Evaluating the Distribution and Dispersal of Invasive Woody Vegetation Related to Land Use in Auburn, Alabama

by

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Abstract

The spread of invasive species in riparian areas is an international problem and has resulted in significant loss of native species in riparian areas worldwide, including the east Alabama region. The objective of this study was to find the frequency and dominance of these invasive shrubs by surveying the extent of *Ligustrum sinense* (Chinese privet), *Elaeagnus pungens* (Silverthorn), and *Tridica sebifera* (Chinese Tallow tree) and its potential relation to urban land use in riparian areas of Auburn, AL. Historical land use may also be important to the current distribution of invasive plants. Using Chinese privet (one of the region's most pervasive species), we explored potential relationships between historical land use and colonization of Chinese privet in Auburn, AL. This study indicates that changes in distribution and richness of invasive plants are occurring in response to urban land use change in riparian areas. Urban sites were positively associated with dominance of invasive plants, primarily Chinese privet. Results of this research highlights the impacts of urbanization and historical land use on colonization and distribution of invasive plants in riparian forests.

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List of Abbreviations

DBH	Diameter at Breast Height
EPA	Environmental Protection Agency
FIA	Forest Inventory Analysis
GIS	Geographic Information System
LULC	Land Use/Land Cover
NOAA	National Oceanic and Atmospheric Administration
NRCS	National Resources and Conservation Service
USCB	United States Census Bureau
USDA	United States Department of Agriculture

Chapter 1: EVALUATING THE CURRENT DISTRIBUTION OF INVASIVE WOODY VEGETATION RELATED TO LAND USE

INTRODUCTION

An invasive species is defined as a species that is *''non-native (or alien) to the ecosystem under consideration and whose introduction causes or is likely to cause economic or environmental harm or harm to human health''* (USDA 2015). Invasive species have caused enormous and, under some situations, irreversible impacts to indigenous species, native communities, and related ecosystem services in most of the parts of the world (Wang and Grant 2012). They also reduce related ecosystem functions by degrading forest lands, wetlands, and agricultural habitats. They reduce the native vegetation and biodiversity, reduce the forest productivity, and disturb wildlife habitat. Once these species are successful in colonizing a landscape, it is often considerably expensive and difficult to remove them (Shuster et al. 2005). It has been estimated that approximately 5000 nonnative plant species have invaded native forest and shrub ecosystems in the United States (Parker et al. 2009).

In addition to environmental degradation, invasive plants have lead to major economic damages. Pimentel et al. (2000) identified 79 exotic species that caused almost $97 billion in damage during the period 1906-1991 in the U.S. In the same research, it was also reported that approximately 50,000 non-indigenous species have caused disturbances at a cost around $137 billion in damage per year. The main cause of this economic loss is that once invasive plants are spread in a landscape, they importantly replace productive forage plants and reduce farmland values. Also, restoration and control of invasive species are costly (Vila and Pujadas 2001).

The invasion of an ecosystem is a function of factors: the amount of propagules introduced to a new location, the features of nonnative species such as rapid growth rate, adaptation to harsh conditions, and the vulnerability of the environment against nonnative plant infestation (Davis et al. 2000). Invasibility of an area is a property which depends on several factors such as the region's climate, the existing degradation to the environment, and the susceptibility of the native plants to competition (Lonsdale 1999). Lonsdale also stated that to determine if a region is susceptible, it is necessary to examine the number of invasive species immigrating and becoming extinct, and then determine the dispersal, establishment, and survival of the invasive plant species. It is important to take the life history characteristics of invasive species into account in order to assess the potential extent of infestation. In some circumstances, a single non-indigenous plant species can infest an entire area in a community. For instance, yellow star thistle (*Centaurea solstitalis*) dominates almost 4 million hectares of grassland in California, thereby decreasing the overall productivity of native grassland in the ecosystem (Pimentel et al. 2000).

