Evaluating Remote Sensing and Prescribed Fire Methods to Aid in the Restoration of Bottomland Hardwood Forests Invaded by Chinese Privet (*Ligustrum sinense*)

by

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Abstract

Chinese privet (*Ligustrum sinense* Lour.) is an invasive shrub with a wide distribution outside its native Southeast Asia range. It is particularly common and problematic in the southeastern U.S., where researchers have documented negative impacts to woody and herbaceous plant communities, which in turn likely affects wildlife habitat quality. This research project evaluated two management tools that could assist land managers in efforts to restore bottomland hardwood forests that have been invaded by *L. sinense*. Our first objective was to evaluate whether free multispectral satellite imagery and free simple-to-use software could be used to map *L. sinense* invasions and aid in planning and budgeting restoration projects. We found that the Semi-Automatic Classification Plugin within QGIS was effective at detecting *L. sinense*, particularly when using late dormant season Sentinel 2 imagery. Our second objective was to evaluate whether prescribed fire could reliably move through bottomland hardwood forests and reduce *L. sinense* slash following cutting operations to improve the efficiency of follow-up treatments on re-sprouts. We found that stand composition had the most important effect on fire spread (plots with greater proportions of tree species with flammable leaf traits tended to burn best), but that the fires were only successful in a limited number of cases. Only small diameter *L. sinense* slash was significantly reduced, and more research is needed to determine whether prescribed fire is beneficial in this context.

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DBH	Diameter at Breast Height
DFC	Desired Future Condition
ESA	European Space Agency
GIS	Geographic Information System
GUI	Graphical User Interface
IPT	Individual Plant Treatment
L8	Landsat 8 Satellite
MD	Minimum Distance
ML	Maximum Likelihood
RBA	Relative Basal Area
RF	Random Forest
RH	Relative Humidity
S2	Sentinel 2 Satellite
SA	Spectral Angle Mapping
SCP	Semi-Automatic Classification Plugin
USGS	United States Geological Survey

Chapter 1

The Ecological Effects of *Ligustrum sinense* Invasion: A Synthesis

Abstract

Chinese privet (*Ligustrum sinense* Lour.) is an evergreen shrub with an expansive non-native global range. Control costs are often high, so land managers must carefully consider whether the plant's negative impacts warrant active management. To help facilitate this decision making process, we have reviewed and synthesized the literature on the ecological effects of *L. sinense* invasion. We also identified research gaps in need of further study. We found ample evidence of negative correlations between *L. sinense* invasion and various measures of herbaceous and woody plant communities. Of particular concern is the possibility that *L. sinense* could suppress forest regeneration and cause these areas to transition from forest to *L. sinense*-dominated thickets. Evidence of negative effects on wildlife are limited and some species may actually benefit from the additional cover and foraging opportunities that *L. sinense* can provide. However, we predict that at high *L. sinense* densities, the effects on plant communities will negatively impact many wildlife species. Further research on the effects of *L. sinense* invasion on large scale forest structure and wildlife populations are needed. Early detection and management is strongly encouraged before the effects of *L. sinense* intensify and control costs become excessive.

Introduction

Chinese privet (*Ligustrum sinense* Lour.) has become a problematic invasive species in many parts of the world. A native of Asia, it has been introduced to every continent except Antarctica (Figure 1.1), and is considered invasive in at least 20 U.S states, 6 Pacific islands, Australia, Italy, Argentina, and Puerto Rico (CABI 2018). *Ligustrum sinense* is particularly problematic in the southeastern U.S, where it is consistently listed as one of the most damaging invasive species in the region (Miller et al. 2004). Introduced for landscaping in 1852, *L. sinense* (along with the very similar European privet [*Ligustrum vulgare* L.]) was estimated in 2008 to cover over 1-million ha in the southeastern U.S. alone (Miller and Chambliss 2008; Maddox et al. 2010). In Mississippi and Alabama, *L. sinense* and *L. vulgare* were estimated to have occupied 0.12 million ha (<2%) of forests in 2003, but are projected to reach 2.83 million ha (31%) by 2023 (Wang et al. 2016). *Ligustrum sinense* control is expensive ($216 - $1,820 per ha, Benez-Secancho et al. 2018; Klepac et al. 2007), due to its propensity to form dense stands and re-sprout following cutting. Understanding the ecological cost of *L. sinense* invasions can help land managers determine if it is worth the monetary cost to implement control measures. Accordingly, we review the known and potential impacts that *L. sinense* has on native vegetation and wildlife communities in this article. We also provide recommendations for future research priorities, based on important research gaps identified in our review.

Ligustrum sinense is an evergreen to deciduous (depending on latitude) single or multi-stemmed shrub/small tree that reaches a maximum height of 10 m (Maddox et al. 2010; Miller and Miller 2005). *Ligustrum sinense* seeds are encased in ovoid drupes and are spread via endozoochory and hydrochory (Foard 2014; Miller and Miller 2005). An Australian study found that *L. sinense* can produce 1300 fruits m^{-2} of canopy, with fruit production positively related to light availability and stem diameter (Westoby et. al. 1983). Flowering occurs during spring and early summer and fruit is available during fall and winter (Miller and Miller 2005). Local dispersal can also occur via root-sprouting (i.e., new shoots can grow from belowground roots; Miller and Miller 2005). *Ligustrum sinense* has broad environmental tolerances and can survive in areas with full sun or heavy shade (Brown and Pezeshki 2000; Dirr 1998; Grove and Clarkson 2005), although light availability may affect survival in some situations (Kuebbing et al. 2015). It is found in a variety of sites including bottomland hardwood forests, upland forests, swamps, cedar glades, and old fields (Cofer et al. 2008; Grove and Clarkson 2005; Miller and Miller 2005; Pokswinki 2009). Riparian areas and forest edges (Figure 1.2) are particularly prone to invasion (Merriam 2003). Transplant experiments and field surveys have shown that *L. sinense* can regenerate under a canopy of conspecifics, indicating that invasions can likely sustain themselves over long periods of time—although for how long is unknown (Greene and Blossey 2012; Grove and Clarkson 2005; McAlpine et al. 2018).

Methods

To evaluate the current understanding of the ecological impacts caused by *L. sinense,* we searched the Web of Science database using the search terms “Chinese privet” and “*Ligustrum sinense*” (last accessed 3/21/2019). All articles pertaining to impacts of *L. sinense* on ecosystems outside its native range were reviewed and their reference lists were searched for additional relevant articles. We relied primarily on peer-reviewed sources, but infrequently cite “grey literature.” We have focused this review on *L. sinense*, but occasionally reference research on other *Ligustrum* species and similar exotic shrubs if relevant *L. sinense* research does not exist for specific subjects. The vast majority of the studies we found were conducted in the eastern

U.S., so we only identified the location of the study in our discussion if it occurred outside this area. We generally cataloged papers as either ‘vegetation impacts’ or ‘wildlife impacts’, although there were several instances where papers covered both topics. Further, we reviewed and included papers that may have been less directly focused on vegetation or wildlife (e.g., impact of *L. sinense* on soil properties) that still contributed to our understanding of the potential impact of *L. sinense* on native plant and animal communities.

Results and Discussion

Vegetation Impacts. Non-native invasive plants can negatively impact native plants by outcompeting them for resources such as nutrients, sunlight, water, and space (Gioria and Osborne 2014). Invasive non-natives may also indirectly affect native species by altering ecological processes such as soil nutrient cycling or fire regimes (Ehrenfeld 2010). We begin this section with summaries of the documented impacts of *L. sinense* on herbaceous and woody plants. We then discuss the possible mechanisms through which *L. sinense* causes these impacts and the potential for native vegetation recovery following *L. sinense* removal. We conclude this section with a discussion of how passenger-driver-backseat driver frameworks may apply to *L. sinense* invasions.

Herbaceous Species. Perhaps the most visually striking impact of dense *L. sinense* stands is the effect they have on herbaceous ground cover (i.e., forbs and grasses). It is common to see very sparse native ground cover under a dense *L. sinense* midstory (Figure 1.3). Studies have confirmed that there can be significantly lower herbaceous species diversity and stem density in areas with high (~100% cover) monospecific cover of *L. sinense*, compared to those with little or no *L. sinense* cover (Kittel 2001; Merriam and Feil 2002). Further, studies that surveyed a range of *L. sinense* cover classes have documented negative relationships between *L. sinense* cover and herbaceous species richness, cover, and stem density (Greene and Blossey 2012; Wilcox and Beck 2007). For instance, Greene and Blossey (2012) found that as *L. sinense* cover increased from 0% to about 60%, mean herbaceous cover decreased from about 58% to 25%, mean species richness decreased from about 10 to 5 species m^{-2}, and mean stem density decreased from about 225 to 100 stems m^{-2}. Maximum herbaceous height also decreased, but it was not statistically significant.

Ligustrum sinense has been implicated in the decline of several species of conservation concern, including the green pitcherplant (*Sarracenia oreophila* [Kearney] Wherry; Schnell et al. 2000), fringed campion (*Silene polypetala* [Walter] Fernald & B.G. Schub.; USFWS 1996), and Schweinitz's sunflower (*Helianthus schweinitzii* Torr. & A. Gray; Urbatsch 2000). However, a study on the threatened large flowered skullcap (*Scutellaria montana* Chapm.) found that low *L. sinense* cover (10-15%) actually had a slightly positive effect on stem growth compared to plots where *L. sinense* and Japanese honeysuckle (*Lonicera japonica* Thunb.) had been removed (Sikkema and Boyd 2015). The presence of *L. sinense* and *L. japonica* may have benefited *S. montana* by reducing grazing pressure and/or modifying abiotic variables such as soil moisture (Sikkema and Boyd 2015). It is not clear how these relationships may change under higher *L. sinense* cover, or whether similar beneficial relationships exist between *L. sinense* and other native species.

Woody Species. Impacts to the herbaceous understory may be the most apparent to the casual observer, but *L. sinense* impact on woody species may have far more prolonged ecological consequences. The density of understory and midstory woody plants, including shrubs, saplings, and seedlings, is often lower in invaded areas compared to uninvaded areas, and has been shown to be negatively correlated with measures *L. sinense* prevalence (Barksdale and Anderson 2015; Hanula et al. 2009; Hart and Holmes 2013; Kittel 2001; Lowenstein and Lowenstein 2005; Merriam and Feil 2002). Specifically, Wilcox and Beck (2007) found that native shrub density in plots with high *L. sinense* cover (90.6 ± 4.6%) was almost 2/3 less than in plots with low (0.4 ± 0.6%) or medium (13.6 ± 1.5%) *L. sinense* cover. Similar patterns have been observed between *L. sinense* and woody species diversity (Burton 2005; Foard et al. 2016; Hart and Holmes 2013; Hanula et al. 2009; Kittel 2001; Lowenstein and Lowenstein 2005; Merriam and Feil 2002; Wilcox and Beck 2007).

Given the evidence that *L. sinense* reduces the diversity and richness of woody seedlings and saplings, multiple authors have expressed concern that over time invaded forests will convert to *L. sinense*-dominated shrublands due to inadequate woody species regeneration (e.g., Green and Blossey 2012; Merriam and Feil 2002; Lowenstein and Lowenstein 2005; Hart and Holmes 2013). In support of this hypothesis, several researchers have detected a negative relationship between *L. sinense* cover and tree density (Barksdale and Anderson 2015; Hanula et al. 2009; Wilcox and Beck 2007). Additionally, Hagan et al. (2014) reported that tree basal area was lower in invaded plots (but see discussion on invasion frameworks below). However, Hanula et al. (2009) and Greene and Blossey (2014) found no significant relationship between tree basal area and *L. sinense* cover, and others have found no significant effect of *L. sinense* on tree diversity (Kittel 2001, Wilcox and Beck 2007). Examples of large scale forest-to-*L. sinense* shrubland conversion are lacking in the literature, perhaps because not enough time has elapsed since *L. sinense* became established in its current non-native distribution. Nonetheless, Hart and Holmes (2013) reported that *L. sinense* can occasionally occupy canopy-dominant positions, perhaps due to lack of native tree regeneration following the creation of canopy gaps from local disturbances. Further supporting the hypothesis that *L. sinense* could lead to forest canopy degradation and loss are reports that the congeneric glossy privet (*Ligustrum lucidum* W.T. Aiton) has nearly replaced forest stands in some areas of Argentina and New Zealand, (Hoyos et al. 2010; Swarbrick et al. 1999).

In addition to effects on the woody regeneration layer, *L. sinense* invasion may also impact forest stands by influencing the growth and survival of mature trees, however there has been limited research on such effects. Foard et al. (2016) observed that mature *Quercus* spp. in an invaded stand had higher rates of self-thinning and slower growth compared to an un-invaded stand (although their sample size was small). However, Brantley (2008) found no detectible influence of *L. sinense* on overstory tree growth, and Hudson et al. (2014) observed no change in tree growth 5 years after *L. sinense* removal. The mixed reports regarding the relationship between tree basal area and *L. sinense* presence further complicate our understanding of how *L. sinense* may or may not affect mature trees (Greene and Blossey 2014; Hagan et al. 2014; Hanula et al. 2009). More research is likely needed to determine the effect of *L. sinense* on overstory tree health, stand regeneration, and woody community composition.

Mechanisms of Impact. Competition for light seems the most obvious mechanism by which *L. sinense* would affect native plants (Greene and Blossey 2012). Competition for light is an important mechanism through which non-native plants influence native species (Gioria and

Osborne 2014). Lower light availability tends to promote shade tolerant species over shade intolerant species (Lin et al. 2002), thus changes in light availability has the potential to restructure plant communities. Reductions in light availability have been observed under *L. lucidium* canopies (Swarbrick et a. 1999), which suggests that *L. sinense* would have a similar effect. However, there are conflicting reports regarding how much light availability differs in *L. sinense* stands of varying densities. For example, Osland et al. (2009) documented roughly 4x greater light availability in areas where privet had been removed compared to areas where it was still present. However, Brantley (2008) and Pokswinki (2009) found no significant relationship between *L. sinense* cover and light intensity. These conflicting results could be due to differences in methodology or the overstory structure of the study sites. It is possible that in closed canopy forests a *L. sinense* midstory may not have a significant additive effect on understory light availability; however, competition for light likely does play a role when *L. sinense* is competing against native species in canopy gaps and along forest edges.

In addition to potential effects on light availability, *L. sinense* may affect native plant communities by altering soil nutrient dynamics. Nutrient cycles in invaded forests may be altered due to changes in litter decomposition rates, litter chemical composition, and leaf abscission timing (Mitchell et al. 2011), as well as the ability of *L. sinense* to intercept litter from the overstory (Faulkner et al. 1989). Litter decomposition rates have been found to increase as the proportion of *L. sinense* in the litter increases (Mitchell et al. 2011), leading to faster turnover rates. Specifically, litter turnover decreased from 7.1 to 2.6 years as the percent of *L. sinense* in the leaf litter increased from 0 to 50% (Mitchell et al. 2011). *Ligustrum sinense* litter is higher in nitrogen, which may promote rapid decomposition and increase nitrogen availability in invaded systems (Mitchell et al. 2011). The alteration of soil nutrient cycles and nutrient availability could affect competitive interactions among plant species (e.g., Aerts and Berendse 1988); although at this point data is only available on how *L. sinense* may affect nutrient dynamics, further research is needed to determine how those changes may impact native plants.

Changes in soil microbial and fungal communities have been suggested as another mechanism through which *L. sinense* influences native and non-native plant communities (Greipsson and DiTommaso 2006; Kuebbing et al. 2014; Kuebbing et al. 2015; Kuebbing et al. 2016), and these changes may be associated with and/or interact with changes in soil nutrient dynamics (Deyn et al. 2004; Ehrenfeld 2003). Research has shown that native species tend to perform better (as measured by shoot and root mass) in soils without a history of invasion by *L. sinense* or Amur honeysuckle (*Lonicera maackii* Rupr.); however, this may not translate to significant differences in native plant communities (Kuebbing et al. 2014; Kuebbing et al. 2016). On the other hand, soils with a history of *L. sinense* and *L. maackii* invasion may actually promote the occurrence of some non-native species (Kuebbing et al. 2014; Kuebbing et al. 2015). Kuebbing et al. (2015, 2016) suggested that soil microbial communities were involved in these legacy soil effects; however, they did not directly measure microbial activity. Kuebbing et al. (2014) attempted to examine whether microbial activity was affected by *L. sinense* and/or *L. maackii* invasion based on enzyme activity measurements, but their results were inconclusive. Although there were higher activity levels of a carbon degrading enzyme in *L. sinense* plots (which could impact nutrient dynamics), the effect was not statistically significant and enzyme activity in general did not vary significantly as a result of *L. sinense* presence (Kuebbing et al. 2014). Research has also shown that *L. sinense* can alter arbuscular mycorrhizal fungi communities in the soil, which could negatively impact native plant species that require specific arbuscular mycorrhizal fungi community associations (Greipsson and DiTommaso 2006). More research is needed to determine how *L. sinense* effects soil microbial and fungal communities, how these alterations may impact native plant species, and how long these altered conditions may last following *L. sinense* removal.

Allelopathy occurs when a plant secretes chemicals into the soil that negatively affect other plant species, thus giving the plant a competitive advantage (Hierro and Callaway 2003). There is evidence that suggests *L. sinense* may exhibit allelopathy, however more research is needed to confirm this. Allelopathic potential was evaluated in several studies through the use of *L. sinense* extracts created by soaking various parts of the plant in water, which were subsequently used in diluted concentrations to water the seeds of multiple plant species. Early studies found that watering with diluted *L. sinense* extracts had negative effects on the germination and growth of radish (*Raphanus sativus* L.) and tomato (*Solanum lycopersicum* L.) seedlings (Grove and Clarkson 2005, Pokwinski 2009). Barnett et al. (2016) conducted a similar experiment with species native to the southeastern U.S., including common persimmon (*Diospyros virginiana* L.), red mulberry (*Morus rubra* L.), soapberry (*Sapindus saponaria* L.), and American beautyberry (*Callicarpa americana* L.). They found negative impacts of *L. sinense* extract on *S. saponaria* and *C. americana* germination rates and *C. americana* root growth. These experiments demonstrated the potential for *L. sinense* allelopathy, but without identification of the specific chemical(s) involved and evidence that such chemical(s) have a significant effect under field conditions, it is not possible to definitively say that *L. sinense* exhibits allelopathy (Pokwinski 2009).

Another mechanism through which *L. sinense* could impact native plant communities is by altering fuel loading and types, thereby affecting fire-dependent ecosystems. Research suggests that *L. sinense* may have mixed effects on forest fuels. Specifically, it likely reduces fine fuels at ground level by outcompeting herbaceous species and reducing litter accumulation (Faulkner et al. 1989, Stocker and Hupp 2008, Mitchell et al. 2011), although there are conflicting reports regarding the effect on litter accumulation (Hagan et al. 2014). Dense midstory vegetation also tends to increase fine fuel moisture, further reducing fire frequency and intensity (Nowacki and Abrams 2008). *Ligustrum sinense* itself has low ignitability (Tiller 2015). The potential for reduced understory fuel loads and increased fuel moisture under an *L. sinense* canopy likely suppresses fires under most conditions. Fire suppression by *L. sinense* could disrupt natural successional processes and endanger fire-adapted communities such as oak-pine forests that benefit from the recruitment opportunities that fire provides (Nowacki and Abrams 2008, Stocker and Hupp 2008). On the other hand, there is some concern that as the climate becomes warmer and dryer, *L. sinense* could become more ignitable and function as a ladder fuel resulting in increased incidence of crown fires (Wang et al. 2016), although this is a controversial prediction that warrants further investigation. An increase in crown fires could accelerate the shift from a tree to shrub dominated community because *L. sinense* is able to quickly re-sprout following fire.

Finally, invasion by non-native plants can also affect native plants through the alteration of wildlife-plant interactions (Traveset and Richardson 2004). For instance, fruiting and flowering non-native plants can compete with native plants for pollinators or seed dispersers (Traveset and Richardson 2004). An example of such an interaction occurs in Europe between the non-native Himalayan balsam (*Impatiens glandulifera* Royle) and the native marsh woundwort (*Stachys palustris* L.), which experiences reduced pollinator visits and seed set when in the presence of *I. glandulifera* (Chittka and Schürkens 2001). This has not been directly studied in regards to *L. sinense*, however it is a possibility worthy of future study.

Potential for Restoration. It is important to determine whether removing *L. sinense* stands leads to the recovery of plant communities that resemble areas with no history of *L. sinense* invasion. Studies looking at restoration of forests invaded by *L. sinense* seem to indicate better short-term recovery potential for herbaceous understory species than woody species, but evidence for both was mixed. Merriam and Feil (2002) took an early pass at this question by removing *L. sinense* from an invaded area and found that herbaceous species richness, herbaceous stem density, and woody stem density increased the first year following *L. sinense* control. Hanula et al. (2009) studied the effects of *L. sinense* removal more extensively in a study that involved removing *L. sinense* in 2-ha plots using mulching and hand-felling followed by herbicide. They then compared the plant communities among control plots (*L. sinense* present, no treatment), treatment plots (mulching or hand-felling), and "desired future condition" (DFC) plots that had little to no *L. sinense* cover. Two years post-treatment, they found that herbaceous cover and diversity in treated plots was higher than control plots and similar to DFC plots (except for cover in hand-felling plots, which was intermediate between controls and mulched/DFC). The treated, control, and DFC plots developed distinct community compositions by year 2 due to the proliferation of early successional species in the treatment plots. Hudson et al. (2014) returned to those same plots 3 years later (5 years post-treatment) and found that herbaceous cover and diversity were still greater in treatment plots than control plots, and similar to DFC plots. Community composition was still distinct among the control, treatment, and DFC plots, with treatment plots having more early successional species such as pokeweed (*Phytolacca americana* L.). Although these studies seem to indicate restoration potential, others have not been as successful. For example, Cutway (2017) observed little or no change in native herbaceous cover or species richness following 8 years of exotic plant removal (including *L. sinense*).

For woody species, Hudson et al. (2014) found that *L. sinense* removal did not have statistically significant effects on shrub/sapling cover, species richness or diversity 5 years after *L. sinense* control operations, although there was a marginally significant increase in cover. Woody seedlings and small saplings were included in their "herbaceous" plant category, so it is difficult to interpret changes in woody regeneration from their results. A study that continuously removed non-native plants (including *L. sinense*) similarly found no significant change in tree seedling species richness or density after 2 years (Vidra et al. 2007). This could be due to a number of factors including unrelated environmental controls (e.g., drought) or the absence of propagules from site-appropriate native species (Vidra et al. 2007). Similar to the aforementioned results related to herbaceous cover, Cutway (2017) observed no recovery of native woody plant abundance following 8 years of exotic plant removal. Although species richness did not significantly change in that study, the author did observe some species turnover throughout the study, including the addition of a few tree species.

One strategy for restoring invaded sites could be to plant native species following *L. sinense* removal. Giant cane (*Arundinaria gigantea* [Walter] Muhl.), for example, may be a good candidate species for restoration projects in invaded riparian areas. Canebrake restoration can improve water quality (Schoonover and Williard 2003), and provide habitat for a wide variety of wildlife species (Platt et al. 2013). Osland et al. (2009) tested how *L. sinense* presence affected survival and growth of transplanted cane. They found that dense *L. sinense* did not affect survival, but that cane in plots where *L. sinense* had been removed had much greater growth in the second season (roughly 2x greater ramet height and diameter, > 2x ramets per genet, and 7x higher genet area). So while control of *L. sinense* in cane restoration areas may not be absolutely required, it is likely to be highly beneficial. Fortunately, there are herbicides that can be used to control *L. sinense* without severe impacts on *A. gigantea* (Klaus and Klaus 2009).

These studies show that the results of *L. sinense* control are often variable and likely depend on other abiotic and biotic factors such as native plant propagule pressure, climate, legacy soil effects, and others. Herbaceous species can respond relatively fast to *L. sinense* removal, but initial post-treatment communities are often made up of early successional species that are dissimilar to those found in uninvaded forests (which may or may not meet land management objectives). More research is needed on the long-term successional trajectories of restored areas and possible strategies for promoting native plant recovery within the context of different land management goals.

