privet had grown sufficiently to be classified as a shrub and the few privet shrubs remaining on the plots were residuals missed during removal. Overall, these results are encouraging since we expected to have to re-treat privet frequently to preserve the integrity of removal plots. These results indicate control following one removal event last at least five years. Continued monitoring is planned to document reinvasion and to decide when follow-up treatment is needed.

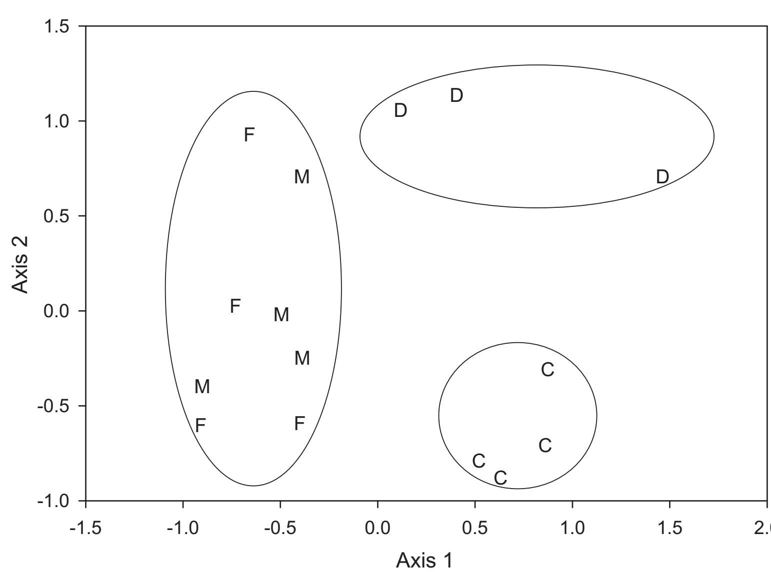

Fig. 5. NMS ordination (Final stress = 9.90) of the herbaceous plant communities found in control, mulched, hand-felling, and desired future condition plots in 2012. Plot symbols refer to treatment: D = Desired future condition, M = Mulched, F = Hand-felling, C = Control. Circles indicate ANOSIM groupings (Table 1).

Table 2
Analysis of similarity (ANOSIM) of herbaceous plant communities in control, hand-felling, mulched, and desired future condition plots in 2007 (07) and 2012 (12) using the Bray–Curtis distance measure. Des = Desired, Fell = Hand-felling, Con = Control, Mulch = Mulched.

	ANOSIM *P*-values						
	Des12	Fell07	Fell12	Con07	Con12	Mulch07	Mulch12
Des07	0.505	0.0281*	0.0285*	0.0324*	0.0287*	0.0278*	0.0569
Des12		0.0283*	0.026*	0.0276*	0.0305*	0.0271*	0.029*
Fell07			0.1669	0.0289*	0.0293*	0.5122	0.0325*
Fell12				0.0289*	0.0288*	0.0579	0.9714
Con07					0.6822	0.0276*	0.0282*
Con12						0.0291*	0.0277*
Mulch07							0.0279*

* $P < 0.05$.

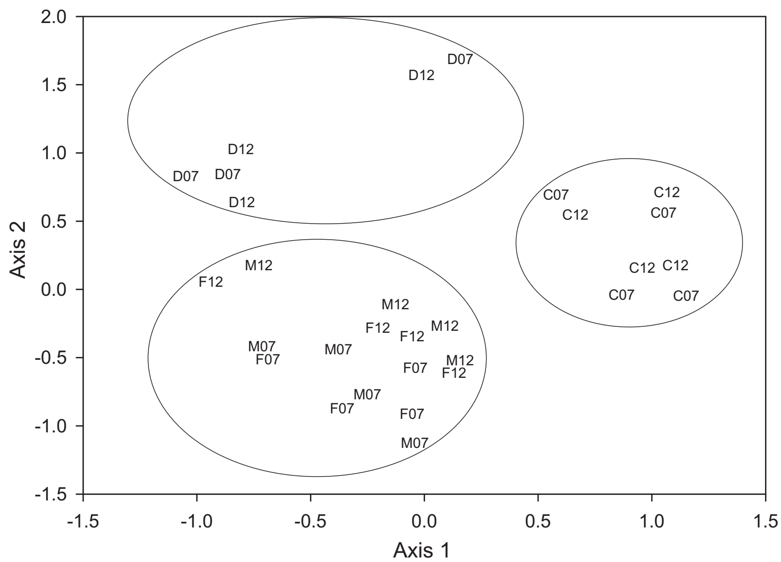

Fig. 6. NMS ordination (Final stress = 13.98) of the herbaceous plant communities found in control, mulched, hand-felling, and desired future condition plots in 2007 (07) and 2012 (12). Plot symbols refer to year and treatment: D = Desired future condition, F = Hand-felling, M = Mulched, C = Control. Data for 2007 are from Hanula et al. (2009). Circles indicate ANOSIM groupings of similar communities for control and desired future condition but not among mulched and felled plots. See Table 2 for dissimilarities within that grouping.