Invasion Frameworks. There is some uncertainty regarding the causal direction of some of the correlations presented above. For instance, Hagan et al. (2014) found that *L. sinense* was more prevalent in areas with lower overstory basal area and higher soil pH, but is this because *L. sinense* is more likely to invade sites with those conditions or because *L. sinense* causes those conditions to develop? It is likely sites with low overstory basal area (possibly due to natural or anthropogenic disturbance) have more available resources for *L. sinense* to take advantage of (Hagan et al. 2014). This question is often presented in terms of whether an invasive species is a driver, passenger, or back-seat driver of ecosystem change. In other words, does the species cause changes (driver), take advantage of changes from other sources (passenger), or take advantage of initial changes and then drive further changes (back-seat driver; Foard 2014). *Ligustrum sinense* has been labeled as both a driver (Greene and Blossey 2012) and back-seat driver (Foard 2014) in the literature. The back-seat driver framework is often a good explanation for the success of *L. sinense*, as it can take advantage of local disturbances such as canopy gaps, anthropogenic edges, or altered hydrological regimes during its initial colonization (Foard 2014). However, its broad ecological tolerances (Dirr 1998) allow it to invade forest interiors as well (Hagan et al. 2014). Greene and Blossey (2012) carried out transplant experiments that involved planting false nettle (*Boehmeria cylindrica* [L.] Sw.), blunt broom sedge (*Carex tribuloides* Wahlenb.), box elder (*Acer negundo* L.) seedlings, and woodoats (*Chasmanthium latifolium* [Michx.] Yates) in plots with high *L. sinense* cover (95 ± 4.2 %), or in plots with no *L. sinense*. They found that survival of *C. tribuloides*, *A. negundo*, and *C. latifolium* was significantly lower in the *L. sinense* plots. This experimental evidence, along with the potential mechanisms of impact detailed above, provide evidence that *L. sinense* can be a driver of vegetation community change. Even so, the lack of an observed vegetation recovery in some of the *L. sinense*-removal studies implies that in some cases there may be other factors driving vegetation changes, and in these cases *L. sinense* could be more of a passenger. The lack of recovery could also be due to what Vidra et al. (2007) termed the “ghost of competition past,” referring to possible long-lasting changes in soil properties or seed banks caused by invasive species. Simple successional dynamics can also prevent short term studies from observing full community recovery. Many of the studies we found had small sample sizes and poor spatial mixing of control and invaded sites, which further complicates interpretations of correlations. Future studies should go beyond simply documenting differences between areas that are or aren’t invaded, and instead utilize study designs that allow hypotheses of causal direction to be tested.

Wildlife Impacts. Wildlife conservation is a top priority for many land owners, both public and private, so understanding how *L. sinense* affects wildlife is important for land management decisions. Some landowners may perceive *L. sinense* as beneficial for wildlife and thus may be reluctant to initiate control measures (Howle et al. 2010). Wildlife diversity is often positively correlated with vegetation structural diversity (Tews et al. 2004). Thus, we would expect that at low to moderate densities *L. sinense* could benefit wildlife communities by providing additional cover and food; however, at high densities the impacts on native vegetation become more apparent and the effect on many wildlife species may become negative.

However, it is difficult to assign a specific *L. sinense* density or cover value as the cut off between a relatively benevolent low-density invasion and a damaging high-density invasion, despite the many documented trends discussed above. This is in part due to differences in the competitive abilities of the local plant community and the different habitat preferences of various wildlife species. For the purposes of this wildlife section, we use the term "low density" invasions to describe situations in which scattered *L. sinense* individuals are present but they have no appreciable effect on native vegetation. High-density invasions refer to situations were *L. sinense* is the dominant understory or midstory species and cover of native species is severely limited. *Ligustrum sinense* in high-density stands tend to have a more upright growth form, further limiting vegetation at ground level (Figure 1.3). Moderate-density invasions are intermediate between the two, with some effects on native vegetation possible but not severe.

Invertebrates. An incredibly diverse group, invertebrates represent the majority of animal life on earth. Given this diversity, it is impossible to make broad generalizations regarding how *L. sinense* invasion may affect these taxa. However there have been a handful of studies that have directly studied the impact of *L. sinense* on invertebrate groups. For example, in research related to the previously mentioned study by Hanula et al. (2009), researchers found that treatment and DFC plots generally had higher species richness and abundance of native bees and butterflies, and that bee and butterfly richness and butterfly abundance were negatively correlated with *L. sinense* cover (Hanula and Horn 2011*a*; Hanula and Horn 2011*b*; Hudson et al. 2013). Beetle communities sampled near ground level had higher richness in treatment plots than control plots; however, communities sampled at 5 and 15 m from the ground were not different between plots with and without privet (Ulyshen et al. 2010). Earthworm abundance tended to be lower in DFC plots, but the proportion of native species was greater in DFC and treatment plots than in control plots (Lobe et al. 2014). Contrary to these findings, another study found no significant differences in ground-dwelling arthropod communities among plots invaded by *L. sinense* or *L. maackii* and control plots (Kuebbing et al. 2014).

Ligustrum sinense invasion could affect invertebrate communities through a variety of mechanisms. The reduction in native herbaceous species under *L. sinense* canopies may limit the availability of native nectar-producing plants and those suitable for hosting pollinator larva (Hanula and Horn 2011*a*). Changes to vegetation structure likely also have an impact. For instance, deer ticks (*Ixodes scapularis* Say) were found to frequently use young *L. sinense* stems when questing for hosts (Goddard 1992). However, it seems likely that at high *L. sinense* densities, where herbaceous ground cover is rare and the *L. sinense* canopy may be 5m or taller, platforms for questing may be limited. This could have implications for disease transmission in vertebrate species.

The chemical makeup of *L. sinense* may also affect invertebrates either by deterring herbivores or changing the chemical properties of the soil and water that the leaves fall onto.

Research on two species, the invasive gypsy moth (*Lymantria dispar* L.) and native lace bug (*Leptoypha mutica* Say), found that both species had relatively low performance when feeding on *L. sinense* compared to plants native to the southeastern US (Kalina et al. 2017; McEwan et al. 2009). Experiments with border privet (*Ligustrum obtusifolium* Siebold and Zucc.) have shown that the plant has high levels of chemicals in its leaves that prohibit the absorption of important nutrients by invertebrate herbivores (Konno et al. 2009), and it is possible that *L. sinense* has similar traits. This could explain why some researchers have observed lower herbivory on *L. sinense* compared to native plants and may contribute to the competitive advantage of *L. sinense* against native plant species (Greene and Blossey 2012; Morris et al. 2002). As discussed above, *L. sinense* may also be able to alter the properties of the soil that it grows in, such as pH, which could partially explain why non-native earthworms were found to have higher relative abundance under *L. sinense* canopies (Lobe et al. 2014). *Ligustrum sinense* leaves could also alter water chemistry if they fall into streams or other water sources which in turn could negatively impact aquatic invertebrates (Llewellyn 2005). Experimental evidence has shown that that *L. sinense* leaf extracts can reduce the survival of some invertebrates, however more research is necessary to determine if such effects occur in the wild (Llewellyn 2005).

Mammals. Low to moderate density *L. sinense* invasions are probably beneficial for some mammal species. White-tailed deer (*Odocoileus virginianus* Zimmermann) and American beaver (*Castor canadensis* Kuhl) show moderate selective preference for *L. sinense* browse (Rossell et al. 2014; Stromayer et al. 1998a), and studies in New Zealand and the U.S. show that various small mammals will eat the fruits (O'Malley et al. 2003; Williams et al. 2000). Winter crude protein levels of *L. sinense* browse are typically >12% (Stromayer et al. 1998*a*), which meets the winter requirements for all age classes of *O. virginianus* (Wallmo et al. 1977). Browsing on *L. sinense* is particularly heavy when acorns are limited, suggesting that it could serve as an important buffer food in fall and winter (Stromayer et al. 1998*a*). However, at high *L. sinense* densities, competition with native plants may reduce food availability for *O. virginianus* during spring and summer (Stromayer et al. 1998*a*). Longer term, the potential reduction of oak regeneration by *L. sinense* could also have negative effects on the many wildlife species that depend on acorns for food in fall and winter, even if *L. sinense* berries and browse are available as an alternative. *Ligustrum sinense* can also grow out of the browsing range of *O. virginianus* (Stromayer et al. 1998*b*), although browsing by *C. canadensis* can keep it cropped at a lower height (Rossell et al. 2014).

McCall and Walck (2014) reported that nonfood use of *L. sinense* (e.g., for cover) by mammals and birds was common. However, *L. sinense* was prevalent on their study site so the high rate of use may have been out of necessity rather than preference. When *L. sinense* forms hedges along forest edges (Figure 1.2), or is present in forest interiors at low to moderate densities, it likely provides thermal and escape cover for *O. virginianus* and other mammal species. However, as *L. sinense* reaches maturity in forest interiors, the amount of cover at ground level can be significantly reduced (Figure 1.3). In addition to impacts on terrestrial mammals, the reduction in native tree regeneration in invaded areas may pose a risk to species that roost in canopy trees, such as grey-headed flying-fox (*Pteropus poliocephalus* Temminck) in Australia (Pallin 2000). On the other hand, Christopher and Barrett (2006) found that two similar rodent species (the white-footed mouse [*Peromyscus leucopus* Rafinesque] and golden mouse [*Ochrotomys nuttalli* Harlan]) co-existed in areas with high *L. sinense* cover, and it is possible the increased vertical structure provided by *L. sinense* helped facilitate this coexistence despite the relative lack of cover at ground level. High rodent capture rates in *L. sinense* patches were also reported by Kittell (2001).

Birds. Invasion by *L. sinense* likely has complex effects on bird species assemblages due to its ability to significantly alter vegetative communities, while also offering a reliable winter food source. The small ovoid drupes of *L. sinense* are persistent throughout winter and early spring (Greenberg and Walter 2010), and some authors have speculated that the fruits could provide an important food source for frugivorous birds during this season of relative scarcity (Lochmiller 1978 [*L. vulgare*]; Miller and Miller 2005; Wilcox and Beck 2007). Researchers have found *Ligustrum* spp. fruit to be a major winter diet component of several species, including Northern Bobwhite (*Colinus virginianus* L.; McRae 1980), the Dusky-legged Guan in Argentina (*Penelope obscura* Temminck; Merler et al. 2001), and Hermit Thrush (*Catharus guttatus* Pallas; Strong et al. 2005). Preference for or against *L. sinense* is species specific: *P. obscura* appeared to select *L. sinense* at greater rates than would be expected given its availability at some sites (Merler et al. 2001), while *C. guttatus* appeared to consume *L. sinense* slightly less often than would be expected given availability (Strong et al. 2005).

Despite the documented use of *L. sinense* as a winter food source, the population- and community-level effects on bird species are relatively unknown. We would expect that species that rely on herbaceous groundcover or overstory trees for foraging, nesting, or cover would be negatively impacted by high-density invasions of *L. sinense* and its effects on those vegetation strata. The effects on species that utilize understory and midstory woody plants may be variable, depending on the specific needs of the species. For example, differences in the height and branching structure of invasive plants and the natives that they replace can impact bird nest success (e.g., Schmidt and Whelan 1999). Nesting attempts in *L. sinense* have been reported in the literature (Heckscher 2004; Wilcox and Beck 2007), but the relative quality of *L. sinense* as a nesting substrate has not be studied.

We identified only one study that used an observational design with multiple replicates to compare bird use of sites with a range of *L. sinense* cover. Wilcox and Beck (2007) found significant positive correlations between *L. sinense* cover and bird species richness and abundance during winter, further supporting the hypothesis that birds will use *L. sinense* for cover and/or food. Interestingly, the majority of birds observed in dense *L. sinense* areas were not frugivorous. Wilcox and Beck (2007) also documented differential use patterns of *L. sinense* invaded areas by birds with different ecological niches, suggesting that *L. sinense* invasion could alter local bird community composition. Additionally, Wilcox and Beck (2007) observed that singing during summer tended to be less common in plots with high-density *L. sinense*, possibly suggesting that mate attraction is more difficult in high-density areas or that breeding territories in high-density areas are not preferred. As discussed in the mammal section above, McCall and Walck (2014) reported that nonfood use of *L. sinense* by birds and/or mammals was common, however whether individuals select for or against *L. sinense* requires further study. Woodcock (*Scolopax rusticola* L.) are known to use areas invaded by *L. sinense* during fall migration and as winter habitat (Miller and Miller 2005; Myatt and Krementz 2007), although whether these are preferred vegetation types is unclear.

The additional food that *L. sinense* provides in winter could have complex effects on global ecological patterns by enticing migrating birds to overwinter in areas that they wouldn’t otherwise, although such effects are speculative. Researchers have suggested that *Ligustrum* spp. may provide such an incentive for species in southern Argentina that would normally overwinter farther north (Merler et al. 2001; Montaldo 1993). In North America it is possible that supplementary feeding stations (i.e., bird feeders) could affect the timing of migrations (Robb et al. 2008), and if this is the case *L. sinense* may have complimentary or additive effects. However, *L. sinense* fruits might ripen too late to be much of an enticement for early fall migrants (McCall and Walck 2014). Species such as American robin (*Turdus migratorius* L.) are known to feed on *L. sinense* fruits on their northbound trip in spring (Miller and Miller 2005), but it seems unlikely that this would alter spring migration patterns. The effects of additional winter food may also interact with or be swamped out by the impacts of climate change on migrations (e.g., Jenni and Kery 2003; Zaifman et al. 2017).

Herpetofauna. The effects of invasive plants on herpetofauna (i.e., reptiles and amphibians) are understudied in general, and there have been no studies, to our knowledge, on the effects of *L. sinense* specifically. Martin and Murray (2011) reviewed the limited available literature on invasive plant impacts on herpetofauna and developed a general predictive framework of possible impacts to herpetofauna habitat quality, food availability, and reproductive success. They predicted that invasive plants that are structurally different than the native plant assemblage will have the greatest effects, and that small bodied herpetofauna with small home ranges will be most affected (either positively or negatively) because they will have greater difficulty moving to uninvaded areas. The general structure of *L. sinense* is not necessarily unique to the systems it commonly invades, such as bottomland hardwood forests, however it seems to reach much higher densities across larger areas than native understory and midstory species.

Many herpetofaunal species are "heliothermic," meaning that they bask in sunlight to aid in thermoregulation (Bogert 1959). Some species, such as freshwater turtles, also choose nest sites with the necessary sun exposure to maintain proper nest temperatures (Bodie et al. 1996). The published data are not clear on the degree to which *L. sinense* invasion reduces light levels at ground level (e.g., Brantley 2008; Osland et al. 2009; Pokswinki 2009), but it seems likely that basking and nest site availability would be negatively correlated to *L. sinense* cover. The first step in exploring how *L. sinense* could affect herpetofaunal thermoregulation is to better quantify light level differences among sites with varying levels of *L. sinense* and native vegetation cover.

Herpetofauna need cover to aid in thermoregulation, moisture retention, and predator avoidance. Common cover types include leaf litter, coarse woody debris (CWD), burrows, and herbaceous vegetation. While the dense midstory that *L. sinense* forms may provide shelter from aerial predators and direct sunlight, it is unlikely to provide adequate shelter from terrestrial predators or extreme weather. Leaf litter, a particularly important cover type for salamanders, may be reduced by *L. sinense* invasion due to its ability to intercept leaves from the canopy layer (Faulkner et al. 1989), increase decomposition rates (Mitchell et al. 2011), and suppress overstory regeneration (e.g., Hart and Holmes 2013). However, Hagan et al. (2014) found that litter depth was positively correlated with *L. sinense* invasion, possibly because leaf litter tends to accumulate in the flatter microsites that *L. sinense* preferred in their study area. Limited evidence shows that cover of CWD may be greater in areas with *L. sinense*, possibly due to increased self-thinning by resource constrained trees (Foard et al. 2016). However, this increase in CWD cover availability may be temporary if overstory tree regeneration is impeded. Herbaceous ground cover reductions associated with *L. sinense* (e.g., Wilcox and Beck 2007) further limit thermal and escape cover for herpetofauna.

Conclusions. Our literature review revealed many well-documented correlations between depauperate native plant communities and *L. sinense* invasion, although more research is needed to confirm the degree to which *L. sinense* is driving these declines and the mechanisms by which it does so. Impacts to the understory are perhaps the most visible today, but long-term reductions in overstory regeneration could dramatically alter future landscapes. Research on wildlife impacts are limited, but changes to vegetation species composition and structure will likely affect wildlife in a variety of ways (some positive but many negative). Ecosystems are made up of complex interactions among species, including competition, herbivory, predation, parasitism, and mutualism. When an invasive species such as *L. sinense* causes substantial changes to the structure and diversity of a system, it can be difficult to measure or predict the resulting changes.

Management recommendations. Low-density *L. sinense* invasions likely have limited negative consequences for plant species, and may have positive effects on some wildlife species (due to increased structural complexity and food resources). The negative impacts of *L. sinense* invasion—i.e., suppression of native plants and the predicted reductions in wildlife habitat quality—tend to be most apparent at higher *L. sinense* densities. However, low-density invasions may be more cost-effective to control than high-density invasions (Benez-Secancho et al. 2018). For this reason, we recommend taking early action to control *L. sinense* when possible in order to prevent future expansion and excessively expensive control operations. Whenever *L. sinense* control operations are implemented, particularly in high-density invasions, the response of native vegetation should be monitored and reported (when possible) to improve our understanding of the best practices for restoring invaded areas. Maddox et al. (2010) and Urbatsch (2000) provide recommendations on control options, and many state Extension agencies also provide guidance (e.g., Enloe and Loewenstein 2018).

Research recommendations. A substantial amount of evidence has accumulated over the past several decades regarding the ecological impacts of *L. sinense* invasion, however we identified several critical knowledge gaps worthy of future study. There is a particular need for additional research on the effects of *L. sinense* invasion on wildlife species. With the exception of a few studies on invertebrates, most research to-date has simply documented use patterns of invaded areas. More research is needed on how *L. sinense* invasion impacts vertebrate species during various seasons and life-history stages, and how potential impacts affect population dynamics and community composition. The impacts of *L. sinense* on wildlife are likely tied to the local scale at which *L. sinense* is able to form high-density stands compared to the daily movement and home-range scale of different wildlife species. Future studies should be conducted at the appropriate scale for the wildlife taxa in question. There is also a need for additional research on the effects of *L. sinense* on overstory trees. Although numerous studies have shown that *L. sinense* can negatively impact the woody regeneration layer, more research is needed to determine the spatial and temporal scales at which conversions from forest to shrubland may occur. Impacts on mature trees are also important to understand, as this could speed forest conversion and has implications for land owners managing for timber or wildlife. With respect to the invasion frameworks discussed above (driver, backseat driver, or passenger), additional research could help establish guidelines for recognizing when *L. sinense* is likely a “driver” of ecological change vs. a “passenger,” which would help land managers prioritize their restoration actions to ensure they are optimizing their restoration and management activities. Finally, further research into restoration options is needed so that land managers have the best tools at their disposal for managing *L. sinense* and have realistic expectations for how the forest plant communities will respond (particularly in situations where *L. sinense* is operating as a “driver” of change). This will likely require further research into the mechanisms by which *L. sinense* impacts native plants so that potential legacy effects can be mitigated, or at least anticipated and planned for in management plans.

References

Aerts R, Berendse F (1988) The effect of increased nutrient availability on vegetation dynamics in wet heathlands. Vegetatio 76: 63-69

Barksdale WF, Anderson CJ (2015) The influence of land use on forest structure, species composition, and soil conditions in headwater-slope wetlands of coastal Alabama, USA. International Journal of Biodiversity Science, Ecosystem Services & Management 11(1): 61-70

Barnett JM, Hudack PF, Dick GO (2016) Removing Chinese privet (*Ligustrum sinense* Lour.) and establishing native plants in a flooded riparian corridor in north-central Texas, USA. Environmental Quality Management. Winter 2016: 121-129 DOI: 10.1002/tqem.21489

Benez-Secanho FJ, Grebner DL, Ezell AW, Grala RK (2018) Financial trade-offs associated with controlling Chinese privet (*Ligustrum sinense* Lour.) in forestlands in the southern USA. Journal of Forestry 116(3): 236-244

Bodie JR, Smith KR, Burke VJ (1996) A comparison of diel nest temperature and nest site selection for two sympatric species of freshwater turtles. The American Midland Naturalist 136(1): 181-186

Bogert CM (1959) How reptiles regulate their body temperature. Scientific American 200(4): 105-120

Borth EB, Custer KW, McEwan RW (2018) Lethal effects of lead leachate from the non-native invasive shrub Amur honeysuckle (*Lonicera maackii*) on a model aquatic organism (*Hyalella azteca*). Écoscience DOI: 10.1080/11956860.2018.1426261

Brantley EF (2008) Influence of Chinese privet (*Ligustrum sinense* Lour.) on riparian forests of the southern Piedmont: net primary productivity, carbon sequestration, and native plant regeneration. PhD Dissertation. Auburn, AL: Auburn Univeristy. 239 p

Brown CE, Pezeshki SR (2000) A study on waterlogging as a potential tool to control *Ligustrum sinense* populations in western Tennessee. Wetlands 20(3): 429-437

Burton ML, Samuelson LJ (2008) Influence of urbanization on riparian forest diversity and structure in the Georgia Piedmont, US. Plant Ecology 195:99-115

Burton ML, Samuelson LJ, Pan S (2005) Riparian woody plant diversity and forest structure along an urban-rural gradient. Urban Ecosystems 8:93-106

CABI (2018) *Ligustrum sinense* [original text by D Shaw]. In: Invasive Species Compendium. Wallingford, UK: CAB International. https://www.cabi.org/isc/datasheet/30763

Chittka L, Schürkens S (2001) Successful invasion of a floral market. Nature 411: 653

Christopher CC, Barrett GW (2006) Coexistence of white-footed mice (*Peromyscus leucopus*) and golden mice (*Ochrotomys nutalli*) in a southeastern forest. Journal of Mammalogy 87(1): 102-107

Cofer MS, Walck JL, Hidayati SN. (2008) Species richness and exotic species invasion in middle Tennessee cedar glades in relation to abiotic and biotic factors. The Journal of the Torrey Botanical Society 135(4): 540-553

Cutway HB (2017) Effects of long-term manual invasive plant removal on forest understory composition. Natural Areas Journal 37(4): 530-539

Deyn GBD, Raaijmakers CE, Van der Putten WH (2004) Plant community development is affected by nutrients and soil biota. Journal of Ecology 92: 824-834

Dirr MA (1998) Manual of woody landscape plants: their identification, ornamental characteristics, culture, propagation and uses. 5th ed. Champaign, IL: Stipes Publishing LLC. p 563

EDDMapS (2019) Early detection and distribution mapping system. The University of Georgia-Center for Invasive Species and Ecosystem Health. https://www.eddmaps.org/distribution/usstate.cfm?sub=3035; last accessed June 24, 2019

Ehrenfeld JG (2003) Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6:503-523

Ehrenfeld JG (2010) Ecosystem consequences of biological invasions. Annual Review of Ecology, Evolution, and Systematics 41:59-80

Enloe S, Loewenstein N (2018) Control options for Chinese privet. Alabama Cooperative Extension System. https://www.aces.edu/blog/topics/control-invasive-plants/control-options-for-chinese-privet/

Faulkner JL, Clebsch EEC, Sanders WL (1989) Use of prescribed burning for managing natural and historic resources in the Chickamauga and Chattanooga National Military Park, USA. Environmental Management 13(5): 603-612

Foard M (2014) Causes and Consequences of Chinese privet (*Ligustrum sinense* Lour.) invasion in hydrologically altered forested wetlands. Master's thesis. Jonesboro, AR: Arkansas State University. 172 p

Foard M, Burnette DJ, Burge DRL, Marsico TD (2016) Influence of river channelization and the invasive shrub, *Ligustrum sinense*, on oak (*Quercus* spp.) growth rates in bottomland hardwood forests. Applied Vegetation Science 19:401-412

Gioria M, Osborne BA (2014) Resource competition in plant invasions: emerging patterns and research needs. Frontiers in Plant Science 5: Article 501 DOI: 10.3389/fpls.2014.00501

Goddard J (1992) Ecological studies of adult *Ixodes scapularis* in central Mississippi: questing activity in relation to time of year, vegetation type, and meteorological conditions. Journal of Medical Entomology 29(3): 501-506

Greene BT, Blossey B (2012) Lost in the weeds: *Ligustrum sinense* reduces native plant growth and survival. Biological Invasions 14:139-150

Greenberg CH, Walter ST (2010) Fleshy fruit removal and nutritional composition of winter-fruiting plants: a comparison of non-native invasive and native species. Natural Areas Journal 30(3): 312-321

Greene B, Blossey B (2014) Patterns of privet: urbanizing watersheds, invasive *Ligustrum sinense*, and performance of native plant species in Piedmont floodplain forests. Ecosystems 17: 990-1001

Greipsson S, DiTommaso A (2006) Invasive non-native plants alter the occurrence of arbuscular mycorrhizal fungi and benefit from this association. Ecological Restoration 24(4): 236-241

Grove E, Clarkson BD (2005) An ecological study of Chinese privet (*Ligustrum sinense* Lour.) in the Waikato region. Hamilton, New Zealand: Centre for Biodiversity and Ecology Research Contract Report No. 41. 17 p

Hagan DL, Mikhailova EA, Shearman TM, Ma PT, Nankaya JS, Hart SK, Valdetero HE, Bridges WC, Yun H (2014) The role of soil and landscape factors in Chinese privet (*Ligustrum sinense*) invasion in the Appalachian Piedmont. Invasive Plant Science and Management 7(3): 483-490

Hanula JL, Horn S (2011*a*) Removing an exotic shrub from riparian forests increases butterfly abundance and diversity. Forest Ecology and Management 262: 674-680

Hanula JL, Horn S (2011*b*) Removing an invasive shrub (Chinese privet) increases bee diversity and abundance in riparian forests of the southeastern United States. Insect Conservation and Diversity 4: 275-283

Hanula JL, Horn S, Taylor JW (2009) Chinese privet (*Ligustrum sinense*) removal and its effect on native plant communities of riparian forests. Invasive Plant Science and Management 2:292-300

Hart JL, Holmes BN (2013) Relationships between *Ligustrum sinense* invasion, biodiversity, and development in a mixed bottomland forest. Invasive Plant Science and Management 6(1): 175-186

Heckscher CM (2004) Veery nest sites in a mid-Atlantic Piedmont forest: vegetative physiognomy and use of alien shrubs. The American Midland Naturalist 151(2): 326-337

Hierro JL, Callaway RM (2003) Allelopathy and exotic plant invasion. Plant and Soil 256:29-39

Howle Mb, Straka TJ, Nespeca MC (2010) Family forest owners' perceptions on chemical methods for invasive species control. Invasive Plant Science and Management 3:253-261

Hoyos LE, Gavier-Pizarro GI, Kuemmerle T, Bucher EH, Radeloff VC, Tecco PA (2010) Invasion of glossy privet (*Ligustrum lucidum*) and native forest loss in the Sierras Chicas of Córdoba, Argentina. Biological Invasions 12:3261-3275

Hudson JR, Hanula JL, Horn S (2013) Removing Chinese privet from riparian forests still benefits pollinators five years later. Biological Conservation 167: 355-362

Hudson JR, Hanula JL, Horn S (2014) Impacts of removing Chinese privet from riparian forests on plant communities and tree growth five years later. Forest Ecology and Mangement 324: 101-108

Jenni L, Kéry M (2003) Timing of autumn bird migration under climate change: advances in long-distance migrants, delays in short-distance migrants. Proceeding of the Royal Society B 270: 1467-1471

Kalina J, Braman SK, Hanula JL (2017) Host utlilization of Chinese privet (Lamiales: Oleaceae) and host choice by *Leptoypha mutica* (Hemiptera: Tingidae). Journal of Entomological Science 52(2): 141-153

Kittel MM (2001) Relationship among invasive Chinese privet, plant diversity, and small mammal captures in southeastern deciduous forests. Master's thesis. Clemson, SC: Clemson University. 35p

Klaus NA, Klaus JM (2009) Evaluating tolerance of herbicide and transplantation by cane (a native bamboo) for canebrake restoration. Restoration Ecology 19(3): 344-350

Klepac J, Rummer RB, Hanula JL, Horn S (2007) Mechanical removal of Chinese privet. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station SRS-43

Konno K, Hirayama C, Shinbo H, Nakamura M (2009) Glycine addition improves feeding performance of non-specialist herbivores on the privet, *Ligustrum obtusifolium*: *In vivo* evidence for the physiological impacts of anti-nutritive plant defense with iridoid and insect adaptation with glycine. Applied Entomology and Zoology 44(4): 595-601

Kuebbing SE, Classen AT, Call JJ, Henning JA, Simberloff D (2015) Plant-soil interactions promote co-occurrence of three nonnative woody shrubs. Ecology 96(8): 2289-2299

Kuebbing SE, Classen AT, Simberloff D (2014) Two co-occurring invasive woody shrubs alter soil properties and promote subdominant invasive species. Journal of Applied Ecology 51: 124-133

Kuebbing SE, Patterson CM, Classen AT, Simberloff D (2016) Co-occurring nonnative woody shrubs have additive and non-additive soil legacies. Ecological Applications 26(6): 1896-1906

Lichstein JW, Grau HR, Aragón R (2004) Recruitment limitation in secondary forests dominated by an exotic tree. Journal of Vegetation Science 15(6): 721-728

Lin J, Harcombe PA, Fulton MR, Hall RW (2002) Sapling growth and survivorship as a function of light in a mesic forest of southeast Texas, USA. Oecologia 132: 428-435

Llewellyn DC (2005) Effect of toxic riparian weeds on the survival of aquatic invertebrates. Australian Zoologist 33(2): 194-209

Lobe JW, Callaham MA Jr., Hendrix PF, Hanula JL (2014) Removal of an invasive shrub (Chinese privet: *Ligustrum sinense* Lour) reduces exotic earthworm abundance and promotes recovery of native North American earthworms. Applied Soil Ecology 83: 133-139

Lochmiller RL (1978) Privet as a potential winter food supplement for songbirds. Bird-Banding 49(3): 279-280

Lorence DH, Sussman RW (1988) Exotic species invasion into Mauritius wet forest remnants. Journal of Tropical Ecology 2(2): 147-162

Lowenstein NJ, Lowenstein EF (2005) Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. Urban Ecosystems 8:79-91

Maddox V, Byrd J Jr., Serviss B (2010) Identification and control of invasive privets (*Ligustrum* spp.) in the middle southern United States. Invasive Plant Science and Management 3(4): 482-488

Martin LJ, Murray BR (2011) A predictive framework and review of the ecological impacts of exotic plant invasions on reptiles and amphibians. Biological Reviews 86: 407-419

McAlpine KG, Timmins SM, Jackman SD, Lamoureaux SL (2018) Composition of the understory in 132 woody weed populations and implications for succession. New Zealand Journal of Ecology 42(2): 277-283

McCall LJ, Walck JL (2014) Dispersal characteristics of two native and two nonnative fleshy-fruited sympatric shrubs. Castanea 79(2): 88-99

McEwan RW, Rieske LK, Arthur MA (2009) Potential interactions between invasive woody shrubs and the gypsy moth (*Lymantria dispar*), an invasive insect herbivore. Biological Invasions 11: 1053-1058

McRae WA (1980) Unusual bobwhite foods on abandoned Piedmont farmlands. Georgia Journal of Science. 38: 49-54

Merler JA, Diuk-Wasser MA, Quintana RD (2001) Winter diet of dusky-legged guan (*Penelope obscura)* at the Paraná River Delta region. Studies on Neotropical Fauna and Environment 36(1): 33-38

Merriam RW (2003) The abundance, distribution and edge associations of six non-indigenous, harmful plants across North Carolina. The Journal of the Torrey Botanical Society 130(4): 283-291

Merriam RW, Feil E (2002) The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions 4: 369-373

Miller JH, Chambliss EB (2008) Estimates of acres covered of nonnative invasive plants in southern forests. Auburn, AL: US Department of Agriculture, Forest Service, Southern Research Station. https://www.invasive.org/fiamaps/summary.pdf

Miller JH, Chambliss EB, Bargeron CT (2004) Invasive plants of the thirteen southern states. https://www.invasive.org/south/seweeds.cfm?sort=3

Miller JH, Miller KA (2005) Forest plants of the Southeast and their wildlife uses. revised ed. Athens, GA: University of Georgia Press. p 360

Mitchell JD, Lockaby BG, Brantley EF (2011) Influence of Chinese privet (*Ligustrum sinense*) on decomposition and nutrient availability in riparian forests. Invasive Plant Science and Management 4: 437-447

Montaldo NH (1993) Dispersión por aves y éxito reproductive de dos especies de *Ligustrum* (Oleaceae) en un relicto de selva subtropical en la Argentina. Revista Chilena de Historia Natural 66:75-85

Morris LL, Walck JL, Hidayati SN (2002) Growth and reproduction of the invasive *Ligustrum sinense* and native *Forestiera ligustrina* (Oleaceae): implications for the invasion and persistence of a nonnative shrub. International Journal of Plant Sciences 163(6): 1001-1010

Myatt NA, Krementz DG (2007) Fall migration and habitat use of American woodcock in the central United States. The Journal of Wildlife Management 71(4): 1197-1205

Nowacki GJ, Abrams MD (2008) The demise of fire and “mesophication” of forests in the eastern United States. Bioscience 58(2):123-138

O’Malley M, Blesh J, Williams M, Barrett GW (2003) Food preferences and bioenergetics of the white-footed mouse (*Peromyscus leucopus*) and the golden mouse (*Ochrotomys nuttalli*). Georgia Journal of Science 61(4): 233-237

Osland MJ, Pahl JW, Richardson CJ (2009) Native bamboo [*Arundinaria gigantea* (Walter) Muhl., Poaceae] establishment and growth after the removal of an invasive non-native shrub (*Ligustrum sinense* Lour., Oleaceae): implications for restoration. Castanea 74(3): 247-258

Pallin N (2000) Ku-ring-gai flying-fox reserve. Ecological Management & Restoration 1(1): 10-20

Platt SG, Rainwater TR, Elsey RM, Brantley CG (2013) Canebrake fauna revisited: additional records of species diversity in a critically endangered ecosystem. The Journal of the American Bamboo Society 26(1): 1-12

Pokswinki SM (2009) Invasive characteristics of Chinese privet (*Ligustrum sinense* Lour.) in a bay swamp in the fall line hills of east-central Alabama. Master’s thesis. Auburn, AL: Auburn University. 69 p

Robb GN, McDonald RA, Chamberlain DE, Bearhop S (2008) Food for thought: supplementary feeding as a driver of ecological change in avian populations. Frontiers in Ecology and the Environment 6(9): 476-484

Rossell CR Jr., Arico S, Clarke HD, Horton JL, Ward JR, Patch SC (2014) Forage selection of native and nonnative woody plants by beaver in a rare-shrub community in the Appalachian Mountains of North Carolina. Southeastern Naturalist 13(4): 649-662

Schmidt KA, Whelan CJ (1999) Effects of exotic *Lonicera* and *Rhamnus* on songbird nest predation. Conservation Biology 13(6): 1502-1506

Schnell D, Catling P, Folkerts G, Frost C, Gardner R, et al. (2000) *Sarracenia oreophila. in*: The IUCN Red List of Threatened Species http://dx.doi.org/10.2305/IUCN.UK.2000.RLTS.T39718A10260069.en

Schoonover JE, Williard KWJ (2003) Ground water nitrate reduction in giant cane and forest riparian buffer zones. Journal of the American Water Resources Association 39(2): 347-354

Sikkema JJ, Boyd JN (2015) Impacts of invasive nonnative plant species on the rare forest herb *Scutellaria montana*. Acta Oecologica 69:182-191

Smith KE, Runion GB, Prior SA, Price AJ, Rogers HH, Torbert HA (2008) Chinese privet (*Ligustrum sinense*) in an elevated CO_2 environment. Botany Research Journal 1(2): 43-48

Stocker R, Hupp KVS (2008) Chapter 6: fire and nonnative invasive plants in the Southeastern Bioregion. Pages 91-112 *in* Zouhar K, Smith JK, Sutherland S, Brooks M. Wildland fire in ecosystems: fire and nonnative invasive plants. Vol. 6. Ogden, UT: US Department of Agriculture, Forest Service, Rocky Mountain Research Station Gen. Tech. Rep. RMRS-GTR-42. 355 p

Stromayer KAK, Warren RJ, Harrington TB (1998*b*) Managing Chinese privet for white-tailed deer. Southern Journal of Applied Forestry 22(4): 227-230

Stromayer KAK, Warren RJ, Johnson AS, Hale PE, Rogers CL, Tucker CL (1998*a*) Chinese privet and the feeding ecology of white-tailed deer: the role of an exotic plant. The Journal of Wildlife Management 62(4): 1321-1329

Strong CM, Brown DR, Stouffer PC (2005) Frugivory by wintering hermit thrush in Louisiana. Southeastern Naturalist 4(4): 627-638

Swarbrick JT, Timmins SM, Bullen KM (1999) The biology of Australian weeds. 36. *Ligustrum lucidum* Aiton and *Ligustrum sinense* Lour. Plant Protection Quarterly 14(4): 122-130

Tiller MB (2015) Effects of yaupon, Chinese privet, and Chinese tallow on understory fuel flammability in east Texas hardwood and pine ecosystems. Master's thesis. Nacogdoches, TX: Stephen F. Austin State University. 171 p

Traveset A, Richardson DM (2006) Biological invasions as disruptors of plant reproductive mutualisms. TRENDS in Ecology and Evolution 21(4): 208-216

Ulyshen MD, Horn S, Hanula JL (2010) Response of beetles (Coleoptera) at three heights to the experimental removal of an invasive shrub, Chinese privet (*Ligustrum sinense*), from floodplain forests. Biological Invasions 12: 1573-1579

Urbatsch L (2000) Chinese privet *Ligustrum sinense* Lour. Baton Rouge, LA: US Department of Agriculture, Natural Resources Conservation Service Plant Guide. 5 p

[USFWS] US Fish and Service (1996) Technical/ agency draft recovery plan for fringed campion, *Silene polypetala* (Walt.) Fern. & Schub. Atlanta, Georgia. 32 p.

Vidra RL, Shear TH, Stucky JM (2007) Effects of vegetation removal on native understory recovery in an exotic–rich urban forest. The Journal of the Torrey Botanical Society 134(3): 410-419

Wallmo OC, Carpenter LH, Regelin WL, Gill RB, Baker DL (1977) Evaluation of deer habitat on a nutritional basis. Journal of Range Management 30(2): 122-127

Wang H-H, Grant WE (2012) Determinants of Chinese and European privet (*Ligustrum sinense* and *Ligustrum vulgare*) invasion and likelihood of further invasion in the southern US forestlands. Invasive Plant Science and Management 5(4): 454-463

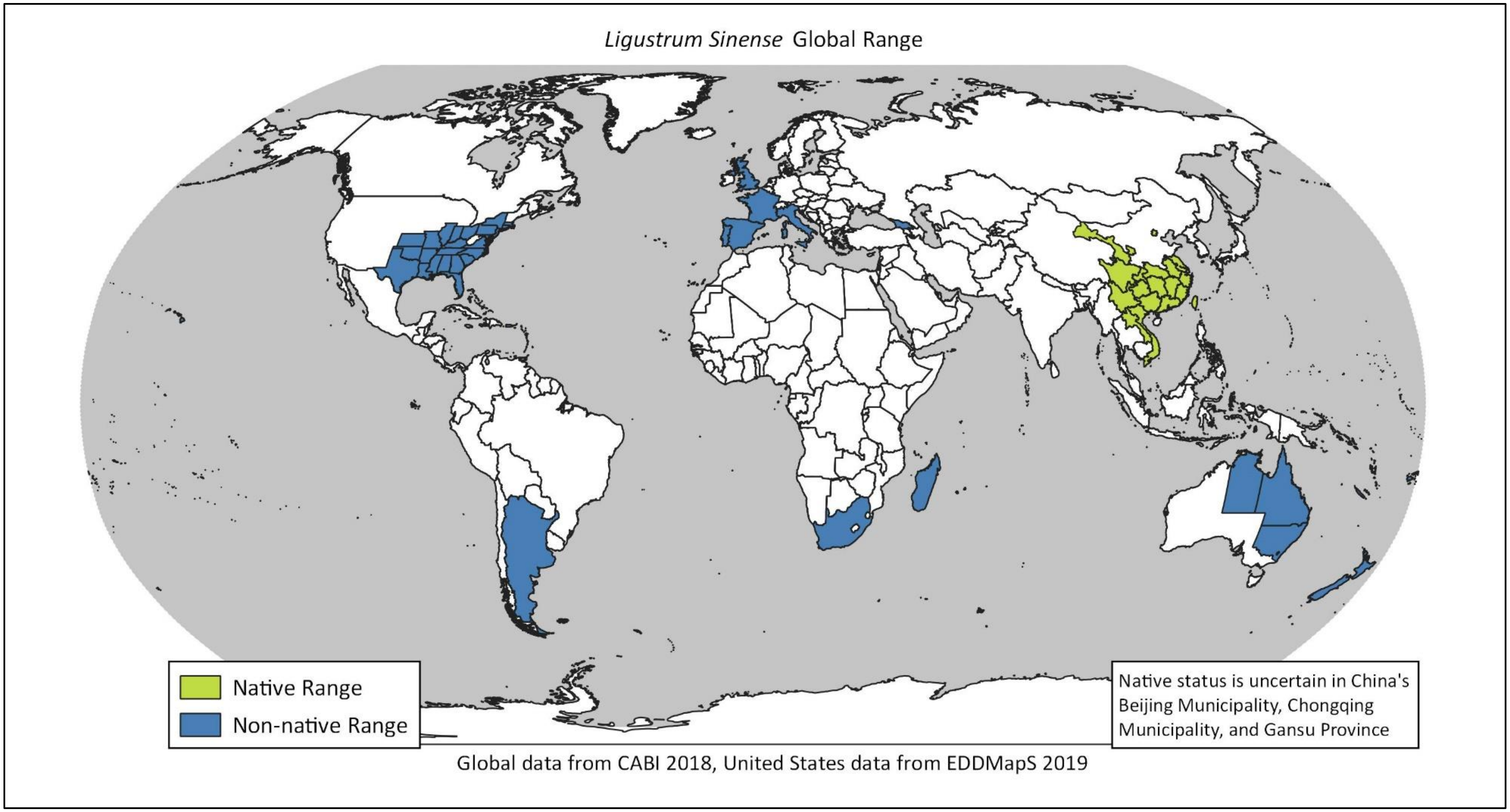
Wang H-H, Wonkka CL, Grant WE, Rogers WE (2016) Range expansion of invasive shrubs: implication for crown fire risk in forestlands of the southern USA. AoB Plants 8: plw012; DOI:10.1093/aobpla/plw012

Westoby M, Dalby J, Adams-Acton L (1983) Fruit production by two species of privet, *Ligustrum sinense* Lour. and *L. lucidum* W.T. Ait. in Sydney. Australian Weeds 2(4): 127-129

Wilcox J, Beck CW (2007) Effects of *Ligustrum sinense* Lour. (Chinese privet) on abundance and diversity of songbirds and native plants in a southeastern nature preserve. Southeastern Naturalist 6(3): 535-550

Williams PA, Karl BJ, Bannister P, Lee WG (2000) Small mammals as potential seed dispersers in New Zealand. Austral Ecology 25: 523-532

Zaifman J, Shan D, Ay A, Jimenez AG (2017) Shifts in bird migration timing in North American long-distance and short-distance migrants are associated with climate change. International Journal of Zoology Article ID 6025646, DOI: 10.1155/2017/602564



Ligustrum Sinense Global Range

Native status is uncertain in China's Beijing Municipality, Chongqing Municipality, and Gansu Province

Global data from CABI 2018, United States data from EDDMapS 2019

Figure 1.1. Global range of *Ligustrum sinense*. Data for the United States, Australia, and China are displayed at the state/province level. All other data are country-wide. Ligustrum sinense is considered “invasive” in only a portion of its non-native range. Records retrieved from CABI 2018 and EDDMapS 2019

Figure 1.2. Example of a *Ligustrum sinense* invasion along a forest edge in Alabama. Photo taken March 15, 2018.

Figure 1.3. Example of the understory of a bottomland hardwood forest in Alabama invaded by *Ligustrum sinense*. Photo courtesy of Jimmy Stiles, June 14, 2019

Chapter 2

Evaluating a Free and Simple Remote Sensing Method for Mapping *Ligustrum sinense* Invasions in Hardwood Forests

Abstract

Chinese privet (*Ligustrum sinense* Lour.) is a common invasive shrub in hardwood forests of the southeastern US and has been shown to negatively affect native herbaceous and woody plants. The ability to map the distribution of *L. sinense* on a property could help land managers plan and budget for control operations. We evaluated whether freely available moderate resolution multispectral imagery (Landsat 8 and Sentinel 2) and open-source GIS software (QGIS with the Semi-Automatic Classification Plugin) could be effective tools for this application. We tested how satellite type, image acquisition date, classification algorithm, and *L. sinense* cover affected detection accuracy. We found that Sentinel 2 imagery in March tended to produce good results, especially when analyzed using the maximum likelihood algorithm. Our best classifier obtained an overall accuracy of 92.3% for areas with ≥40% *L. sinense* cover. We recommend that land managers interested in applying this tool use an adaptive process for developing training polygons and test multiple images and classification algorithms in order to achieve optimal results.

Introduction

Chinese privet (*Ligustrum sinense* Lour.) is an invasive shrub with a broad global range outside its native distribution (CABI 2018). It is particularly problematic in the southeastern US, where it and congeneric European privet (*L. vulgare* L.) were estimated in 2008 to cover over a million hectares (Miller and Chambliss 2008; Miller et al. 2004). *Ligustrum sinense* can outcompete native plant species, potentially degrading wildlife habitat and limiting forest regeneration. Control costs are generally high ($216-$1820 per ha; Benez-Secancho et al. 2018; Klepac et al. 2007), necessitating careful planning and budgeting on behalf of land managers who are interested in forest restoration. The objective of this study was to evaluate whether free satellite imagery and simple to use open source software could be an effective tool for land managers who need to map *L. sinense* invasions to help plan hardwood forest restoration projects.

Ligustrum sinense was introduced to the southeastern US for landscaping in 1852 and has since spread throughout the region, primarily through endozoochory and hydrochory (Foard 2014; Maddox et al. 2010; Miller and Miller 2005). Individuals can have a single or multi-stemmed growth form and may reach 10m tall (Maddox et al. 2010; Miller and Miller 2005). The phenology of the plant is variable depending on the local climate and it has been described as evergreen or semi-evergreen (Maddox et al. 2010). Negative correlations between *L. sinense* abundance and native plant abundance and diversity have been documented by many studies (e.g. Greene and Blossey 2012; Wilcox and Beck 2007), and some authors are concerned that the lack of woody regeneration under *L. sinense* canopies could lead to severe forest degradation over time (Green and Blossey 2012; Merriam and Feil 2002; Lowenstein and Lowenstein 2005; Hart and Holmes 2013). The plant has broad environmental tolerances and can be found in upland and bottomland sites (Cofer et al. 2008; Grove and Clarkson 2005; Miller and Miller 2005; Pokswinki 2009).

Public and private land managers who are interested in controlling *L. sinense* would benefit from being able to estimate the acreage requiring treatment on a particular property so that costs can be calculated and budgeted for. On large properties it would be time consuming and difficult to determine the invaded acreage based solely on field surveys. In situations where *L. sinense* is growing under a deciduous hardwood overstory, the phenological differences between *L. sinense* and the overstory can be exploited during the dormant season to map *L. sinense* coverage using satellite or aerial data. Ward (2002) took advantage of these phenological differences to map *L. sinense* based on manual interpretation of 1-m resolution color infrared or black and white aerial photographs. This method seemed to be relatively successful, although Ward (2002) did not conduct a formal accuracy assessment. However, there are some notable downsides to this approach. Manual photo interpretation is time consuming and accuracy is highly dependent on the skill of the interpreter. Additionally, high resolution leaf-off imagery is not always freely available, possibly requiring data to be purchased.

Singh et al. (2015) utilized a more data-intensive approach for mapping *L. sinense* presence. They used 1-m resolution LiDAR (light detection and ranging) data and 1-m resolution leaf-off color infrared IKONOS imagery (both resampled to 5 m) to create 80 model variables (43 canopy and 23 topographic metrics derived from LiDAR and 14 spectral metrics derived from IKONOS imagery). These variables were used in logistic regression and random forest (RF) classification models. The best performing models were RF models based on LiDAR derived metrics, which took into consideration vegetation structure, topography, and spectral characteristics. The downside to this method is that LiDAR is not always freely available and can be expensive to acquire (Kelly and Tommaso 2015). There is also a relatively high level of technical expertise needed to process LiDAR data and run RF classifiers in a programming language such as R. The cost and expertise required to implement this technique may serve as a barrier to its implementation by land managers.