Richardson et al. (2007) stressed that restoration of plant communities to pre-invasion conditions is probably not feasible. These authors recommend pragmatic and innovative approaches to restoration of riparian system structure and function that take into consideration human needs. The goal condition of the plant communities for our removal plots was to be similar to that of desired future condition plots. Although herbaceous plant cover, species richness, and diversity were similar (Figs. 2–4), removal plots were not similar to desired plots (Table 1 and Fig. 5). Desired plots contained more grass species, mainly river oats, panic grasses, and wild rye grass (Table 3), but they also contained Japanese stilt grass (*Microstegium vinimeum*) and Japanese honeysuckle (*Lonicera japonica*). The latter two are undesirable species but beneath an undisturbed canopy, they appeared to have less of an impact on plant communities than Chinese privet. However, Japanese stilt grass does reduce tree seedling recruitment and herbaceous plant development following a disturbance in stands where it is well established (Oswalt et al., 2007; Judge et al., 2008). Plant communities in two removal plots were trending toward desired plots (Figs. 5 and 6). Both included more Japanese stilt grass, native grass species listed above, and fewer woody saplings than the other removal plots, which may have contributed to their proximity to desired plots in ordinations.

After five years of native plant recovery and secondary succession on our plots, total plant cover increased by approximately 10% in mulched plots and 20% in hand-felling plots from what was measured in 2007 (Hanula et al., 2009). Plant communities in removal plots were grouped together in ordinations (Fig. 6) and a number of common, early colonizing species like pokeweed and nettle were still present five years after initial disturbance (Table 2).

Woody saplings covered more of the removal plots in 2012 than 2007, and species like box elder, sweet-gum, and green ash made up the majority of saplings in 2012. Oosting (1942) was the first to measure sapling regeneration of the Piedmont and reported an early dominance of sweet-gum which was what we expected to happen as well. However, in our study box elder was clearly the dominant species forming dense stands in multiple removal plots. Indeed, sweet-gum was an indicator species of removal plots in 2007 but not 2012 while box elder exhibited the opposite trend (Table 3). Other studies have reported dense stands of box elder and primarily attribute them to disturbed, regenerating sites like ours (Cowell, 1998; Zipperer, 2002; DeWine and Cooper, 2007). Box elder is also invasive in Europe, probably due to factors such as its prolific seed production and ability to invade disturbed areas with bare ground and open canopies (Bottollier-Curtet et al., 2012). These same attributes were likely important in its success on our plots where box elder was more common along the edge of the plots bordering the river where basal area was lowest. Zipperer (2002) showed that forests with high densities of box elder were less invaded by non-native plants so, although privet and box elder are both shade tolerant, box elder is fast growing and may be able

Table 3
Significant indicator species (Monte Carlo test, $P < 0.05$) for 2007 and 2012 plant communities on desired future condition, untreated control, and Chinese privet removal plots. Mulched and hand-felling plots were combined into the removal category for analysis since they had similar plant communities.

Plant species	Indicator Value			
	Desired	Removal	Control	P
2007				
Bignonia capreolata	100*	0	0	0.0022
Carex sp.	67*	0	0	0.0284
Chasmanthium latifolium	67*	0	0	0.0284
Desmodium sp.	67*	0	0	0.0284
Dichanthelium sp.	94*	0	2	0.0062
Erechtites hieracifolia	0	96*	1	0.0004
Halesia carolina	100*	0	0	0.0022
Ligustrum sinense	0	4	93*	0.0006
Lindera sp.	67*	0	0	0.0284
Liquidambar styraciflua	0	67*	12	0.0492
Lonicera japonica	72*	0	14	0.0296
Phytolacca americana	0	98*	1	0.0004
Rubus sp.	71*	0	7	0.0184
Urtica sp.	4	87*	4	0.0012
Viola sp.	0	76*	12	0.0068
2012				
Acer negundo	1	74*	24	0.0054
Bignonia capreolata	92*	1	1	0.0032
Carex sp.	0	88*	0	0.0068
Carpinus caroliniana	90*	3	1	0.0102
Chasmanthium latifolium	89*	3	0	0.0058
Dichanthelium sp.	89*	4	2	0.0036
Elymus virginicus	95*	1	0	0.002
Ligustrum sinense	2	8	89*	0.0002
Lindera sp.	67*	0	0	0.0274
Mitchella repens	63*	1	0	0.0284
Phytolacca americana	0	100*	0	0.0004
Smilax sp.	77*	11	6	0.0124
Toxicodendron radicans	58*	22	13	0.0234
Unknown legume	67*	0	0	0.0274
Urtica sp.	2	93*	2	0.0004

* Indicator for treatment, out of 100 possible.

to outcompete privet allowing a more natural succession to occur (DeWine and Cooper 2008).