Fortunately there are free and easier to use data sources that could be used for mapping *L. sinense*. Moderate resolution, multispectral satellite imagery is commonly used for land cover mapping (Gómez et al. 2016; Phiri and Morgenroth 2017; Xie et al. 2008), including invasive plant detection (e.g., Resasco et al. 2007). These satellite sensors measure the reflectivity of the earth’s surface at multiple wavelengths, or bands, of the electromagnetic spectrum. This includes the visible spectrum (i.e., blue, green, and red), as well as the wavelengths outside the visible spectrum such as infrared. Different land cover types reflect sunlight with varying intensities across the electromagnetic spectrum due to variation in pigmentation, texture, water content, and other factors (Knipling 1970, NASA 1999). These differences in reflectivity, known as spectral signatures, can be used to distinguish among land cover types (Knipling 1970, NASA 1999). Healthy vegetation is particularly easy to distinguish, versus non-vegetated areas or dormant vegetation, due to the near-infrared reflecting properties of leaf cell tissues (Knipling 1970). Moderate resolution multispectral imagery is provided free to the public through the United States’ Landsat and European Space Agency’s Sentinel 2 (S2) programs (ESA 2015, USGS 2019). Landsat 8 (L8), the most recent iteration of the Landsat series, uses its onboard Operational Land Imager to collect 9 band imagery at 30-m spatial resolution (except for the 15-m panchromatic band; USGS 2019). Sentinel 2 uses its Multispectral Instrument to collect 13 band imagery at resolutions of 10-, 20-, and 60-m (ESA 2015).

Singh et al. (2018) tested the effectiveness of mapping *L. sinense* in North Carolina using Landsat 5 imagery and a RF classifier implemented in the R statistical software. They tested a range of models that included various combinations of Landsat bands, vegetation indices based on the Landsat bands, and topographic indices based on digital elevation models. They found that imagery from early to mid-March captured the greatest phenological differences between *L. sinense* and uninvaded deciduous forest, and thus resulted in the most accurate detection models. Singh et al. (2018) effectively demonstrated that Landsat imagery can be used to map *L. sinense* coverage with accuracy that is sufficient for monitoring and management purposes.

The method employed by Singh et al. (2018) utilizes free data (Landsat 5), making it more accessible than previous methods (i.e., Singh et al. 2015; Ward 2002). However, its reliance on the R programming language and the incorporation of vegetation and topographic indices means that it requires a level of technical skill that may still be beyond the abilities of many public and private land managers, due to a lack of relevant training. In order for a remote sensing technique to be accessible for land managers we believe that it needs to be implemented in software with a point-and-click graphical user interface (GUI) and a straight-forward, well documented workflow. Fortunately such software exists in the form of the Semi-Automatic Classification Plugin (SCP; Congedo 2019) within QGIS (QGIS Development Team 2019). This software is open source (i.e., free), has a simple to use GUI, and there are excellent support materials and tutorials available online, all of which make this a seemingly ideal option for users who have limited geographic information system (GIS) experience.

The primary objective of this study was to determine whether the SCP could be an effective tool for mapping *L. sinense* cover in a bottomland hardwood forest. Secondary objectives were to evaluate the influence of *L. sinense* cover, imagery type (S2 vs. L8), classification algorithm, and imagery acquisition date on classification accuracy.

Methods

Study Site. We conducted our study on a 2,300 ha private property located in the floodplain of the Black Warrior River in west-central Alabama, on the border between Hale and Tuscaloosa counties (Figure 2.1). The property was dominated by bottomland hardwood forests, with some interspersed loblolly pine (*Pinus taeda* L.) stands, hay fields, wildlife food plots, swamps, and oxbow lakes. Bottomland hardwood forests on the property occupied a range of geomorphic and topographic positions, with forests at various successional stages. Common forest species included cherrybark oak (*Quercus pagoda* Raf.), sweetgum (*Liquidambar styraciflua* L.), swamp chestnut oak (*Quercus michauxii* Nutt.), and bitternut hickory (*Carya cordiformis* [Wangenh.] K. Koch). Bald cypress (*Taxodium distichum* L.) and water tupelo (*Nyssa aquatic* L.) occurred in forested swamps and along the edges of oxbow lakes. The bottomland hardwood forests on the property exhibited a range of *L. sinense* cover, including uninvaded areas and dense *L. sinense* monocultures. The proportion of invaded and uninvaded hardwood forests was relatively equal.

Imagery Acquisition. We downloaded four S2 and four L8 scenes from earthexplorer.usgs.gov. For each satellite we chose two early- to mid-March images and two January images (Table 1). Only images from 2017 or later were considered to limit potential changes in *L. sinense* cover that may have occurred between the image acquisition date and our field survey. Early- to mid-March is considered late dormant season and has been identified by previous researchers as the period of maximum phenological difference between *L. sinense* and deciduous hardwoods (Singh et al. 2018). January is the middle of the dormant season and provided a useful comparison to the late dormant season March imagery. Landsat 8 scenes were downloaded as Collection 1, Level 1 products (USGS 2019) and Sentinel 2 scenes were downloaded as Level 1C products (ESA 2015). Atmospheric correction to surface reflectance was unnecessary because a separate set of training signatures were calculated for each image, precluding the need for radiance values to be standardized (Song et al. 2001). The bands for each image were clipped to our study site and a separate band stack was created for each image. Band stacks for L8 images included bands 2-7, while band stacks for S2 images included bands 2-8,8A, and 11-12, based on preset options in the SCP. Multiband stacks had a spatial resolution of 10-m for S2 and 30-m for L8.

Supervised Classification. We implemented a supervised classification approach using SCP (version 6.2.9) in QGIS (version 3.6.2). In a supervised classification the user creates a set of training areas that are representative of the land cover classes of interest. The software then calculates the spectral signatures of all pixels within those training areas. The spectral signature of a pixel is a representation of the intensity of the light being reflected within each of the bands of the electromagnetic spectrum sampled by the satellite sensor. Once the training signatures have been created, the software sorts all the pixels in the image into the appropriate land cover classes by comparing the spectral signature of each image pixel to the training signatures and choosing the best match. There are multiple algorithms available within the SCP that sort pixels based on different definitions of “best match.” We tested three of the available options: minimum distance (MD), maximum likelihood (ML), and spectral angle mapping (SA), with no minimum thresholds (Congedo 2019). We classified each of our 8 images using all 3 classification algorithms, producing a total of 24 classified maps. Classified maps are referred to in this study using the following naming convention: Satellite YYYYMMDD algorithm (e.g., S2 20170309 ML for a Sentinel 2 image acquired on March 3, 2017 classified using the maximum likelihood algorithm).

Although we were primarily interested in mapping *L. sinense* distribution, the classification algorithms require multiple land cover types in the analysis for comparison. We included the following land cover types: *L. sinense* invaded hardwoods, uninvaded hardwoods, swamp, open water, fields, and pine stands. We delineated 3 training polygons for each land cover type based on prior knowledge of the study site, visual interpretation of the satellite imagery, and (in rare cases) ground surveys (Figure 2.2). Training polygons for the *L. sinense* invaded category were primarily in areas with significant *L. sinense* cover, although we did not measure cover or set specific thresholds for the training areas. We refined the training polygons by conducting a series of informal trial-and-error classifications (primarily using ML and MD algorithms) on a subset of our L8 and S2 imagery. We adjusted the training polygons—and thus the training spectral signatures—as necessary until these initial classification attempts showed an adequate level of accuracy. This adaptive approach to creating and refining the training polygons is similar to what a land manager would use when applying this technique. Once we were satisfied with the training polygons we calculated a separate set of training signatures for each L8 and S2 scene and ran the final classification algorithms.

Accuracy Assessment. We conducted an accuracy assessment with reference data based on 250 random points surveyed during late winter/spring 2019 (Figure 2.1). For each random point we sampled 2 plots, one corresponding to the nearest L8 pixel and one corresponding to the nearest S2 pixel (two S2 plots were excluded because they fell outside the property boundary). The L8 plots were 30 m in diameter and the S2 plots were 10 m in diameter, which allowed land cover to be assessed at the pixel scale for each satellite image type. We navigated to the center point of each plot via GPS and visualized the edges of the plot using a Nikon Forestry 550 laser range finder (Nikon Vision CO., Ltd, Tokyo, Japan). We used a Garmin 64st recreational grade GPS (Garmin Ltd., Olathe, Kansas, US) for the first 74 plots, however concerns over potentially low positional accuracy led us to switch to a Trimble Geo7x GPS for the final 176 plots (Trimble Inc., Sunnyvale, California, US). We used circular plots rather than square plots for the sake of time. At each plot we recorded the land cover type and visually estimated the percent *L. sinense* cover within the plot. An informal assessment of classification accuracy differences between plots surveyed using the two GPS receivers did not reveal a significant difference.

We were specifically interested in *L. sinense* classification accuracy so we recoded the maps into a binary invaded/uninvaded scheme. We assessed how *L. sinense* cover affected classification accuracy by using a range of thresholds (1, 10, 20, 30, 40, 50, 60, 70, 80, and 90%) as the cut-offs for what would be classified as an invaded plot in the reference data. For the lowest threshold (1%) we classified a field plot as *L. sinense* invaded if it had any *L. sinense* plants, even a single individual. For higher thresholds (e.g., 40%) we only classified the plot as invaded in the reference data if it had *L. sinense* cover equal to or greater than the threshold. This helped determine how the classified maps should be interpreted (i.e., is this a map of all *L. sinense* on the property or a map of areas with greater than X% *L. sinense* cover). We calculated overall accuracy, user's accuracy, and producer's accuracy for each map at each threshold level, and the results were displayed using accuracy curves (Morisette and Khorram 2000), color coded based on image month, satellite type, and classification algorithm. Overall accuracy was calculated using the following equation:

$$\frac{TP + TN}{Total}$$

where TP = true positive (the map and the reference data agree that *L. sinense* is present), TN = true negative (the map and the reference data agree *L. sinense* is absent), and $Total$ = the total number of plots (Congalton 1991, Sanchez-Hernandez et al. 2007). Producer's accuracy was calculated as:

$$\frac{TP}{TP + FN}$$

where FN = false negative (the map predicts *L. sinense* is absent but the reference data say it is present) (Congalton 1991; Sanchez-Hernandez et al. 2007). User's accuracy was calculated as:

$$\frac{TP}{TP + FP}$$

where FP = false positive (the map says *L. sinense* is present but the reference data say it is absent) (Congalton 1991; Sanchez-Hernandez et al. 2007). Estimates of the area invaded by *L. sinense* were extracted from each map based on pixel counts and compared.

Results and Discussion

We found that the various combinations of satellite type, image date, and classification algorithm tended to highlight the same general areas on the maps as invaded, although there was some variation among all maps and a few major exceptions (Figures 2.3 and 2.4). The highest overall accuracy (92.3%) was obtained by S2 20170309 ML at a *L. sinense* cover threshold of 40% (Figure 2.5). This is on par with the overall accuracy (89.4%) achieved by the top model in Singh et al. (2018), although they did not take into account cover thresholds in their presence/absence reference data and doing so may have improved their results.

It is worth noting that the S2 20170309 image played an important role in our adaptive training site development phase, in part because it showed the greatest visual contrast between invaded and uninvaded areas in the infrared false color composite (Figure 2.2). Thus the finding that S2 20170309 ML had the highest overall accuracy could be partially due to the fact that the training polygons were somewhat tailored to that image and classification algorithm. The fact that there was a strong visual contrast in the infrared false color composite also shows that there was a high degree of spectral separation in this image, which almost certainly played a role in the high accuracy as well.

The average estimate of invaded area across all maps was 670.91 ha (± 134.99 SD), excluding 3 of the maps that failed to produce useful estimates (Figure 2.6, see discussion below). The estimate from the map with the highest overall accuracy (S2 20170309 ML) was 554.40 ha, however the differences in the optimal threshold levels interpreted from the accuracy curves of the different maps complicates comparisons of invaded areas (see below).

***Ligustrum sinense* Cover**. By analyzing the accuracy curves for all three accuracy measures on a single graph we can evaluate the best *L. sinense* cover threshold for interpreting a particular map. For example, Figure 2.5 shows that overall accuracy peaked at the 40% cover threshold and user’s and producer’s accuracy cross at 40% for S2 20170309 ML. This trade-off between user’s and producer's accuracy occurred because changing the cover threshold affected the proportion of false positives and false negatives in the accuracy assessment. At low cover thresholds there were few false positives because most of the plots where the map predicted *L. sinense* is present have at least some *L. sinense*, which is why user's accuracy is high. However there are a lot of false negatives at low cover thresholds (hence the low producer's accuracy) because at low *L. sinense* densities the spectral signature of the pixel is closer to that of an uninvaded site than that of a densely invaded site (which comprised most of the *L. sinense* invaded training polygons). As the cover threshold was increased the number of false negatives dropped (i.e., producer's accuracy went up) because the software was more effective at detecting areas with higher *L. sinense* cover. However, false positives increased (and user's accuracy went down) at higher thresholds because the map predicted some areas as invaded that did not meet the *L. sinense* cover threshold and thus were classified as "uninvaded" in the reference data. Using Figure 2.5 we can see that if we interpret S2 20170309 ML as a map of *L. sinense* presence/absence (regardless of cover level) we can only assume 66.9% overall accuracy and 34.9% producer's accuracy, but 100% user's accuracy. If we interpret the same map as a map of areas with at least 40% *L. sinense* cover, then we can assume an overall accuracy of 92.3%, 77.8% producer's accuracy, and 79.5% user's accuracy. This trade-off between the different accuracy metrics is different for each map, and we observed a wide range in the accuracy curves across our maps (Figure 2.7). For all maps there was a significant increase in overall accuracy when moving from 1% to 10% cover threshold (Figure 2.7), suggesting that this technique is not effective at detecting very low density, incipient invasions. If detecting low density invasions was the goal then creating training sites specifically tailored to those spectral signatures may help, but ultimately it may be necessary to use imagery with higher spatial and spectral resolution.

Imagery with high spatial- and spectral-resolution can improve detection of low density and/or spectrally indistinct species, however it may be less efficient at mapping high density invasions and is less practical for land managers to utilize due to high costs and technical complexities (He et al. 2011; Underwood et al. 2007).

Satellite Type. The accuracy curves in the top row of Figure 2.7 are color coded to represent the maps based on S2 and L8 imagery. The maps themselves are shown in Figures 2.3 and 2.4. We tended to see higher overall accuracy and producer's accuracy in the S2 maps, although there were a few exceptions and the relationship for user's accuracy was less clear. Both the higher spatial and spectral resolutions of the S2 imagery likely played a role in improving accuracy. Higher spatial resolution (i.e., smaller pixels) reduces the prevalence of mixed pixels, or pixels that represent more than one cover type on the ground. Mixed pixels may be more likely to be misclassified both by the mapping software and the ground surveyor. The increase in spectral resolution (i.e., more bands) increases the amount of information in each spectral signature, allowing better differentiation of similar land cover types. The finding that S2 performed better than L8 is similar to that of previous studies that used other classification algorithms to detect a variety of land cover types (Forkuor et al. 2017; Pesaresi et al. 2016; Sibanda et al. 2016; Sothe et al. 2017).

Classification Algorithm. Row 2 of Figure 2.7 contains accuracy curves coded to represent the classification algorithm used to create each map. The ML algorithm tended to perform best, with a few notable exceptions. The SA and MD algorithms did not show a clear pattern of difference.

Three out of the four ML based L8 maps failed to produce useable results (i.e., they predicted nearly complete coverage of water or fields, Figure 2.4) and were omitted from Figure 2.7. So while ML appeared to be the best option for analyzing the S2 imagery, it was not a great option for the L8 imagery. This may be because the ML algorithm requires adequate training sample sizes to calculate a covariance matrix (Congedo 2019), and since the L8 imagery has a coarser spatial resolution there are fewer pixels per training site and thus fewer training pixels in the training sample. However, past experience has shown that the SCP provides an explicit warning when the training sample is too small for the covariance matrix to be calculated and such a warning was not given during these classifications. Other researchers have also found the ML method to be robust to small training samples (Li et al. 2014). Thus the reason for the poor ML performance on L8 imagery in our study is unknown.

Moderate density pine mixed with hardwoods tended to be confused as *L. sinense* invaded hardwoods on several of the maps. The ML algorithm was less prone to making this mistake, as demonstrated in Figures 2.3, 2.4, and 2.8, although satellite type and month also appeared to play a role. Patches of native evergreen hardwoods such as American holly (*Ilex opaca* Aiton) were also confused as *L. sinense* on many of the maps, however there was not as clear of a relationship with classification algorithm as there was with the moderate density pine associated errors.

Imagery Acquisition Date. The bottom row of Figure 2.7 contains accuracy curves color coded according to the month that the imagery was collected by the satellites. Imagery collected in March tends to have higher overall and user’s accuracy, with some exceptions, and there was not a clear pattern with producer’s accuracy. These findings tend to confirm those of Singh et al.

(2018), who found that early- to mid-March imagery tended to perform the best due to greater phenological differences between the *L. sinense* and deciduous overstory. At our site we observed that the *L. sinense* had a brief leaf drop in late January and early February that was followed by a flush of fresh growth by late February and early March, during which time most of the hardwood canopy was still dormant or just beginning bud break. The exact timing of the optimal phenological differences between *L. sinense* and the overstory is dependent on local climate and annual weather patterns. Sentinel 2 imagery has a higher temporal resolution (i.e., shorter revisit time), meaning that it is more likely that cloud-free imagery will be available during the period of greatest phenological difference. The revisit time for the pair of S2 satellites is about 5 days, while L8 has a 16 day revisit time (ESA 2015, USGS 2019).

Conclusions. Our study has demonstrated that relatively dense *L. sinense* stands in deciduous hardwood forests can be effectively mapped using a free and simple remote sensing method, although the method is likely inadequate for low-density incipient invasions. We also discovered that there can be significant variation in accuracy results based on the type of satellite imagery, the date of the image acquisition, and the classification algorithm, even when the same training sites are used. Choosing appropriate training sites also has a large impact on accuracy, although that was not formally assessed in this study We tended to find that S2 imagery acquired in March and processed using the ML algorithm performed well, although these patterns may not hold true for all situations. We recommend that land managers interested in deploying this method use an adaptive process for map development that includes testing at least a few variations of training sites, images, and classification algorithms to find what works best on a particular site, using our results as a guide. A “multiple classifier system” approach that combines the results of multiple classifications could be a useful way to handle uncertainty in choosing the best map (Du et al. 2012), but further evaluation is needed to determine whether such a technique could be easily implemented in QGIS.

References

Benez-Secanho FJ, Grebner DL, Ezell AW, Grala RK (2018) Financial trade-offs associated with controlling Chinese privet (*Ligustrum sinense* Lour.) in forestlands in the southern USA. Journal of Forestry 116(3): 236-244

CABI (2018) *Ligustrum sinense* [original text by D Shaw]. In: Invasive Species Compendium. Wallingford, UK: CAB International. https://www.cabi.org/isc/datasheet/30763

Cofer MS, Walck JL, Hidayati SN. (2008) Species richness and exotic species invasion in middle Tennessee cedar glades in relation to abiotic and biotic factors. The Journal of the Torrey Botanical Society 135(4): 540-553

Congalton RG (1991) A review of assessing the accuracy of classifications of remotely sensed data. Remote Sensing of Environment 37:35-46

Congedo L (2019) Semi-automatic plugin documentation. Release 6.2.0.1, https://fromgistors. blogspot.com/p/user-manual.html

Du P, Xia J, Zhang W, Tan K, Liu Y, Liu S (2012) Multiple classifier system for remote sensing image classification: a review. Sensors 12: 4764-4792

[ESA] European Space Agency (2015) Sentinel-2 user handbook. Paris, France: European Space Agency. 64 p

Foard M (2014) Causes and Consequences of Chinese privet (*Ligustrum sinense* Lour.) invasion in hydrologically altered forested wetlands. Master's thesis. Jonesboro, AR: Arkansas State University. 172 p

Forkuor G, Dimobe K, Serme I, Tondoh JE (2017) Landsat-8 vs. Sentinel 2: examining the added value of Sentinel-2's red edge bands to land-use and land-cover mapping in Burkina Faso. GIScience and Remote Sensing 55(3): 331-354

Gómez C, White JC, Wulder MA (2016) Optical remotely sensed time series data for land cover classification: a review. ISPRS Journal of Photogrammetry and Remote Sensing 116: 55-72

Green BT, Blossey B (2012) Lost in the weeds: *Ligustrum sinense* reduces native plant growth and survival. Biological Invasions 14:139-150

Grove E, Clarkson BD (2005) An ecological study of Chinese privet (*Ligustrum sinense* Lour.) in the Waikato region. Hamilton, New Zealand: Centre for Biodiversity and Ecology Research Contract Report No. 41. 17 p

Hart JL, Holmes BN (2013) Relationships between *Ligustrum sinense* invasion, biodiversity, and development in a mixed bottomland forest. Invasive Plant Science and Management 6(1): 175-186

He KS, Rocchini D, Neteler M, Nagendra H (2011) Benefits of hyperspectral remote sensing for tracing plant invasions. Diversity and Distributions 17: 381-392

Kelly M, Tommaso SD (2015) Mapping forests with lidar provides flexible, accurate data with many uses. California Agriculture 69(1): 14-20

Klepac J, Rummer RB, Hanula JL, Horn S (2007) Mechanical removal of Chinese privet. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station SRS-43

Knipling EB (1970) Physical and physiological basis for the reflectance of visible and near-infrared radiation from vegetation. Remote Sensing of Environment 1:155-159

Li C, Wang J, Wang L, Hu L, Gong P (2014) Comparison of classification algorithms and training sample sizes in urban land classification with Landsat Thematic Mapper imagery. Remote Sensing 6:964-983

Lowenstein NJ, Lowenstein EF (2005) Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. Urban Ecosystems 8:79-91

Maddox V, Byrd J Jr., Serviss B (2010) Identification and control of invasive privets (*Ligustrum* spp.) in the middle southern United States. Invasive Plant Science and Management 3(4): 482-488

Merriam RW, Feil E (2002) The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions 4: 369-373

Miller JH, Chambliss EB (2008) Estimates of acres covered of nonnative invasive plants in southern forests. Auburn, AL: US Department of Agriculture, Forest Service, Southern Research Station. https://www.invasive.org/fiamaps/summary.pdf

Miller JH, Chambliss EB, Bargeron CT (2004) Invasive plants of the thirteen southern states. https://www.invasive.org/south/seweeds.cfm?sort=3

Miller JH, Miller KA (2005) Forest plants of the Southeast and their wildlife uses. revised ed. Athens, GA: University of Georgia Press. p 360

Morisette JT, Khorram S (2000) Accuracy assessment curves for satellite-based change detection. Photogrammetric Engineering and Remote Sensing 66(7): 875-880

[NASA] National Aeronautics and Space Administration (1999) Land cover classification: how scientists differentiate between land cover types. https://earthobservatory.nasa.gov /features/LandCover/land_cover_2.php

Pesaresi M, Corbane C, Julea A, Florczyk AJ, Syrris V, Soille P (2016) Assessment of the added-value of Sentinel-2 for detecting built-up areas. Remote Sensing 8, 299 doi: 10.3390/rs8040299

Phiri D, Morgenroth J (2017) Development in Landsat land cover classification methods: a review. Remote Sensing 9: 967-992

Pokswinki SM (2009) Invasive characteristics of Chinese privet (*Ligustrum sinense* Lour.) in a bay swamp in the fall line hills of east-central Alabama. Master's thesis. Auburn, AL: Auburn University. 69 p

QGIS Development Team (2019) QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org/

Resasco J, Hale AN, Henry MC, Gorchov DL (2007) Detecting an invasive shrub in a deciduoud forest understory using late-fall Landsat sensor imagery. International Journal of Remote Sensing 28(16): 3739-3745

Sanchez-Hernadez C, Boyd DS, Foody GM (2007) One-class classification for mapping a specific land-cover class: SVDD classification of fenland. IEEE Transactions on Geoscience and Remote Sensing 45(4): 1061-1073

Sibanda M, Mutanga O, Rouget M (2016) Discriminating rangeland management practices using simulated HyspIRI, Landsat 8 OLI, Sentinel 2 MSI, and VENµS spectral data. IEEE Journal of Selected Topics in Applied Earth Observation and Remote Sensing 9(9): 3957-3969

Singh KK, Chen Y, Smart L, Gray J, Meentemeyer RK (2018) Intra-annual phenology for detecting understory plant invasion in urban forests. ISPRS Journal of Photogrammetry and Remote Sensing 142: 151-161

Singh KK, Davis AJ, Meentemeyer RK (2015) Detecting understory plant invasion in urban forests using LiDAR. International Journal of Applied Earth Observation and Geoinformation 38: 267-279

Song C, Woodcock CE, Seto KC, Lenney MP, Macomber SA (2001) Classification and change detection using Landsat TM data: when and how to correct atmospheric effects. Remote Sensing of Environment 75:230-244

Sothe C, Almeida CMD, Liesenberg V, Schimalski MB (2017) Evaluating Sentinel-2 and Landsat-8 data to map successional forest stages in a subtropical forest in southern Brazil. Remote Sensing 9, 838; doi:10.3390/rs9080838

Underwood EC, Ustin SL, Ramirez CM (2007) A comparison of spatial and spectral image resolution for mapping invasive plants in coastal California. Environmental Management 39(1): 63-83

[USGS] US Geological Survey (2019) Landsat 8 (L8) data user’s handbook. 4th ed. Sioux Falls, SD: US Geological Survey, Earth Resources Observation and Science Center, LSDS-1574. 115 p

Ward RW (2002) Extent and dispersal rates of Chinese privet (*Ligustrum sinense*) invasion in the upper Oconee River floodplain, north Georgia. Southeastern Geographer 42(1):29-48

Wilcox J, Beck CW (2007) Effects of *Ligustrum sinense* Lour. (Chinese privet) on abundance and diversity of songbirds and native plants in a southeastern nature preserve. Southeastern Naturalist 6(3): 535-550

Xie Y, Sha Z, Yu M (2008) Remote sensing imagery in vegetation mapping: a review. Journal of Plant Ecology. 1(1): 9-23

Table 2.1. Sentinel 2 and Landsat 8 Imagery used in this study. Dates follow the YYYY/MM/DD format.

Landsat 8	Sentinel 2
2017/03/09	2017/01/28
2018/03/12	2017/03/09
2019/01/10	2018/03/14
2019/01/26	2019/01/28

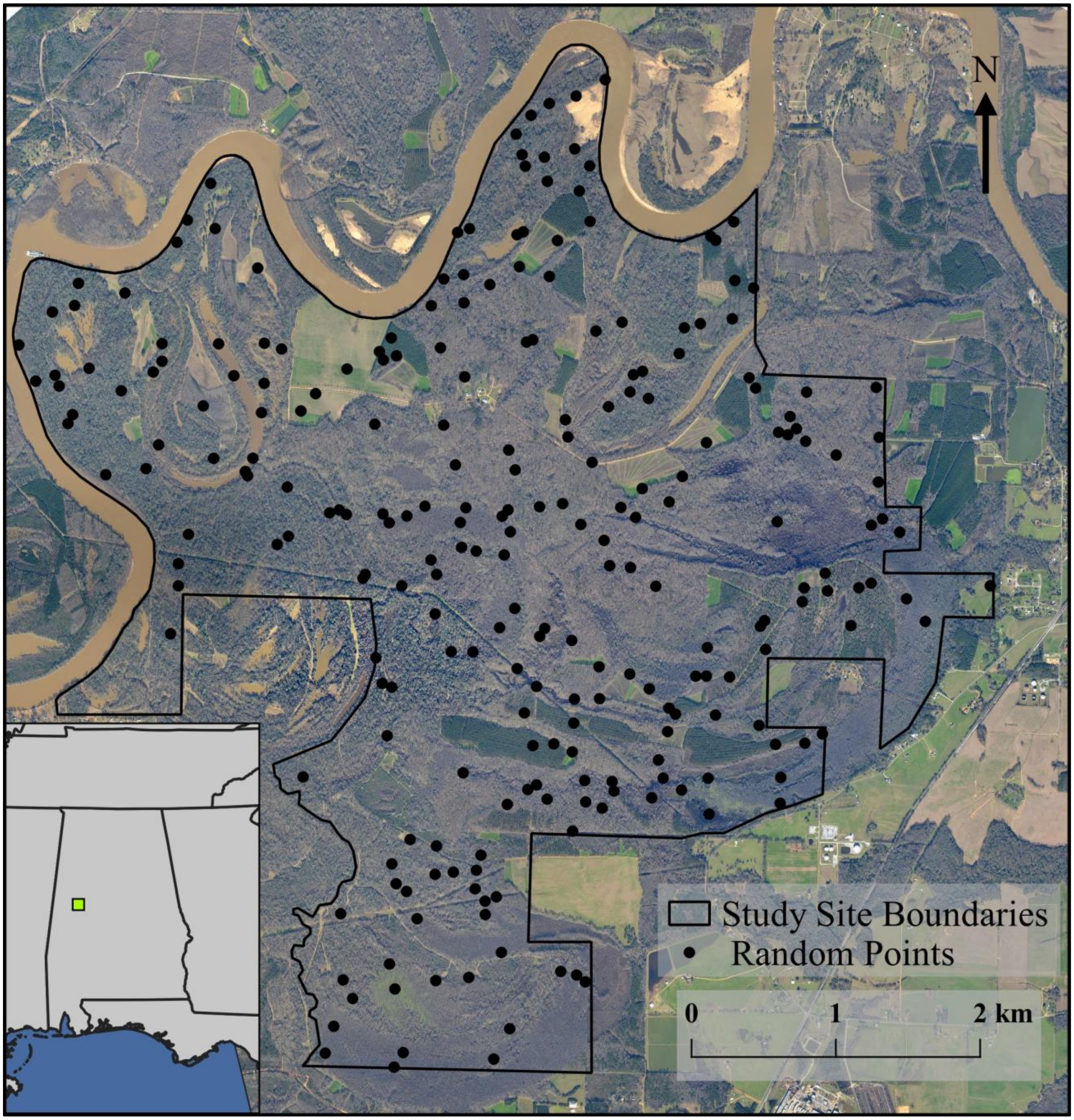


Figure 2.1. Aerial photograph of the study site where we evaluated satellite imagery-based detection of *Ligustrum sinense*. The random points are where we assessed classification accuracy. The yellow square on the inset map shows the study site location relative to the State of Alabama (not to scale).

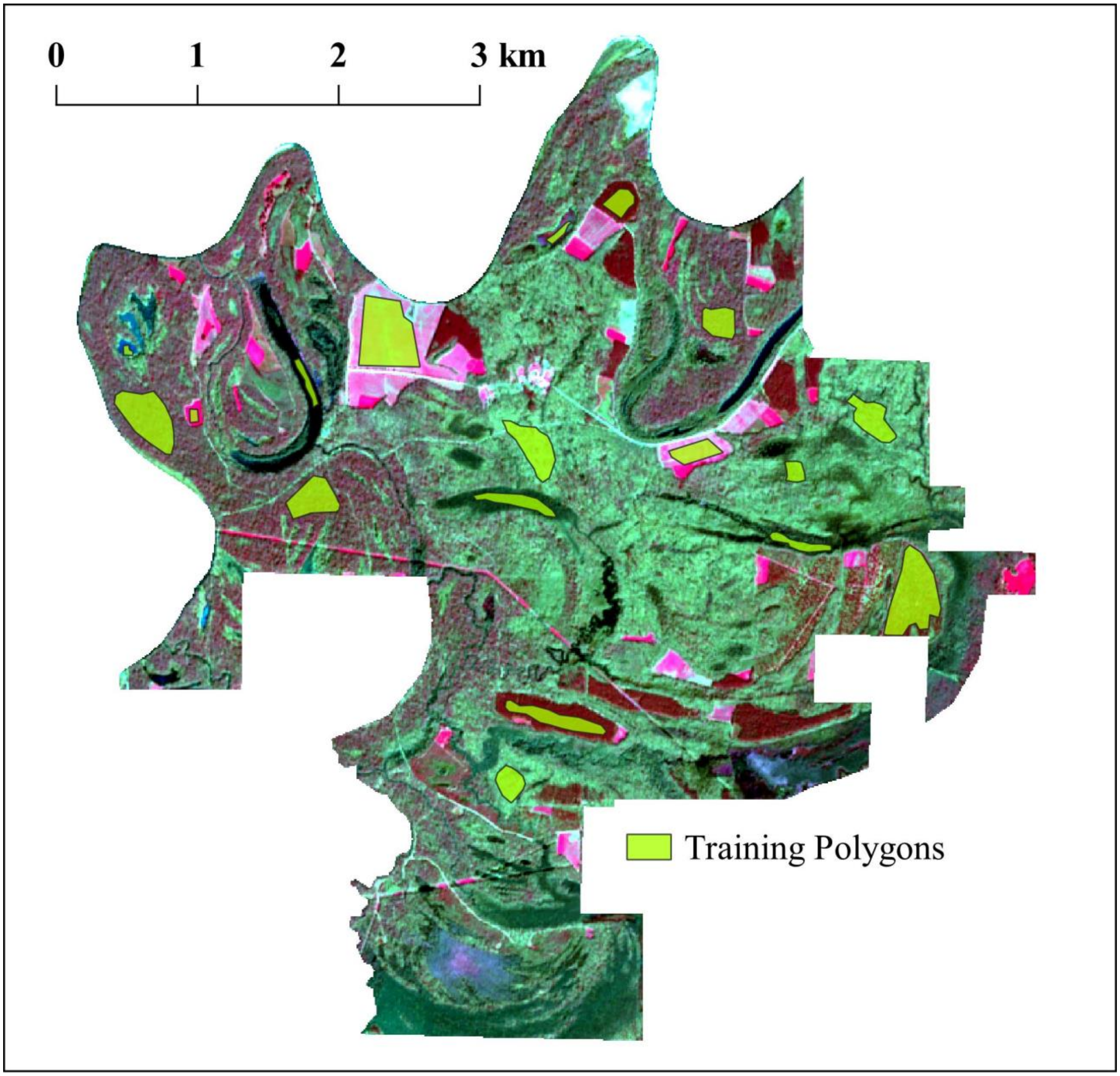


Figure 2.2. Infrared false color composite Sentinel 2 image (2017/03/09) of the study site where we evaluated satellite-imagery based detection of *Ligustrum sinense*. Training polygons for the supervised classification are shown in yellow.

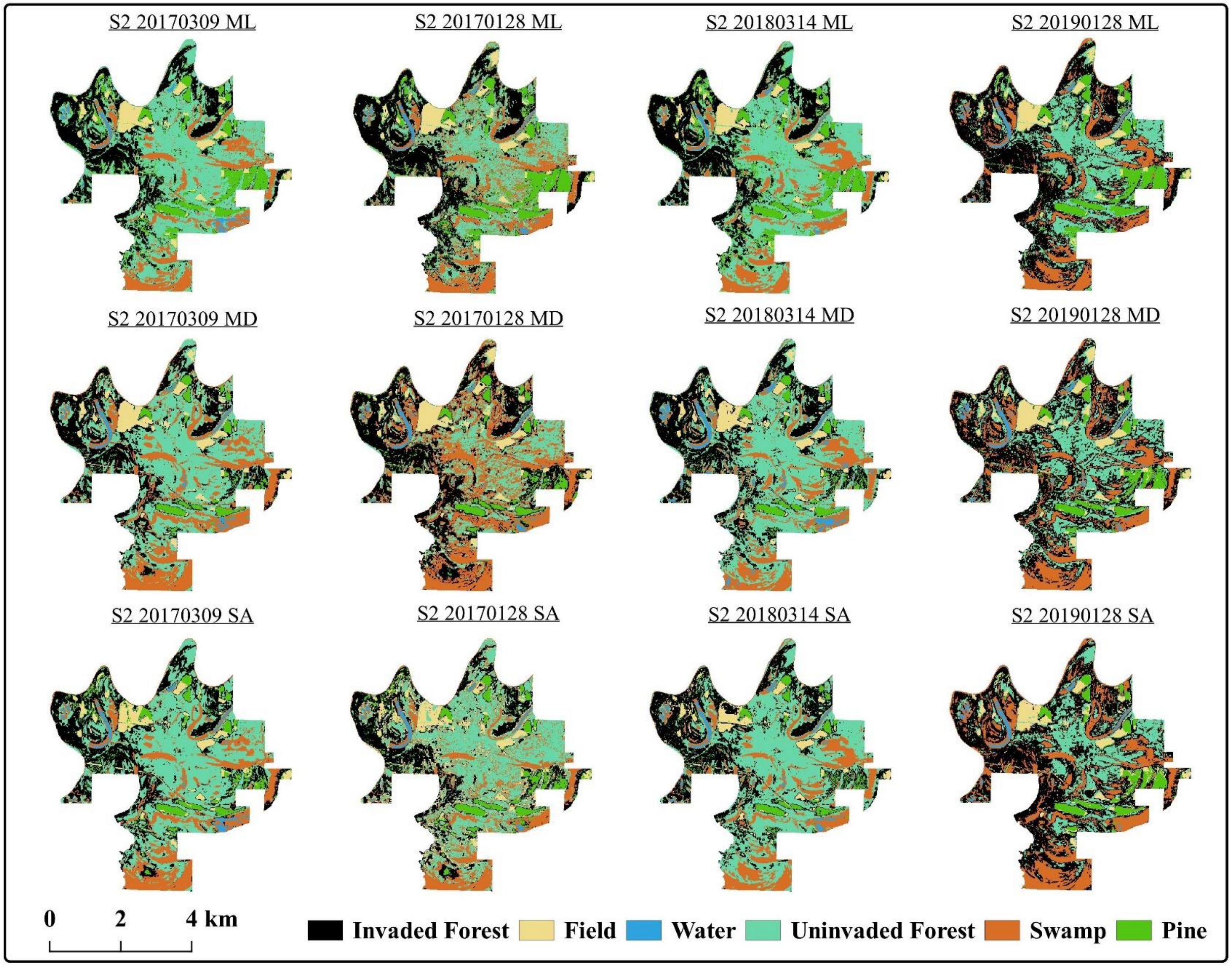


Figure 2.3. Classified Sentinel 2 maps. Columns share the same date, rows share the same classification method.

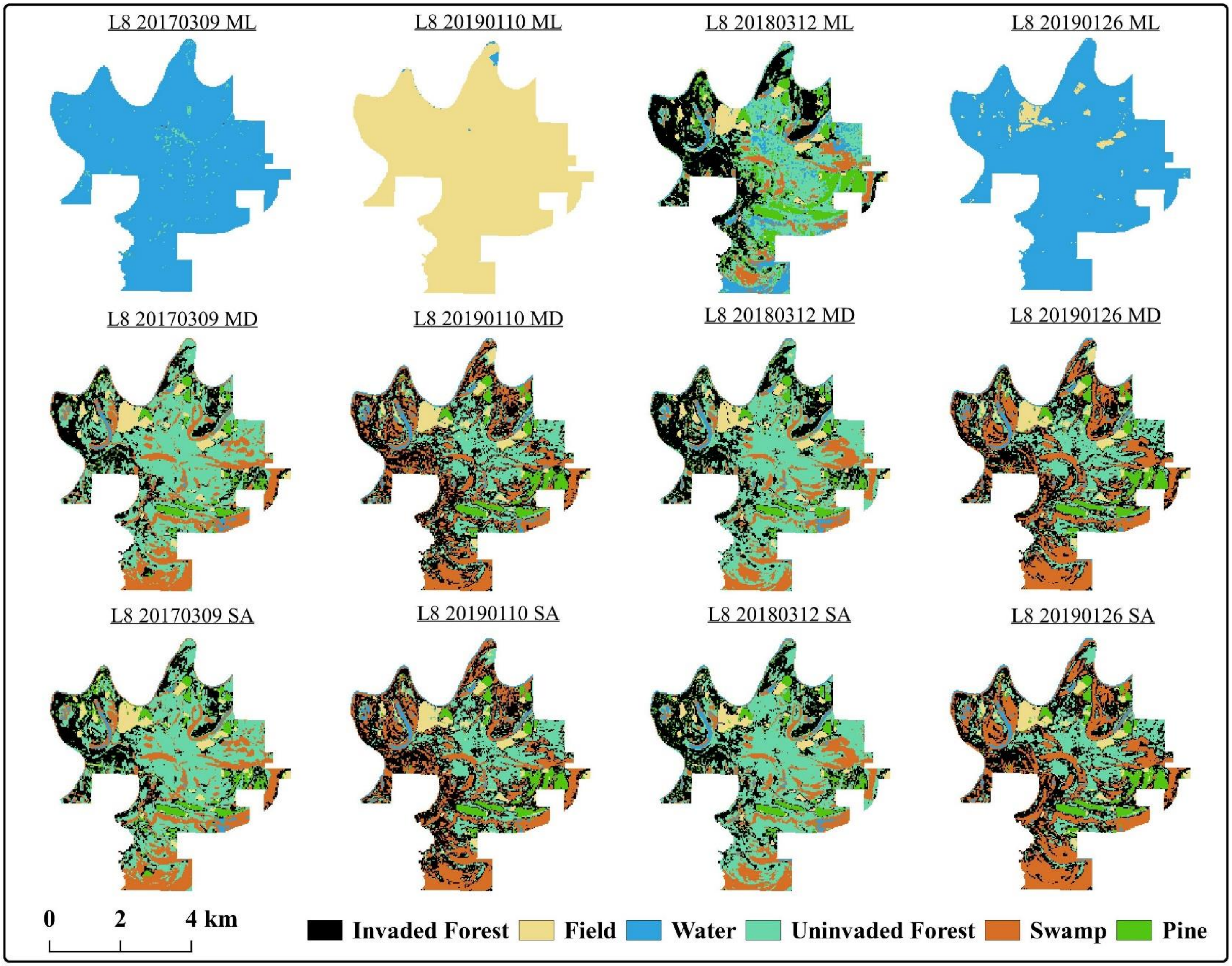


Figure 2.4. Classified Landsat 8 maps. Columns share the same date, rows share the same classification method.

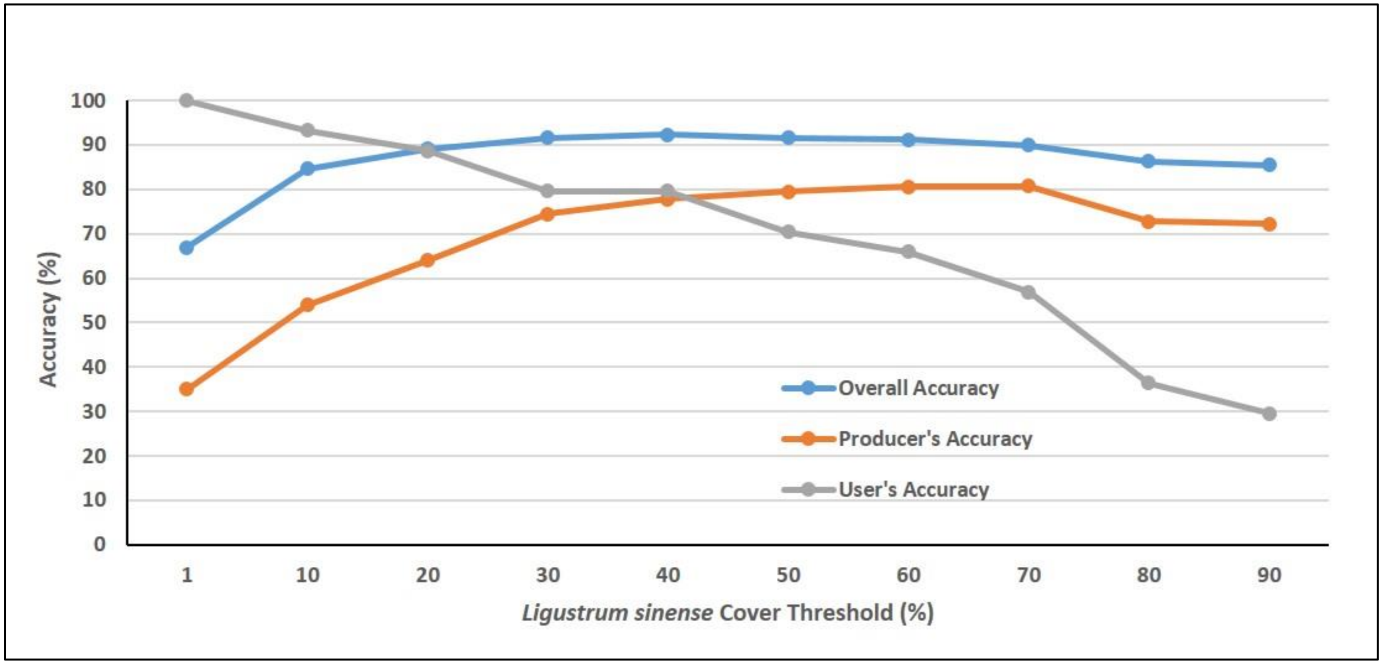


Figure 2.5. Accuracy statistics for maximum likelihood classification of Sentinel 2 imagery acquired on 2017/03/09 (S2 20170309 ML). This graph demonstrates that the S2 20170309 ML map is best interpreted as a map of areas with ≥40% *Ligustrum sinense* cover.

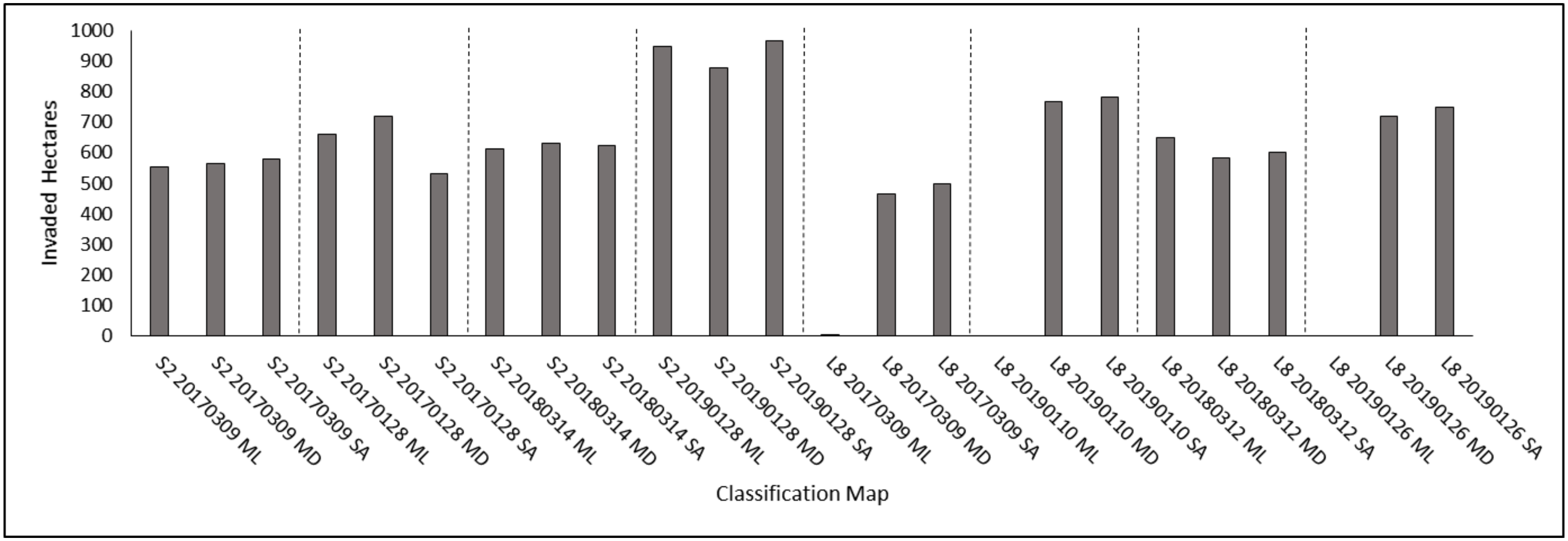


Figure 2.6. Comparison of estimates of hectares invaded by *Ligustrum sinense* among classification maps. Image dates are separated by dotted lines.