Chinese privet removal had no effect on the growth of mature trees after five years. Hartman and McCarthy (2007) found growth of trees before invasion by *Lonicera maccki* to be higher than after invasion so it interfered with tree growth. One key difference in the studies, other than shrub species and location, may be that Hartman and McCarthy (2007) were able to document when invasion occurred and examine growth before and after that time, i.e. they measured the declining growth as invasion occurred while we measured growth five years after removal. Mature canopy trees may have little response to changes in the understory (Kelty et al., 1987). Juodvalkis et al. (2005) surmise that younger trees may take advantage of reduced competition to increase their crown while older trees do not. Even though we measured growth of all trees >4 cm DBH, size may not correspond to age, and small trees on our plots may have been suppressed, older trees and unable to respond to changes in the understory.

Chinese privet densities in removal plots were greatest within 10–20 m of the river edge (Fig. 7) although not significantly higher than at 30, 40, 70, 110 and 130 m. Merriam (2003) found a strong correlation of river and stream banks to Chinese privet abundance, and Wang and Grant (2012) found that of landscape features, proximity to a body of water had the highest predictive value for likelihood of invasion. Since our plots had two edges bordered by large privet shrubs we expected both edges to have an increased number of privet stems when compared to the middle of removal plots, which was 70 m from either edge, but that was not clearly the case.

Hughes et al. (2012) concluded that the key to recovery and early resistance to reinvasion lies in the abundance of species with pioneer traits like the ability to colonize early in succession. As described above, regenerating trees like box elder may be the key to resistance on our plots. Our observations suggest that the distribution of box elder, also in the more open areas near the river, coincided with areas of greater privet reinvasion. Box elder, a deciduous species, may capture more light during the growing season and be able to compete for dominance with privet during that time of the year. But evergreen plants like privet have an advantage over regenerating deciduous trees and shrubs especially in mild climates like the southeast by having a longer growing season and being able to grow during warm spells throughout the winter (Givnish 2002, and references therein).

Plant removal is not feasible in all situations. Ecosystem level intervention, such as biological control (Blossey, 1999; Gabler and Siemann, 2012), may provide an alternative approach to

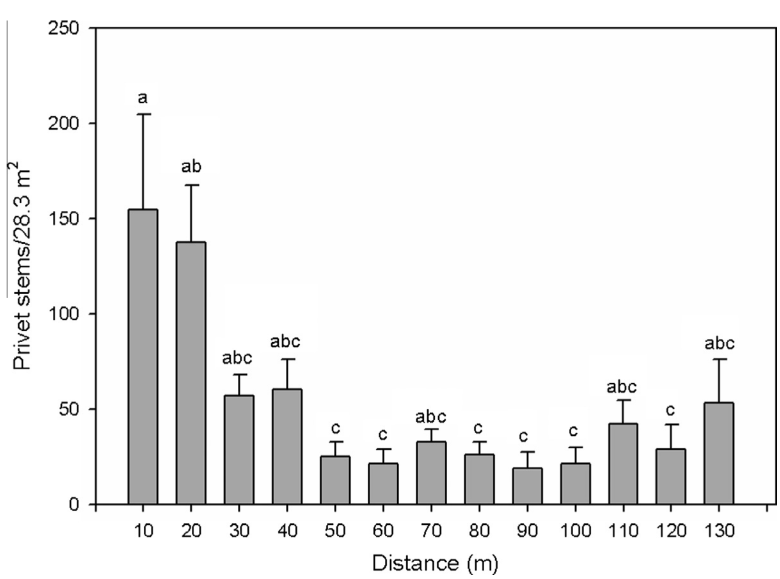

Fig. 7. Privet density (stems/28.3 m^2 ± SE) in subplots located at 10 m intervals from the plot boundary nearest the river edge in plots were Chinese privet had been removed 5 years earlier. Bars with the same letter above them are not significantly different (REGWQ multiple comparison test, $\alpha = 0.05$).

controlling privet. Unfortunately, such a control has not been identified for privet. Blossey (1999) stressed that before releasing biological agents for the control of an invasive plant, it should be demonstrated that the exotic has negative impacts on the native community and that beneficial effects of removal can be sustained long term, both of which are documented by our study.

5. Conclusion

Removing Chinese privet from riparian areas is beneficial to plant communities, promoting biodiversity and secondary succession. Five years after privet removal these areas are still relatively privet free and are still usable for human enjoyment and recreation, but the plant communities on them are early successional and not similar to the desired future condition we chose. While this study did not demonstrate increased tree growth, which would be a further incentive for removing privet, it may be that more time is needed for trees to recover from years of competition with this shrub. Additionally, a longer timeframe is needed to quantify re-infestation of Chinese privet in these areas. Determining when privet reinvasion has advanced to the point further control will be needed and a threshold of infestation that is detrimental was an important consideration for this study. Five years after removal, privet levels were relatively low and further control is currently unnecessary.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2014.04.013.

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