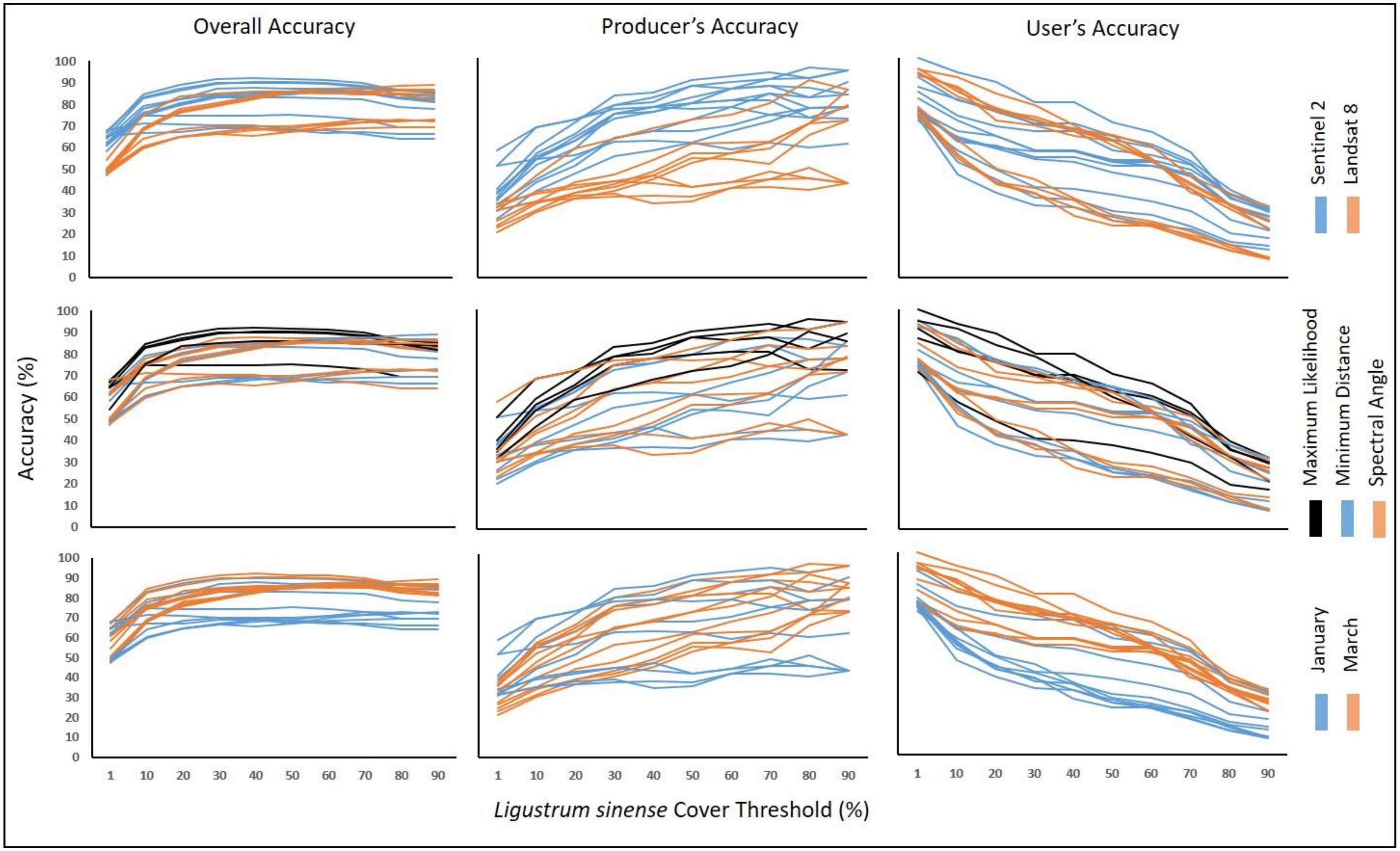


Figure 2.7. Classification accuracy curves (overall, producer's, and user's). Columns share accuracy type, rows share color scheme. The color schemes demonstrate how satellite type, image date, and classification algorithm affect the three accuracy measures. Accuracy curves were omitted for the three Landsat 8 maximum likelihood-based maps that failed.

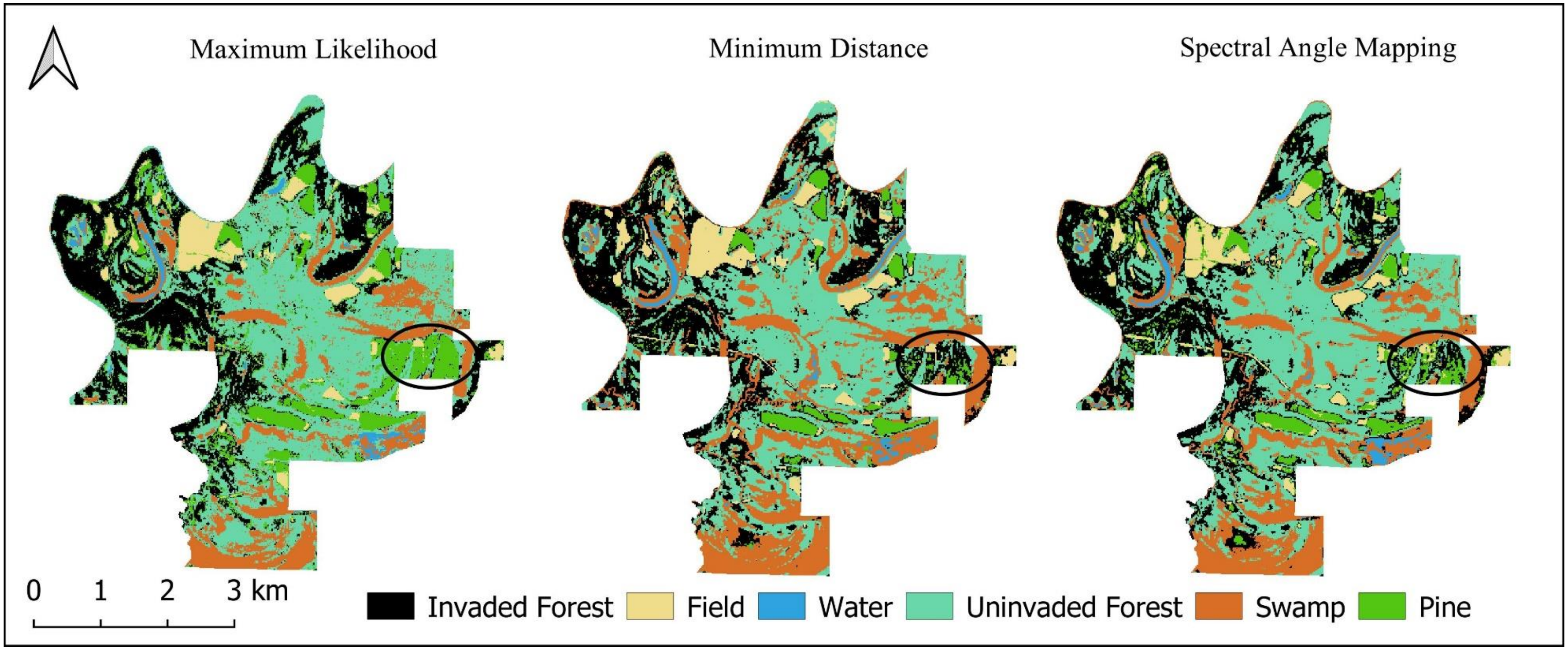


Figure 2.8. Comparison of different classification algorithms applied to a Sentinel 2 image collected on 2017/03/09. The circle on the east side of the maps shows an area of moderate density pine forest that was partially confused as *Ligustrum sinense* invaded forest by the minimum distance and spectral angle algorithms, but not the maximum likelihood algorithm.

Chapter 3

Applying Prescribed Fire in Bottomland Hardwood Forests and Evaluating its Capacity to Reduce *Ligustrum sinense* Slash Following Cutting Operations

Abstract

Controlling Chinese privet (*Ligustrum sinense* Lour.), an invasive shrub, often involves a combination of mechanical and herbicide treatments. However, cutting *L. sinense* down creates heavy slash which can reduce the efficiency of follow-up herbicide treatments on re-sprouts. We evaluated whether prescribed fire could be used in bottomland hardwood forests to reduce slash build-up following cutting operations. We conducted twenty-two small scale prescribed fires to assess how in-stand weather (i.e., microclimate), stand composition, and litter measurements affected fire behavior. We found that stand composition had the most important effect on fire spread, with plots that had higher proportions of tree species with more flammable leaf traits tending to burn best. We tracked *L. sinense* slash consumption and found that only the smallest diameter slash (0 – 0.64cm) was significantly reduced by the fires. Our findings show that the ability to use prescribed fire in bottomland systems is limited to certain sites, and even within those sites it may not be a useful tool for reducing slash and increasing the effectiveness of *L. sinense* control operations. However, more research is needed to determine how prescribed fire may play a role in *L. sinense* management and bottomland hardwood management in general.

Introduction

Chinese privet (*Ligustrum sinense* Lour.), an invasive semi-evergreen shrub, is one of the most widespread and ecologically damaging non-native plants in the southeastern United States. *Ligustrum sinense* can form dense stands that reduce native plant cover and biodiversity, and there is particular concern that it may threaten long-term forest sustainability by limiting overstory tree recruitment (Green and Blossey 2012; Hart and Holmes 2013; Lowenstein and Lowenstein 2005; Merriam and Feil 2002; Wilcox and Beck 2007). *Ligustrum sinense* has broad ecological tolerances and can be found in both upland and bottomland sites (Cofer et al. 2008; Grove and Clarkson 2005; Miller and Miller 2005; Pokswinki 2009).

Common *L. sinense* control methods include foliar herbicide application via aerial, skidder, or backpack applicators, basal bark herbicide treatment, or mechanical removal via chainsaws or mulchers, followed by cut-stump herbicide application and/or re-sprout foliar herbicide application (Benez-Secanho et al. 2018; Hanula et al. 2009; Klepac et al. 2007; Rhodes and Israel 2014). Each of these methods has costs and benefits. Foliar spraying of mature plants via broadcast aerial or skidder application is generally the most cost effective method for large areas (Benez-Secanho et al. 2018). However, foliar application will leave standing dead *L. sinense* (as will basal bark treatment), which may be aesthetically undesirable for some landowners. Additionally, concerns over non-target impacts from herbicide drift over wetlands and waterways (Krynak et al. 2017) may complicate aerial applications, and skidder maneuverability may be limited in dense stands (Benez-Secanho et al. 2018). Mechanical removal via mulchers or chainsaws eliminates the standing dead problem and allows more targeted herbicide application to stumps and/or re-sprouts. Mulching machine costs are generally higher than chainsaw costs, possibly making chainsaws a more appealing option (Benez-Secanho et al. 2018). Heavy machinery may also have negative soil impacts when operated in moist bottomland sites (Cambi et al. 2015), potentially limiting both mulcher and skidder sprayer operations. However, cutting dense *L. sinense* stands with chainsaws results in considerable slash build-up (Figure 3.1). This slash may be undesirable from an aesthetic standpoint and inhibits subsequent stand management actions (e.g., herbicide applications on *L. sinense* seedlings and re-sprouts), possibly increasing costs due to reduced efficiency (Hanula et al. 2009).

Prescribed fire may be a useful tool for reducing *L. sinense* slash accumulation following cutting, thus increasing the efficiency of follow-up management actions. Ideally the slash could be significantly reduced with a single burn, which would allow timely access for subsequent individual plant treatments (IPT), such as herbicide foliar spraying of re-sprouts. Conventional wisdom holds that prescribed fire should not be used in bottomland hardwood forests for fear of damaging overstory trees (Kaufert 1933). However, the presence of fire adapted species such as giant cane (*Arundinaria gigantea* [Walter] Muhl.) in bottomland hardwood forests suggest that occasional fires are natural in these systems (Gagnon 2009). While bole damage from fires may reduce merchantable timber values (Kaufert 1933), the resulting cavities can provide valuable wildlife habitat (Gooding and Langford 2004; Hellgren and Vaughan 1989), and many bottomland hardwood forests are managed for objectives other than timber production.

Burning in mesic bottomland systems is poorly understood and likely challenging due to the predominantly moist conditions resulting from periodic flooding, high proportions of mesophytic tree species, and damp microclimates (Nowacki and Abrams 2008). Common mesophytic trees include species such as sweetgum (*Liquidambar styraciflua* L.), winged elm (*Ulmus alata* Michx.), and red maple (*Acer rubrum* L.) (Kreye et al. 2013; Kreye et al. 2018). These species have leaves that are thin and flat, which limits air flow in the fuel bed, slows drying times, and reduces flammability (Kreye et al. 2013; Nowacki and Abrams 2008). However, some tree species found on mesic sites, such as cherrybark oak (*Quercus pagoda* Raf..), have leaf morphologies that are similar to upland species (i.e., southern red oak [*Q. falcate* Michx.]) that are known to be more conducive to fire spread (Kreye et al. 2018). These relatively fire-prone species tend to have more rigid leaf structures and some will curl while drying, which tends to promote further moisture loss and—in combination with other related traits—improves flammability (Kreye et al. 2013; Nowacki and Abrams 2008; Varner et al. 2015). Fire-prone species also tend to have greater leaf mass and the leaves decompose slower (Nowacki and Abrams 2008), thus creating higher leaf litter loads.

There were two primary objectives of this study. The first objective was to determine how stand characteristics and in-stand weather (i.e., microclimate) variables influenced our ability to use prescribed fire in bottomland areas. We hypothesized that sites with more tree species with fire-promoting leaf traits would burn better due in part to greater fuel loads and lower litter moisture. Areas with higher *L. sinense* slash loads would also likely burn better because the slash would provide a source of dry fuel, and leaf litter draped among the slash would also dry faster. We anticipated that wind speed and temperature would be positively correlated with fire spread, whereas relative humidity (RH) would be negatively correlated with fire spread (Schroeder and Buck 1970; Waldrop and Goodrick 2012). Additionally, we hoped to identify thresholds for these weather variables conducive to fire success. Our second objective was to determine how well fire reduced *L. sinense* slash accumulation. We hypothesized that fire would reduce all slash size classes, but smaller diameter fuels would be more readily consumed than larger diameter fuels, due in part to the slower drying times of larger fuels (Schroeder and Buck 1970; Ward 2013).

Methods

Study Site. We conducted our study on a 2300 ha site located in the floodplain of the Black Warrior River, on the border between Hale and Tuscaloosa counties in west-central Alabama (Figure 3.2). The site is dominated by primary and secondary bottomland hardwood forest. Across our experimental plots, *Q. pagoda* was the dominant species followed by *L. styraciflua*. We observed a total of 26 species and there was a high degree of variation in stand composition among our plots (Table 3.1, Appendix 2). The climate is warm temperate humid (Kottek et al. 2006), and the dominant soil types are Urbo-Mooreville-Una complex and Ellisville silt loam (NRCS 2018). We established 32 experimental plots distributed among 3 sites on the property, each plot roughly 20x20 m (0.04 ha) in size (Figure 3.2).

Stand Composition Measurements. We identified and measured the diameter at breast height (DBH) of every tree >8cm DBH in each plot. We classified fire-prone tree species based on Kreye et al. (2013, 2018; Table 3.1). We did not observe the same “pyrophytic” upland species used in the studies by these authors (with one exception), but we classified species as “fire-prone” if they were in the same genus and had similar fire-promoting leaf traits as the “pyrophytic” species in Kreye et al. (2013, 2018). In labeling those species as fire-prone we are only referring to leaf traits and their potential effect on fire behavior, not the ecological or physiological preferences or tolerances of the species (as might be the case with a “pyrophytic” designation). We then calculated the proportion of the total tree basal area of each plot that was composed of fire-prone species (i.e., the relative basal area [RBA] of fire-prone species), which was used as an index for the proportion of the leaf litter that was likely made up of fire-prone species.

Slash Measurements. Before cutting operations commenced we sampled the basal diameter of standing *L. sinense* within each plot to aid in the estimation of post-cut slash. We established two 2-m wide diagonal belt transects in each plot (from each corner to the opposite corner), and measured the basal diameter of all *L. sinense* stems over 1.4 m tall within each transect. We estimated the above ground *L. sinense* biomass, and thus the slash created by the cutting operations, in each plot based on the following allometric equation provided by Oswald et al. (2017):

$$Above\ Ground\ Biomass\ (g) = 54.17502(basal\ diameter\ [cm]^{2.66992}) * 1.06487$$

Plots in sites A and B (Figure 3.2) were cut with chainsaws and machetes during August and October 2017. Site C plots were cut in August 2018.

Following cutting operations, we established permanent slash transects to monitor fire-related slash reduction using a modified version of the technique described in Brown (1974). We randomly chose one corner of each plot and ran a baseline transect to the opposite corner. Five 2-m long permanent slash transects were systematically anchored on this baseline at 5 m intervals and radiated out at random azimuths. We tallied the number of dead *L. sinense* branches that intersected each slash transect, organized into standard fuel diameter classes based on moisture time lags (1 hour [0 - 0.64 cm], 10 hour [0.64 – 2.54 cm], and 100 hour [2.54 – 7.62 cm]; Brown 1974). We were careful not to disturb the arrangement of slash and leaf litter while conducting the surveys, as doing so may have affected fire behavior and slash consumption. These transects were measured before and after the prescribed fire trials. During the post-treatment survey we noted whether the transects had visible signs of fire scorching. Traditionally, planar slash counts are converted to an estimate of weight per unit area (e.g., kg ha^{-1}). However, doing so requires knowing several species-specific traits of the slash: average diameter per size class, specific gravity per size class, and a nonhorizontal angle correction factor (Brown 1974). We could not identify such constants in the literature for *L. sinense* or a suitable proxy species, so we conducted our pre-/post-burn analyses on the raw slash counts.

Litter Measurements. Before each fire we collected four litter samples using 30.5x30.5 cm quadrats. These quadrats were subjectively placed in areas with little to no slash build-up (generally one per plot corner) to facilitate standardized litter collection (litter collection from within slash piles was too difficult to standardize). The litter samples were dried in a forced air oven at 65° C for at least 1 week. Average litter percent moisture and dry weight were calculated from the samples (Pollet and Brown 2007).

Prescribed Fires. We conducted prescribed fires during January, March, April, and May of 2018 (Site A and B) and March of 2019 (Site C). We burned plots in a subjective order based on proximity (e.g., burning multiple plots that were close together on the same day), and to maximize fire success (e.g., burning plots on days when they seemed most likely to burn well). This study was part of a larger study that evaluated other aspects of fire effects and we had limited site accessibility, so maximizing efficiency and effectiveness took precedent over randomizing treatment order. We were only able to collect data on 22 prescribed burns (20 with slash data) due to logistical difficulties and site accessibility issues, including historic flooding in January 2019. This flooding affected many of our plots by depositing silt on the leaf litter and/or carrying away leaf litter, which suppressed fire activity and is a factor to consider when attempting to burn in bottomland systems. We did not attempt to burn in plots that had been severally impacted by flooding.

At the start of each prescribed fire we recorded in-stand temperature, relative humidity, and wind speed using a Kestrel 3000 wind meter (Kestrel Meters, Boothwyn, PA, USA). Each fire began with a backing fire, head fire, or ring fire (determined based on weather and plot conditions). We let this initial ignition burn until it went out (self-extinguished). We then visually estimated the percent of the plot that burned. We sometimes re-ignited a plot using a head fire, strip head fire, ring fire, or interior ignition, however these re-ignition attempts were not standardized and not analyzed as part of this study. They did however affect the number of slash transects that were scorched.

Statistical Analyses. To address our first objective, how plot and weather variables affected fire spread, we used the percent of the plot that burned following the first ignition as our dependent variable (“burn percent”). Further plot ignitions were less standardized and could have been affected by the burn pattern of the initial ignition, so for the purposes of statistical analysis we only focused on the first ignition. All analyses were conducted in base R version 3.5.1 (R Core Team 2018) unless otherwise specified. We used ANOVA to determine whether ignition pattern (backing, head fire, ring fire) affected the burn percentage. Due to our small sample size (n = 22) we could not effectively model all 7 continuous variables simultaneously in a multiple regression. We assessed the pairwise Pearson’s correlations and asymptotic p-values for all continuous variables using R package “Hmisc” (Harrell et al. 2019). Variable correlations that were related to our *a priori* hypotheses and were significant ($\alpha = 0.05$), or considered important for interpretation of other relationships, were further analyzed using simple linear regression. Scatterplots and histograms of residuals were examined for gross violations of linear model assumptions.

We used an additional, more qualitative, approach to examine potential useful thresholds for future fire prescriptions. We sorted the samples according to burn percentage along apparent breaks in the data that corresponded to varying levels of fire success: failure (≤ 10% plot burned), marginal success (20-60% plot burned), and successful (≥70% plot burned). For each of these groups we reported the mean and range of values for each of the plot and weather variables.

We used linear mixed effects models to compare pre- vs. post-treatment slash counts for each of the three fuel size classes (1-hr, 10-hr, 100-hr). A nested design was used with random effects for transects nested within random effects for plots to account for non-independent samples (Harrison et al. 2018). We reported the mean and standard errors for each size class in the pre-treatment surveys, along with the differences between the pre- and post-treatment surveys. We conducted additional mixed-effect linear regressions in which un-scorched transects were treated as controls and an interaction term between survey (pre and post) and treatment (scorched and no scorch) was used to separate the effects of the fire from natural decomposition, in a variation of a before-after-control-impact design. Mixed effects models were implemented in R package “lme4” (Bates et al. 2015) with the extension package “lmerTest” (Kuznetsova et al. 2017).

Results and Discussion

Variable Relationships with Plot Burn Percentage. We found no significant effect of ignition pattern on burn percentage (p = 0.753). Qualitatively speaking, we tended to find that head fires or strip-head fires were required to minimize the time necessary to burn plots. Head and strip-head fires are pushed by the wind and pre-heat fuels in front of the flaming front, which increases rate of spread (Wade 2013*a*, Waldrop and Goodrick 2012). Backing fires tend to have shorter flame lengths (Wade 2013*a*), which could reduce the ability of the fire to burn fuels suspended in the top layers of the slash piles. Prescribed burn managers must also take into account weather conditions, stand characteristics, burn unit layout, and landscape context when deciding on an ignition pattern. We recommend consulting Waldrop and Goodrick (2012) for further guidance.

Correlations among our continuous variables are shown in Table 3.2. Contrary to our expectations, none of the weather variables were significantly correlated with burn percentage. The relationship between fire behavior and weather variables such as in-stand wind, temperature, and RH are well known among firefighters and prescribed burners (Waldrop and Goodrick 2012). Decreasing RH and increasing temperature can lower fuel moisture, which reduces the heat input required to ignite fuels and increases rate of spread (Wade 2013*b*; Waldrop and Goodrick 2012). However, the fire does not respond immediately to these changes because the fuel takes time to lose moisture (Wade 2013*b*, Waldrop and Goodrick 2012). Assessing the effect of weather variables at the moment of ignition likely missed this lag effect. Additionally, the wind was often light and variable in our stands, which prevented the fire from making significant forward progress, but our wind measurements did not adequately capture this variability. Wind speeds under dense forest canopies will be lower than in unprotected locations (Andrews 2012, Waldrop and Goodrick 2012), and the dense *L. sinense* stands that surrounded our burn plots likely further reduced surface-level wind speeds. Another potential reason for the lack of significant correlations is that we were selective about the days when we attempted burning, generally only doing so if RH was around or below 40% (with some exceptions). If we had attempted burning on days with a wider range of weather conditions and if we had measured the weather variables at a more appropriate temporal scale we may have seen stronger correlations.

Our qualitative assessment of thresholds (Table 3.3) and our informal observations in the field suggest that an RH <30% with a consistent wind is preferable, but more data are needed to confirm this. We should note that burning with an RH <30% is considered dangerous on most sites due to high fire intensity and increased rate of spread (Waldrop and Goodrick 2012); however, such conditions may be necessary to burn in moist bottomland sites. Nevertheless, caution should be taken to ensure the fire does not spread into more receptive upland fuels. It is clear that under the range of conditions in this study, stand composition and litter variables were more important to fire success than weather variables.

The correlation table (Table 3.2) confirmed a significant relationship between burn percentage and relative basal area of fire-prone species (r = 0.63, p = 0.002; Figure 3.3). Litter loading was correlated with both fire-prone species RBA (r = 0.47 p = 0.028; Figure 3.4) and burn percent (r = 0.46, p = 0.033; Figure 3.4). Litter moisture was also correlated with fire-prone species RBA (r = -0.54, p = 0.009; Figure 3.4), but was not significantly related to burn percent (r = -0.36, p = 0.102; Figure 3.4). The finding that the fire-prone species RBA was positively correlated with burn percent supports previous experimental studies (Kane et al. 2008; Kreye et al. 2013; Kreye et al. 2018). However, Kreye et al. (2013, 2018) reported increased drying of leaf litter (i.e., lower litter moisture) as an important component in how “pyrophytic” species affect fire, and while we observed a significant negative correlation between litter moisture and fire-prone species RBA, we were surprised that the relationship between litter moisture and burn percent was not significant. This may have been an artifact of small sample size and the fact that we sampled the full depth of the leaf litter layer, whereas perhaps only the top layer of relatively intact and drier leaves are important for fire spread. That being said, Kreye et al. (2018) reported that fuel beds with high proportions "mesophytic" species are less flammable than "pyrophytic" litter beds even when fuel moisture is very low, which demonstrates that differences in leaf properties among these two groups affect fire behavior through mechanisms beyond moisture retention. Litter loading was significantly related to both fire-prone species RBA and burn percent in our study, suggesting that the effect of fire-prone species on the burn percent was influenced more by litter loading than litter moisture. Other leaf and fuel bed characteristics we did not measure, such as packing density, surface area to volume ratio, and chemical composition likely affected these patterns as well (Kane et al. 2008, Nowacki and Abrams 2008). Differences in forest structure among sites with varying proportions of fire-prone species may also effect fire behavior due to differences in solar radiation, RH, and in-stand wind (Kreye et al. 2018; Nauertz et al. 2004; Nowacki and Abrams 2008). Future research could include more precise measurements of leaf litter composition within plots, as our method based on relative basal area may have been affected by leaves entering the plot from trees rooted outside the plot, as well as different relationships between basal area and leaf production among tree species. Greater sample sizes could also allow future researchers to test for the effect of individual species on fire behavior, as opposed to the approach we used that lumped all species with potentially flammable leaf traits together (some of which are likely more flammable than others).

We expected plots with higher *L. sinense* slash loading to burn better due to greater fuel availability and increased litter drying due to litter draping in the slash, however we found no significant relationship between *L. sinense* slash loading and burn percent (r = 0.35, p = 0.116). The significant negative relationship between litter moisture and *L. sinense* slash (r = -0.53, p = 0.012) would seem to support our original hypothesis regarding litter drying in the slash, but we did not take litter samples from within slash piles due to the difficulty in getting a standard sample and so could not affectively test that hypothesis. Instead, it is possible that the reduced litter moisture in areas that were previously occupied by dense *L. sinense* stands is due to increased solar radiation at ground level following cutting operations. We also observed that some of the plots with lower pre-cut *L. sinense* density tended to have greater levels of green ground cover (forbs and grasses), which likely increased moisture at ground level and could affect fire behavior.

Slash reduction. We observed limited slash consumption during the prescribed fires, although some localized areas had greater consumption. The fires tended to burn under slash that had a low packing density and few suspended leaves (i.e., lots of dead space with few fuels to carry the fire vertically), whereas slash consumption tended to be greater in areas where the slash was denser and there were more suspended leaves. The comparison of pre- vs. post-treatment slash counts in our 2-m long transects showed a significant reduction in the 1-hr size class fuels, with post-treatment counts showing a reduction of 12.09 ± 2.25 branches from the pre-treatment mean of 35.63 ±3.93 branches ($p < 0.001$, Figure 3.5). Interestingly the 10-hr size class fuels showed a small but statistically significant increase of 1.88 ± 0.35 branches from the pre-treatment mean of 5.74 ± 0.52 branches ($p < 0.001$, Figure 3.5). This is likely due to measurement error. When measuring the transects we were careful not to disturb the arrangement of slash and leaf litter to avoid affecting the burn pattern, but this meant we could not measure the diameter of every branch (to ensure proper size class binning), and some branches were likely obscured by leaves. After the fires burned off some/most of the leaf litter and other fine fuels it is possible that previously obscured branches became visible and were counted for the first time in the post-treatment survey. This also suggests that the difference between the pre- and post-treatment 1-hr class tallies may be greater than our data show due to missed counts of 1-hr fuels in the pre-treatment survey (which subsequently burned up in the fire and thus were never counted). The 100-hr fuel class showed a decrease of 0.19 ± 0.08 branches from a pre-treatment mean of 0.90 ± 0.14 ($p = 0.023$, Figure 3.5). The 100-hr fuels were likely minimally affected by survey error due to their large size, which allowed easier identification even under dense slash.

Due to patchy and incomplete burns, only 63% of the slash transects were affected by the fires. By comparing the transects that showed at least some signs of scorching versus those that were not scorched, we could evaluate how much of the reduction in slash between the pre- and post-treatment surveys could be attributed to the fires versus natural decomposition (several months passed between surveys) or systematic survey error. In these analyses the interaction between survey (pre and post) and treatment (scorch vs. no-scorch) represents the effect of the fire on the difference between the pre- and post-treatment surveys. The 1-hr size class fuels showed a significant interaction, with fire removing 9.80 ± 4.59 more branches than natural decomposition alone ($p = 0.035$, Figure 3.6). In the scorched transects there were 15.71 (± 3.05) fewer 1-hr branches, on average, recorded during the post-treatment survey ($p < 0.001$), whereas in the no-scorch transects there was only a reduction of 5.92 (± 2.97) 1-hr branches on average, and the difference was only marginally significant ($p = 0.054$). The 10-hr and 100-hr size classes did not have significant interactions between surveys and treatments (p = 0.281 and 0.31, respectively), which shows that the fire did not significantly affect the slash in these size classes.

Although we hypothesized that smaller diameter slash would be reduced to a greater extent than larger classes, we were surprised by the lack of consumption of 10-hr sized fuels. The finding that 1-hr hardwood fuels are more prone to consumption that 10- or 100-hr fuels is unsurprising and has been reported elsewhere in the literature (Kolaks et al. 2004). However, Kolaks et al. (2004) also reported a significant reduction in 10-hr fuels. This discrepancy could be explained by either methodological issues (discussed above) and/or less intense fires in our study. The limited slash consumption in our study could also be partially explained by the fact that most of our burns occurred months after leaf drop, which meant that there was plenty of time for heavy rains—and in some cases flooding—to knock down the leaves that had been suspended in the slash. Without these "ladder" fuels the fires often burned along the ground under the slash piles without consuming significant amounts of the fuel above. Burning sooner after leaf drop (i.e., late fall) may improve slash consumption because there will be more suspended litter, but this assumption has not been tested. Our sampling method also missed some areas where the slash was consumed more effectively in localized hotspots.

It is possible that the reduction in fine fuels could help reduce search effort for re-sprouts during follow-up IPT, but the presence of the larger diameter slash will still impede movement in cut areas. The increased light availability at ground-level following the removal of leaf litter and other fine fuels may differentially benefit *L. sinense* sprouts and native species, although how this affects long term recovery is yet to be determined. Another possibility worthy of future study is that burning too soon after herbicide application could actually reduce the effectiveness of the treatment by preventing adequate translocation of the herbicide within the plant's tissues (Urbatsch 2000).

Conclusions. Overall, we documented a low success rate in terms of fires that were able to spread across most of a plot (≥70%) without extinguishing (n=3), and half of our fires died out after only burning 10% or less of the plot (Appendix 1). These findings show that fires in bottomland hardwood forests are likely to be patchy even at small scales. Our threshold analysis (Table 3.3) suggests that bottomland areas with >50% of the total basal area comprised of fire-prone species tended to burn at least marginally well, although this was not a consistent rule and there were some exceptions. Although our bottomland hardwood plots were somewhat unique in that most contained a layer of *L. sinense* slash, the lack of a significant relationship between slash loading and burn percent suggests that our findings may have more general implications for fire ecology and management in bottomland hardwood forests (see below). That being said, more research is needed to determine whether our findings hold true in bottomland hardwood forests unaffected by *L. sinense* invasion. What role fire may play in the natural ecology of bottomland hardwood systems and how fire prescriptions could be tailored to meet management objectives should also be the subject of future research.

A single prescribed fire was not able to significantly reduce moderate to large slash size classes, although it did reduce fine fuels. Burning may only marginally increase the efficiency of post-cut IPT on *L. sinense* re-sprouts. In areas where the leaf litter and small diameter slash has been burned off, it may be easier to locate re-sprouting plants, but operators will still be impeded by the presence of moderate to large diameter slash. More research is needed to determine whether burning can significantly improve the efficiency or effectiveness of follow-up IPT, including research on how the timing of burning could improve slash consumption. Burning may also affect re-sprouting, which we are currently assessing as part of related research.

Implications for Fire Management in Bottomland Forests. The species that we considered fire-prone (Table 3.1) are usually associated with ecological zones IV, V, and VI of bottomland forest wetlands, which include a range of sites from seasonally and temporarily flooded hardwood wetlands to the intermediately flooded upland transition zone, and may be most common on higher micro-topographic ridges (Sharitz and Mitsch 1993). These species are primarily considered late-successional, but some likely benefit from canopy disturbance to aid in regeneration (Carey 1992*a*; Carey 1992*b*; Coladonato 1992*b*; Sharitz and Mitsch 1993; Sullivan 1993). How common these species are is site-specific and will depend on local stand history and successional dynamics, topography, and hydrology (Sharitz and Mitsch 1993). Among our plots, *Q. pagoda* was the most common species, followed by *L. styraciflua*, water oak (*Quercus nigra* L.), and tulip poplar (*Liriodendron tulipifera* L.; Appendix 2). These plots were not random samples of the floodplain and plot-specific stand histories are unknown. The plots occupied similar topographic positions near the riverbank (Site C) or oxbow bank (Sites A and B; Figure 3.2), although there were differences in flooding frequency among plots due to small micro-topographic variations. We observed that tree assemblages can change drastically within a short distance, implying that large scale fires will be patchy in these systems. Low-lying areas between ridges within the floodplain most likely serve as barriers to fire spread due to greater moisture retention and tree assemblages with less flammable leaf litter. The influence of complex floodplain micro-topography, hydrology, and forest composition on large-scale fire patterns would be an interesting subject for future research, which could inform estimates of appropriate fire return intervals. There is currently little information available on appropriate fire return intervals in bottomland forests; although Gagnon (2009) discusses 5 -13 years as a possibility based on historical records and the ecology of *A. gigantea*, with fires likely related to droughts (which reduce barriers to fire spread), canopy disturbances, and human activities. However, how frequently bottomland hardwood forests burn in the absence of *A. gigantea* stands serving as a contiguous fuel source, as well as how bottomland hardwood communities respond to different return intervals, is worthy of future study.

We again note that our classification of fire-prone species was based on subjective assessments of leaf traits (guided by Kreye et al. 2013 and Kreye et al. 2018), and the physiological tolerance of these species to fire was not formally assessed. We observed some top-kill of native tree saplings, and very light scorching at the bases of some large canopy dominant trees (including some fire-prone species). Fire damage may not become apparent for several years (Hepting 1935), so further monitoring of these plots is necessary to determine whether this minor scorching develops into long-term damage. It has been suggested that flammable leaf traits that promote fire are an evolutionary adaptation that confers a competitive advantage to species that can survive or quickly recover from fire (Mutch 1970), however this hypothesis is controversial and lacking strong supporting evidence (Bowman et al. 2014). It is important to consider that in this study the term “fire-prone” is relative, since the majority of our plots did not burn well. Whether the bottomland species that were labeled as fire-prone in this study benefit from fire, and if so at what intensity/ return interval, is worthy of further research. The species that we considered fire-prone have a range of fire damage resistance, including the “moderately resistant” swamp chestnut oak (*Q. michauxii* Nutt.)—although literature on this species tends to lump it with *Q. montana* Willd. (Carey 1992*b*) and it is possible that the upland variant is more resistant to fire than the bottomland species. On the other end of the spectrum is mockernut hickory (*C. tomentosa* [Lam.] Nutt.), which is considered “extremely sensitive” to fire (Coladonato 1992*b*). The other fire-prone species in this study fall somewhere between these two species in terms of fire tolerance (Carey 1992*a*, Sullivan 1993, Coladonato 1992*a*), and all of these species will re-sprout following fire (Carey 1992*a*, Carey 1992*b*, Coladonato 1992*a*, Coladonato 1992*b*, Sullivan 1993). Fire may promote regeneration in *Q. pagoda* (Carey 1992*a*) and *Q. michauxii* (Carey 1992*b*), something predicted by the “fire-oak hypothesis” (Brose et al. 2013). More research on how fire may influence hardwood regeneration and survival in bottomland forests is warranted.

Another important consideration when using prescribed fire as part of an invasive plant management program is how other invasive species may respond to the newly created stand conditions. We observed Japanese stiltgrass (*Microstegium vimineum* [Trin.] A. Camus) begin to spread from roadsides into some of our burned plots, possibly due to the increased light availability and reduced leaf litter, creating a more favorable habitat for germination and growth for this species (Glasgow and Matlack 2007). Continued monitoring of these plots will be needed to determine whether other invasive species take advantage of these newly created stand conditions.

References

[NRCS] Natural Resources Conservation Service, US Department of Agriculture (2018) Web Soil Survey. https://websoilsurvey.sc.egov.usda.gov/?referrer=Citation.htm-HomeLink1

Andrews PL (2012) Modeling wind adjustment factor and midflame wind speed for Rothermel's surface fire spread model. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station Gen. Tech. Rep. RMRS-GTR-266. 39 p

Bates D, Maechler M, Bolker B, Walker S (2015) Fitting linear mixed-effects models using lme4. Journal of Statistical Software 67(1): 1-48

Benez-Secanho FJ, Grebner DL, Ezell AW, Grala RK (2018) Financial trade-offs associated with controlling Chinese privet (*Ligustrum sinense* Lour.) in forestlands in the southern USA. Journal of Forestry 116(3): 236-244

Bowman DMJS, French BJ, Prior LD (2014) Have plants evolved to self-immolate? Frontiers in Plant Science Volume 5 Article 590 doi:10.3389/fpls.2014.00590

Brose PH, Dey DC, Phillips RJ, Waldrop TA (2013) A meta-analysis of the fire-oak hypothesis: does prescribed burning promote oak reproduction in eastern North America? Forest Science 59(3): 322-334

Brown JK (1974) Handbook for inventorying downed woody material. Ogden, Utah: USDA Forest Service General Technical Report INT-16. 25 p

Cambi M, Certini G, Neri F, Marchi E (2015) The impact of heavy traffic on forest soils: A review. Forest Ecology and Management 338: 124-138

Carey JK (1992*a*) *Quercus falcata*, *Q. pagoda*. In: Fire Effects Information System, US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database /feis/plants/tree/quefal/all.html

Carey JK (1992*b*) *Quercus michauxii, Q. montana*. In: Fire Effects Information System, US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database/feis/plants /tree/quespp3/all.html#FIRE%20ECOLOGY

Cofer MS, Walck JL, Hidayati SN (2008) Species richness and exotic species invasion in middle Tennessee cedar glades in relation to abiotic and biotic factors. The Journal of the Torrey Botanical Society 135(4): 540-553

Coladonato M (1992*a*) *Carya cordiformis*. In: Fire Effects Information System. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database/feis/plants /tree/carcor/all.html#FIRE%20ECOLOGY

Coladonato M (1992*b*) *Carya tomentosa*. In: Fire Effects Information System. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database/feis/plants/tree/cartom/all.html#FIRE%20ECOLOGY

Dormann CF, Elith J, Bacher S, Buchmann C, Carl G, Carré G, García Marquéz JR, Gruber B, Lafourcade B, Leitão PJ, Münkemüller T, McClean C, Osborne PE, Reineking B, Schröder B, Skidmore AK, Zurell D, Lautenback S (2013) Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. Ecography 36: 27-46

Fox J, Weisber S (2011) An {R} comparison to applied regression. 2nd ed. Thousand Oaks, CA: Sage

Gagnon PR (2009) Fire in floodplain forests in the southeastern USA: insights from disturbance ecology of native bamboo. Wetlands 29(2): 520-526

Glasgow LS, Matlack GR (2007) The effects of prescribed burning and canopy openness on establishment of two non-native plant species in a deciduous forest, southeast Ohio, USA. Forest Ecology and Management. 238: 319-329

Gooding G, Langford JR (2004) Characteristics of tree roosts of Rafinesque’s big eared bat and Southeastern bat in northeastern Louisiana. Southwestern Association of Naturalists 49(1): 61-67

Green BT, Blossey B (2012) Lost in the weeds: *Ligustrum sinense* reduces native plant growth and survival. Biological Invasions 14:139-150

Grove E, Clarkson BD (2005) An ecological study of Chinese privet (*Ligustrum sinense* Lour.) in the Waikato region. Hamilton, New Zealand: Centre for Biodiversity and Ecology Research Contract Report No. 41. 17 p

Hanula JL, Horn S, Taylor JW (2009) Chinese privet (*Ligustrum sinense*) removal and its effect on native plant communities of riparian forests. Invasive Plant Science and Management 2:292-300

Harrell FE, with contribution from Dupont C and many others (2019) Hmisc: Harrell miscellaneous. R package version 4.2-0. https://CRAN.R-project.org/package=Hmisc

Harrison XA, Donaldson L, Correa-Cano ME, Evans J, Fisher DN, Goodwin CED, Robinson BS, Hodgson DJ, Inger R (2018) A brief introduction to mixed effects modelling and multi-model inference in ecology. PeerJ 6:e4794; DOI 10.7717/peerj.4794

Hart JL, Holmes BN (2013) Relationships between *Ligustrum sinense* invasion, biodiversity, and development in a mixed bottomland forest. Invasive Plant Science and Management 6(1): 175-186

Hellgren EC, Vaughan MR (1989) Denning ecology of black bears in a southeastern wetland. The Journal of Wildlife Management 53(2): 347-353

Hepting GH (1935) Decay following fire in young Mississippi Delta hardwoods. Washington, DC: US Department of Agriculture Technical Bulletin 494

Kane JM, Varner JM, Hiers JK (2008) The burning characteristics of southeastern oaks: discriminating fire facilitators from fire impeders. Forest Ecology and Management 256: 2039-2045

Kaufert FH (1933) Fire and decay injury in the southern bottomland hardwoods. Journal of Forestry 31:64-67

Klepac J, Rummer RB, Hanula JL, Horn S (2007) Mechanical removal of Chinese privet. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station SRS-43

Kolaks JJ, Cutter BE, Loewenstein EF, Grabner KW, Hartman GW, Kabrick JM (2004) The effect of thinning and prescribed fire on fuel loading in the central hardwood region of Missouri. Pages 168-178 *in* Proceedings of the 14th Central Hardwoods Forest Conference. Wooster, OH: Gen. Tech. Rep. NE-316. Newtown Square, PA: U.S. Department of Agriculture. Forest Service. Northeastern Research Station

Kottek M, Grieser J, Beck C, Rudolf B, Rubel F (2006) World map of the Köppen-Geiger climate classification updated. Meteorologische Zeitschrift 15(3):259-263

Kreye Jk, Varner JM, Hamby GW, Kane JM (2018) Mesophytic litter dampens flammability in fire-excluded pyrophytic oak-hickory woodlands. Ecosphere 9(1): 1-10

Kreye JK, Varner JM, Hiers JK, Mola J (2013) Toward a mechanism for eastern North American forest mesophication: differential litter drying across 17 species. Ecological Applications 23(8): 1976-1986

Krynak KL, Burke DJ, Benard MF (2017) Rodeo™ herbicide negatively affects Blanchard's cricket frogs (*Acris blanchardi*) survival and alters the skin-associated bacterial community. Journal of Herpetology 51(3): 402-410

Kuznetsova A, Brockhoff PB, Christensen RHB (2017) lmerTest package: tests in linear mixed effects models. Journal of Statistical Software 82(13): 1-26

Lowenstein NJ, Lowenstein EF (2005) Non-native plants in the understory of riparian forests across a land use gradient in the Southeast. Urban Ecosystems 8:79-91

Merriam RW, Feil E (2002) The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions 4: 369-373

Miller JH, Miller KA (2005) Forest plants of the Southeast and their wildlife uses. revised ed. Athens, GA: University of Georgia Press. p 360

Mutch, RW (1970) Wildland fires and ecosystems—a hypothesis. Ecology 51(6):1046-1051

Naurertz EA, Crow TR, Zasada JC, Teclaw RM (2004) Microclimatic variation between managed and unmanaged northern hardwood forests in upper Michigan, USA. St. Paul, MN: US Department of Agriculture, Forest Service, North Central Research Station General Technical Report NC-236

Nowacki GJ, Abrams MD (2008) The demise of fire and "mesophication" of forests in the eastern United States. Bioscience 58(2):123-138

Oswald BP, Beierle MJ, Farrish KW, Williams HM, Hung I (2017) Biomass Estimations of invasives yaupon, Chinese privet, and Chinese tallow in east Texas hardwood and pine ecosystems. Forest Research 6: 208 DOI 10.4172/2168-9776.1000208

Pokswinki SM (2009) Invasive characteristics of Chinese privet (*Ligustrum sinense* Lour.) in a bay swamp in the fall line hills of east-central Alabama. Master's thesis. Auburn, AL: Auburn University. 69 p

Pollet J, Brown A (2007) Fuel moisture sampling guide. Salt Lake City, Utah: Bureau of Land Management. 30 p

R Core Team (2018) R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. https://www.R-project.org/

Rhodes GN, Israel TD (2014) Chinese privet. Knoxville, Tennessee: University of Tennessee Extension Institute of Agriculture Pasture Weed Fact Sheet W 324. 3 p

Schroeder MJ, Buck CC (1970) Fire weather: a guide for application of meteorological information to forest fire control operations. US Department of Agriculture Forest Service Agriculture Handbook 360. 229 p

Shartz RR, Mitsch WJ (1993) Southern floodplain forests. Pages 311-372 *in* Martin WH, Boyce SG, Echternacht AC, eds. Biodiversity of the southeastern United States/ lowland terrestrial communities. Hoboken, New Jersey: John Wiley & Sons, Inc.

Sullivan J (1993) *Quercus shumardii*. In: Fire Effects Information System. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. https://www.fs.fed.us/database/feis/plants/tree/queshu/all.html#FIRE%20ECOLOGY

Varner JM, Kane JM, Kreye JK, Engber E (2015) The flammability of forest and woodland litter: a synthesis. Current Forestry Reports 1:91-99

Wade D (2013*a*) Backfire technique for prescribed burning. Southern Fire Exchange Fact Sheet 2013-2. 5 p

Wade D (2013*b*) Fuel moisture and prescribed burning. Southern Fire Exchange Fact Sheet 2013-5. 4 p

Waldrop TA, Goodrick SL (2012) Introduction to prescribed fires in southern ecosystems. Asheville, NC: US Department of Agriculture Southern Research Station Science Update SRS-054. 80 p

Wilcox J, Beck CW (2007) Effects of *Ligustrum sinense* Lour. (Chinese privet) on abundance and diversity of songbirds and native plants in a southeastern nature preserve. Southeastern Naturalist 6(3): 535-550

Table 3.1. Tree species observed within plots, with fire-prone species designated with an asterisk (*) based on similarities to "pyrophytic" species in Kreye et al. (2013, 2018). These classifications were used to calculate the relative basal area (RBA) of fire-prone species in each plot.

Tree Species
*Cherrybark Oak (*Quercus pagoda* Raf.)
*Shumard's Oak (*Quercus shumardii* Buckley)
*Swamp Chestnut Oak (*Quercus michauxii* Nutt.)
*Unidentified Oak (*Quercus* sp.)
*Bitternut Hickory (*Carya cordiformis* [Wang.] K.Koch)
*Mockernut hickory (*Carya tomentosa* [Lam.] Nutt.)
Water Oak (*Quercus nigra* L.)
Hophornbeam (*Ostrya virginiana* [Mill.] K. Koch)
Ironwood (*Carpinus caroliniana* Walter)
American Beech (*Fagus grandifolia* Ehrh.)
Red Maple (*Acer rubrum* L.)
Common sweetleaf (*Symplocos tinctoria* [L.] L'Her.)
American Sycamore (*Platanus occidentalis* L.)
Boxelder (*Acer negundo* L.)
Tulip Poplar (*Liriodendron tulipifera* L.)
Muscadine grape (*Vitis rotundifolia* Michx.)
Tupelo (*Nyssa* sp.)
Sweetgum (*Liquidambar styraciflua* L.)
Sugarberry (*Celtis laevigata* Willd.)
American Holly (*Ilex opaca* Aiton)
Black Cherry (*Prunus serotina* Ehrh.)
Green Ash (*Fraxinus pennsylvanica* Marshall)
Red Mulberry (*Morus rubra* L.)
Chinaberry (*Melia azedarach* L.)
American Elm (*Ulmus americana* L.)
Slippery Elm (*Ulmus rubra* Muhl.)

Table 3.2. Correlation matrix among linear variables associated with each prescribed fire. Correlation coefficients (r) followed by p-values in parenthesis. Significant correlations ($p < 0.05$) in bold. Burn = percent burn, Temp = temperature (C°), Wind = wind speed (km h^{-1}), RH = relative humidity (%), LM = litter moisture (%), LL = litter loading (kg ha^{-1}), SL = *Ligustrum sinense* slash loading (kg ha^{-1}), FP = fire-prone species relative basal area.

	Burn	Temp	Wind	RH	LM	LL	LSL
Burn							
Temp	-0.05 (0.839)						
Wind	0.28 (0.204)	-0.27 (0.220)					
RH	-0.22 (0.318)	-0.04 (0.851)	-0.42 (0.052)				
LM	-0.36 (0.102)	-0.4 (0.067)	-0.23 (0.293)	0.05 (0.840)			
LL	**0.46 (0.033)**	-0.1 (0.656)	0.31 (0.165)	0.17 (0.452)	-0.22 (0.319)		
SL	0.35 (0.116)	-0.13 (0.564)	0.38 (0.080)	-0.31 (0.157)	**-0.53 (0.012)**	0.39 (0.073)	
FP	**0.63 (0.002)**	-0.01 (0.976)	**0.45 (0.037)**	-0.17 (0.446)	**-0.54 (0.009)**	**0.47 (0.028)**	**0.64 (0.001)**

Table 3.3. Variable means and ranges (in parentheses) associated with fire success categories. These could be used to interpret useful thresholds for future prescribed fire operations in bottomland hardwood forests.

	Failure (≤ 10%, n = 11)	Marginal (20-60%, n = 8)	Success (≥70%, n = 3)
Fire-prone Tree RBA	0.24 (0.00 - 0.56)	0.54 (0.19 - 0.95)	0.70 (0.61-0.85)
Litter Loading (kg ha^{-1})	2,445 (1,180 - 3,328)	3,137 (2,026 - 4,751)	3,671 (3,145 - 4,398)
Litter Moisture (%)	38.23 (25.01 - 52.49)	35.66 (11.81 - 66.34)	19.75 (11.70 - 29.88)
Slash Loading (kg ha^{-1})	8,999 (908 - 39,965)	21,596 (6,218 - 52,022)	19,349 (7,547 - 34,870)
Relative Humidity (%)	35.75 (16.10 - 80.10)	26.16 (16.40 - 44.00)	28.80 (25.00 - 32.30)
Wind Speed (km h^{-1})	2.08 (0.00 - 4.02)	2.62 (1.61 - 3.54)	3.49 (1.61 - 5.47)
Temperature (°C)	19.07 (12.22 - 29.78)	19.62 (13.33 - 26.50)	19.43 (18.61 - 19.94)

Figure 3.1. Example of slash created by cutting heavy *L. sinense* infestation with chainsaws. This slash can reduce the efficiency of follow-up treatments on *L. sinense* re-sprouts and seedlings. Photo courtesy of Christopher Anderson.

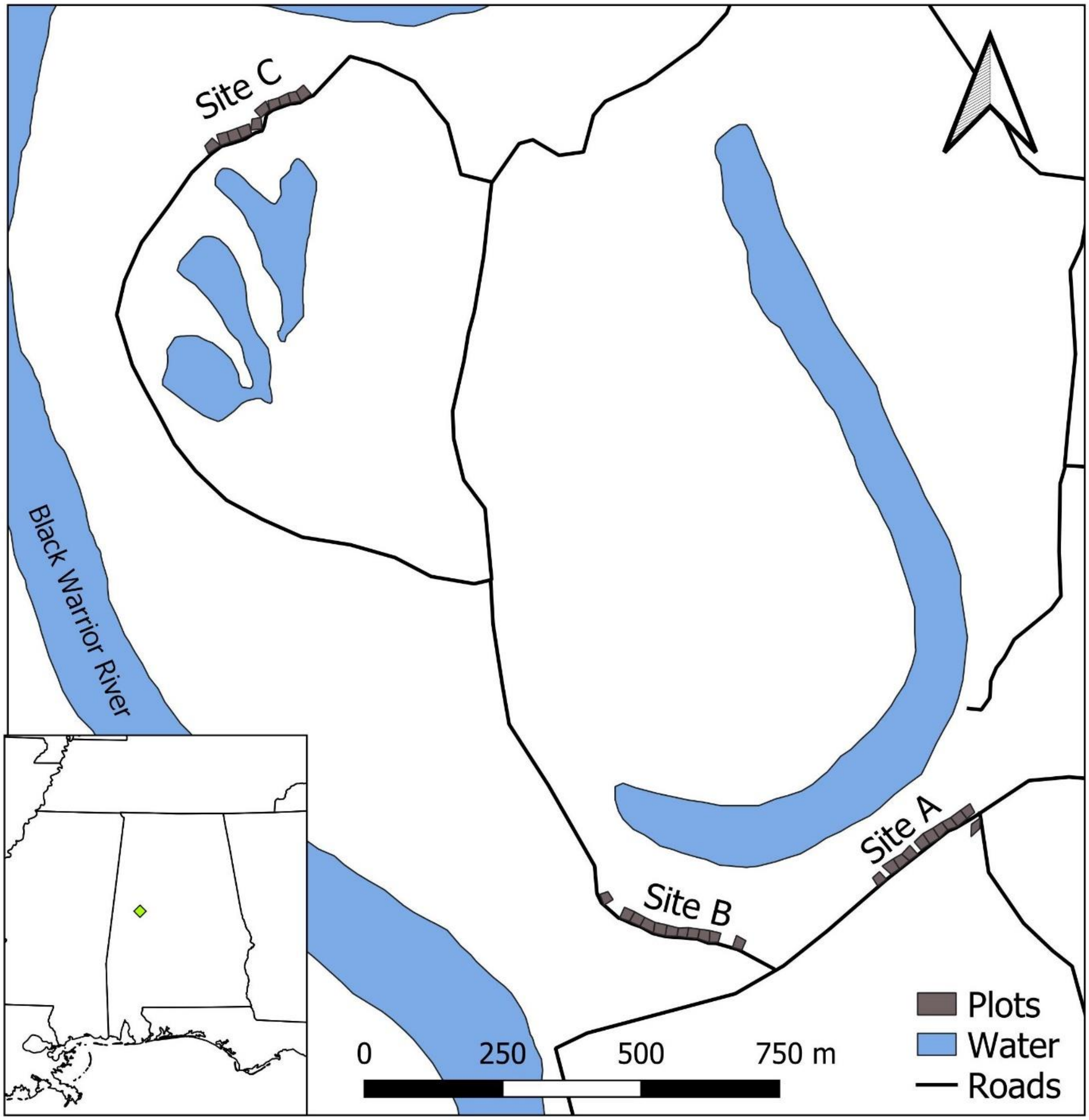


Figure 3.2. Map of the study area in the floodplain of the Black Warrior River near Moundville, Alabama where we evaluated prescribed fires. Site A = 11 plots, Site B = 11 plots, Site C = 10 plots. All three sites where similarly located near the river or an oxbow lake, however some plots were more susceptible to flooding than others due to micro-topographic variations. Inset map shows relative position of study site in Alabama, US (not to scale).

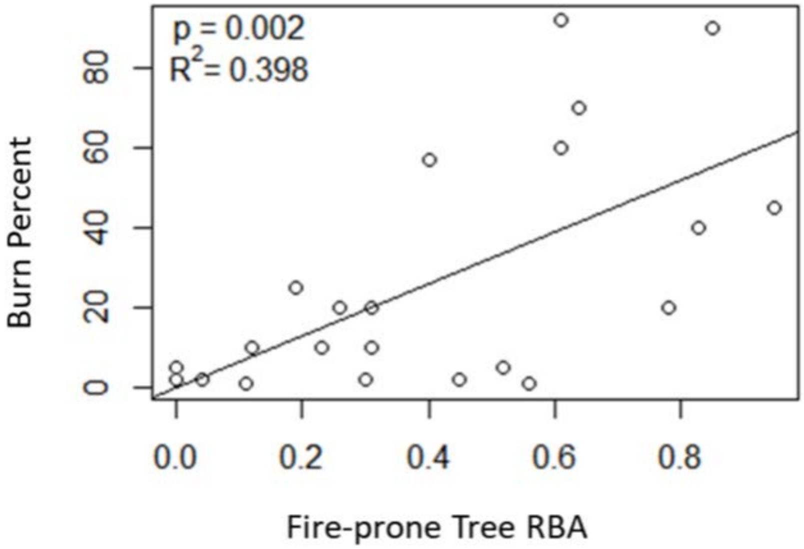


Figure 3.3. Scatter plot representing the relationship between fire-prone tree relative basal area (RBA) and burn percent.

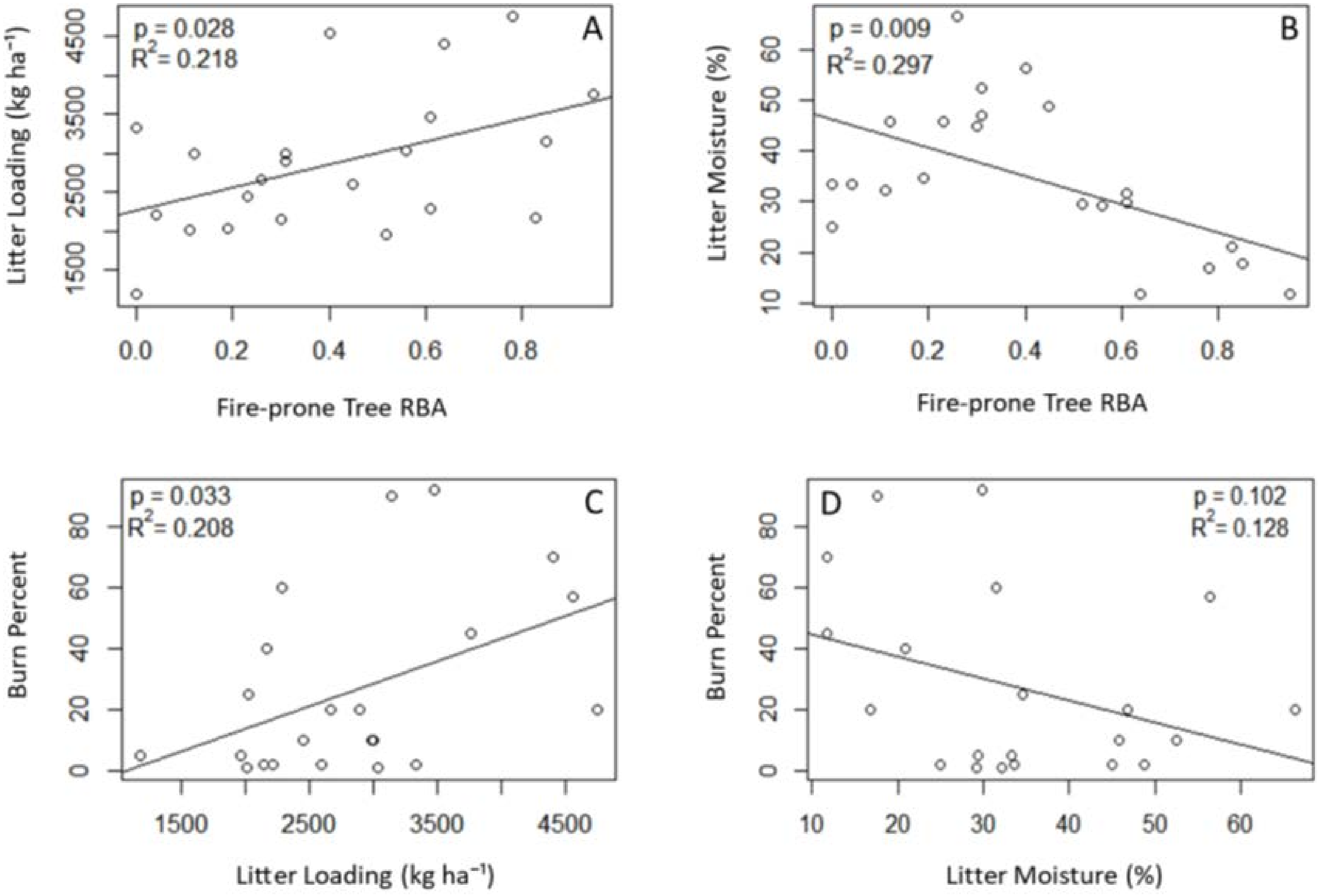


Figure 3.4. Scatter plots of the relationship between fire-prone species relative basal area (RBA) and litter loading (A), fire-prone RBA and litter moisture (B), litter loading and burn percent (C), and litter moisture and burn percent (D).

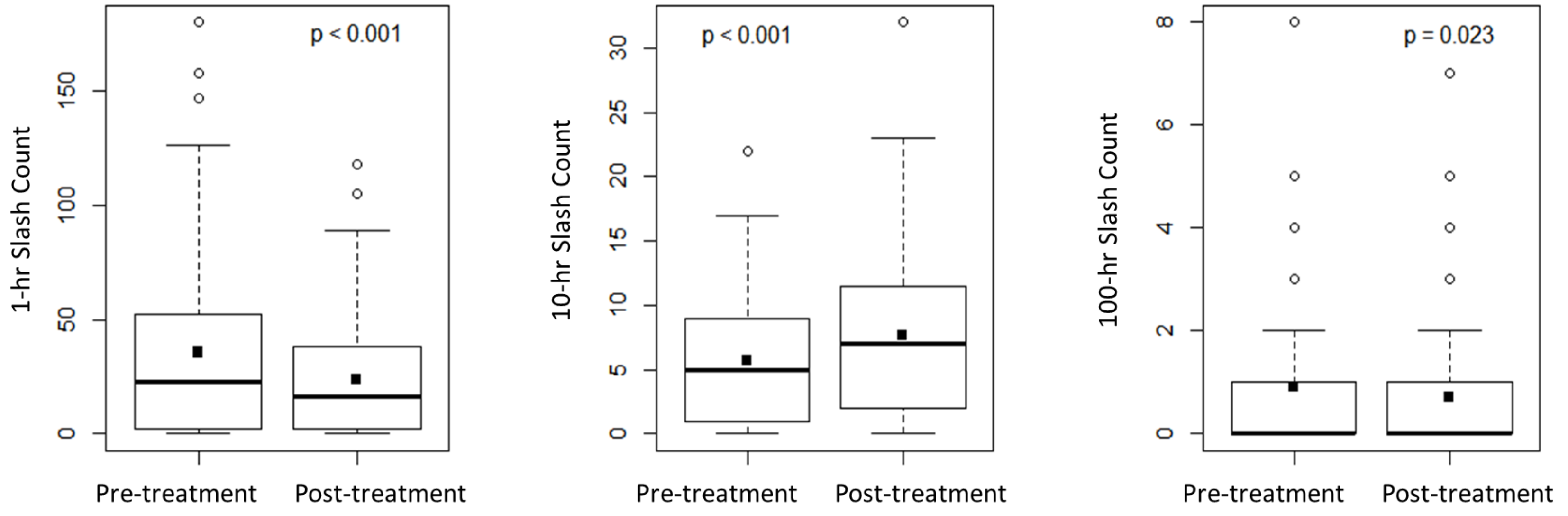


Figure 3.5. Comparison of pre- vs. post-treatment slash counts for 1-hr (0 – 0.64 cm), 10-hr (0.64 – 2.54 cm), and 100-hr (2.54 – 7.62 cm) size class fuels. Center lines of box plots represent sample medians, black squares represent sample means.

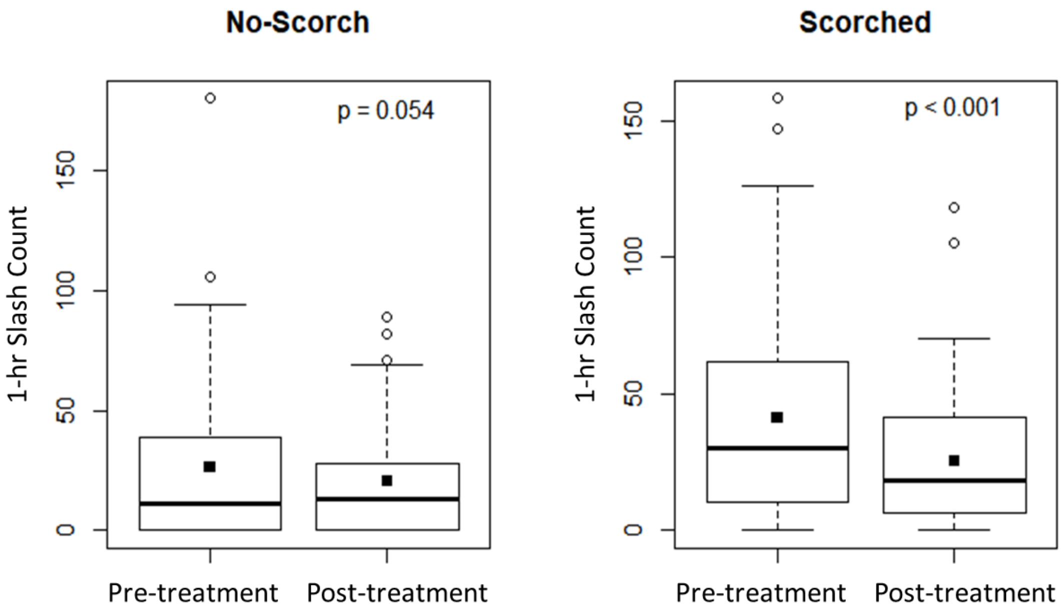


Figure 3.6. Comparison of pre- vs. post-treatment 1-hr size class slash counts in transects that were scorched by the fire vs. those that were not. Center lines of boxplots represent sample medians, black squares represent sample means.

Appendices

Appendix 1. In-stand weather, stand composition, and fuel data for each prescribed burn. RH = relative humidity, RBA = relative basal area.

Plot	Burn Date	Ignition Pattern	Burn Percent	Start RH (%)	Start Temperature (C°)	Start Wind (km hr^{-1})	Litter Moisture (%)	Litter Loading (kg ha^{-1})	Slash Loading (kg ha^{-1})	Fire-prone RBA
B-10	2018/04/02	head	1	80.1	20.8	1.0	29.3	3034.4	4104.66	0.56
A-3	2018/04/19	head	1	44.5	18.3	2.6	32.1	2014.3	2562.72	0.11
A-8	2018/03/09	back	2	40.0	12.2	3.1	45.0	2146.4	1451.84	0.30
A-5	2018/03/16	back	2	16.1	24.4	4.0	48.8	2592.3	1106.90	0.45
B-1	2018/04/20	head	2	37.8	18.4	0.0	33.5	2208.0	15922.25	0.04
B-11	2018/05/11	Unknown	2	56.9	29.8	0.0	25.0	3328.6	5604.07	0.00
B-7	2018/03/14	head	5	22.1	12.5	4.0	29.5	1959.6	39965.63	0.52
A-7	2018/03/16	ring	5	17.1	24.1	2.3	33.3	1180.0	1338.44	0.00
B-5	2018/03/08	head	10	18.8	21.7	1.1	45.8	2449.9	12544.21	0.23
B-4	2018/03/08	head	10	30.4	15.2	1.1	52.5	2986.3	13481.28	0.31
A-9	2018/03/08	ring	10	29.4	12.2	3.7	45.8	2998.4	908.80	0.12
A-10	2018/01/26	Unknown	20	44.0	15.6	1.6	66.3	2665.1	8028.59	0.26
B-3	2018/03/03	head	20	16.4	20.2	2.7	46.8	2891.3	18635.11	0.31
C-5	2019/03/22	head	20	23.9	20.0	3.5	16.9	4751.3	29225.99	0.78
A-6	2018/03/16	Unknown	25	17.1	24.1	2.3	34.6	2026.6	6218.20	0.19
B-6	2018/04/01	back	40	23.6	26.5	2.4	21.0	2162.8	17810.81	0.83
C-4	2019/03/22	back	45	23.9	20.0	3.5	11.8	3765.9	52022.23	0.95
B-2	2018/01/26	Unknown	57	40.0	13.3	3.2	56.4	4553.8	14999.74	0.40
B-8	2018/03/14	head	60	20.4	17.3	1.6	31.5	2280.9	25832.40	0.61
C-2	2019/03/22	head	70	29.1	19.7	5.5	11.7	4398.3	34870.60	0.64
C-3	2019/03/22	back	90	25.0	19.9	3.4	17.7	3145.9	15630.47	0.85
B-9	2018/03/15	head	92	32.3	18.6	1.6	29.9	3471.2	7547.03	0.61

Appendix 2. Tree species and plot-level basal areas (m^2 ha^{-1}; based on surveys of individuals >8cm DBH).

Species	A-3	A-5	A-6	A-7	A-8	A-9	A-10	B-1	B-2	B-3	B-4	B-5	B-6	B-7	B-8	B-9	B-10	B-11	C-2	C-3	C-4	C-5	Total
Cherrybark Oak (*Quercus pagoda*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	31.31	0.00	11.55	23.46	11.55	0.00	25.90	24.12	28.61	6.07	162.58
Sweetgum (*Liquidambar styraciflua*)	1.73	14.46	0.00	0.00	7.44	19.12	11.72	12.34	0.00	1.92	3.59	13.97	1.83	0.00	0.00	0.23	6.12	4.57	10.59	0.00	0.00	0.33	109.97
Water Oak (*Quercus nigra*)	0.00	0.00	13.38	0.00	0.00	9.38	0.00	20.70	3.66	0.00	0.00	7.22	0.00	0.00	10.58	13.50	9.33	0.50	0.48	0.00	0.00	0.00	88.75
Tulip Poplar (*Liriodendron tulipifera*)	14.22	8.83	0.00	17.63	12.79	15.08	15.35	0.43	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	84.61
Bitternut Hickory (*Carya cordiformis*)	0.00	0.00	3.74	0.00	9.22	0.32	0.00	0.00	8.64	5.69	0.00	0.00	0.00	4.17	7.66	2.14	6.00	0.00	3.77	0.00	0.99	1.45	53.78
Mockernut hickory (*Carya tomentosa*)	3.32	21.62	0.00	0.00	0.00	0.00	2.00	1.64	0.00	0.00	3.77	8.05	0.00	0.47	0.92	2.14	0.00	0.00	0.00	0.00	0.00	0.00	43.93
Sugarberry (*Celtis laevigata*)	0.14	0.00	0.00	0.00	0.20	0.96	0.00	0.00	4.76	7.95	2.38	1.62	2.55	6.18	0.00	0.00	0.00	4.07	1.65	0.00	1.21	0.00	33.67
Shumard's Oak (*Quercus shumardii*)	0.00	0.00	0.00	0.00	0.00	6.45	8.71	0.00	0.00	0.00	0.00	0.00	0.00	5.91	1.17	3.79	4.88	0.00	0.00	0.00	0.00	0.00	30.91
American Sycamore (*Platanus occidentalis*)	0.00	0.00	0.15	0.00	0.00	8.79	0.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.08	0.00	7.24	0.00	0.00	0.00	0.00	19.46
Ironwood (*Carpinus caroliniana*)	0.74	0.40	0.00	0.00	1.23	3.10	2.06	1.08	0.00	0.00	0.00	0.25	0.18	0.00	0.00	0.56	0.94	1.81	0.14	0.20	0.00	0.00	12.71
Green Ash (*Fraxinus pennsylvanica*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	6.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.75	1.76	0.00	0.00	11.55
American Beech (*Fagus grandifolia*)	7.26	0.16	0.00	1.40	0.00	0.00	0.00	2.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	11.16
Boxelder (*Acer negundo*)	0.64	0.71	0.42	0.94	0.00	0.00	0.96	0.00	0.00	0.00	1.17	1.29	0.00	1.59	0.38	0.00	0.00	0.47	0.00	0.00	0.43	1.82	10.81
Red Maple (*Acer rubrum*)	1.14	2.12	1.99	0.20	0.00	0.00	0.00	0.00	1.34	0.00	0.00	0.00	0.00	0.00	0.00	2.97	0.00	0.00	0.00	0.00	0.00	0.00	9.76
American Holly (*Ilex opaca*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.96	0.00	0.00	0.20	0.00	2.10	1.85	0.00	0.38	0.62	0.00	0.00	0.00	0.00	6.12
Swamp Chestnut Oak (*Quercus michauxii*)	0.00	0.00	0.00	0.00	0.00	0.94	0.00	0.00	2.41	0.00	0.00	0.00	0.00	0.00	0.00	1.27	0.00	0.00	0.00	0.00	0.00	0.00	4.62
American Elm (*Ulmus americana*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.00	0.22	1.46	0.00	0.00	0.18	0.00	0.00	0.00	2.30	0.00	0.00	4.45
Red Mulberry (*Morus rubra*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.98	0.00	1.34	0.00	1.37	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.69
Slippery Elm (*Ulmus rubra*)	0.00	0.00	0.00	0.00	0.00	0.44	0.00	0.00	0.00	0.00	1.17	0.50	0.00	0.00	0.36	0.00	0.00	0.52	0.00	0.00	0.00	0.00	2.99
Tupelo (*Nyssa* sp.)	0.00	0.00	0.00	0.92	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.28
Hophornbeam (*Ostrya virginiana*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.00	0.54	0.44	0.00	0.00	0.00	0.00	0.00	1.27
Black Cherry (*Prunus serotina*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.24
Common sweetleaf (*Symplocos tinctoria*)	0.00	0.00	0.35	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.00	0.00	0.00	0.00	0.00	0.76
Chinaberry (*Melia azedarach*)	0.49	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.49
Unidentified Oak (*Quercus sp.*)	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43
Muscadine grape (*Vitis rotundifolia*)	0.00	0.00	0.00	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16
Total	29.69	48.29	20.02	21.38	30.88	64.58	41.00	39.79	27.81	18.15	12.08	34.99	38.05	20.42	34.84	53.87	39.92	19.81	46.28	28.39	31.23	9.68	