According to USDA Economic Research Service records, in 2009 the forestlands of the southern United States comprise 62% of U.S. timber supply while also contributing important ecological services such as forest productivity, recreation, wildlife habitat, and biodiversity. Invasive plant species in these forests have degraded wildlife habitat, replaced native plant species, and decreased the viability of numerous forest management activities (Wang and Grant 2012). The loss of forest ecological services by invasive species has often occurred when farmlands introduced invasive species, and with horticultural and ornamental usage of nonnative plants became more prevalent in the southern U.S. Also, ambiguity in landscape invasions, such as comparison of invaded and non-invaded areas or richness of plant diversity and dispersal, have somewhat diminished the impact of invasive plants' spread on landscapes (Greene and Blossey 2012).

The potential for plant infestations to replace native plant communities in a forested area can have important consequences on community and ecosystem properties. For instance, because of belowground competition between native and non-native plant nutrient components, such as nitrogen (N) may become insufficient for native plants (Ehrenfeld 2003). Thus, changes in nutrient levels and cycling can result in alterations to vegetative composition (Ehrenfield 2003). The replacement of native plant assemblages by invasive plants can also alter ecosystem properties such as net primary production (NPP), biotic decomposition, and water uptake (Brantley 2008). Forest biodiversity is also clearly impacted with the increased infestation by invasive species. Over the last few decades, the characteristics of exotic invasive species have received broadening interest from conservationists, ecologists, and land managers Gordon (1998). Gordon also stated that the shifts in community dominance and increasing colonization of non-native plant specimens have been recognized as a threat to abundance and persistence of species. Invasive species also significantly threaten many plant species listed under the Endangered Species Act (Wilcove et al. 1998).

Land Use Change and Invasive Species

The relationship between land use and invasive species is important because of the long history of land use change in the United States. Human activities in the Southeast U.S. have altered and disturbed forest lands, especially since European settlement. There have been four main time periods of dominant land disturbance and change during this time (Wear 2002):

1) agricultural conversion from 17^{th} century to the 19^{th} century,
2) conversion from agriculture to timber management in 20^{th} century,

3) farm abandonment and natural reforestation between 20th and 21st centuries,
4) urbanization and environmental disturbance in 21st century.

Urban land use has considerably increased in the Southeast U.S. over the past decades (Rusch et al. 2003). Urbanization can have a substantial impact on ecosystems, by disturbing native habitat, altering hydrology, and increasing the abundance of invasive species (Lundgren et al. 2004, Loewenstein and Loewenstein 2005, Grove and Clarkson 2005). Groffman et al. (2003) stated that urbanization was one of the most threatening and harmful factors affecting vegetative composition, and despite being shown to impact waterways and floodplains, there were still insufficient analyses to address the dramatic impacts on watersheds. Forecasts report that urban land use is expected to increase from almost 8.1 million hectares in 1992 to 22.3 million hectares in 2020. By 2060, it is projected that urban lands could be between 30 - 43 million hectares in the Southern United States (Wear and Greis 2013). In the United States, about 11.8 million hectares of forest lands are expected to be replaced with urban lands between 2000 and 2050. (Wear and Greis 2013).

Urbanization is a factor affecting the introduction and establishment of invasive species. in 2010, United States Census Bureau (USCB) defined the term "urban" as '*territory, population, and housing units located within an urbanized area or an urban cluster which has a population density of at least 1,000 people per square mile and surrounding census blocks with an overall density of at least 500 people per square miles'*. Rural also defined as *'all territory, population, and housing units located outside of urban areas and urban clusters*' (USCB 2010). Between 1950 and 2000, urbanization in the United States rose from 1% to 2%, and low-density urban settlement increased from 5% to 25% of land cover (Pizarro et al. 2010). For example, between 2000 and 2010, there were notable increases in the area of urban and urbanized areas in the

United States and the respective populations associated with urban lands (Table 1.1). Urbanization often increases the occurrence of invasive species as homeowners have historically planted ornamental, non-native landscapes around houses. Many species that were initially planted around houses eventually spread to nearby lands, and may become invasive (Mack and Erneberg 2002). Consequently, urbanization commonly leads to a increased dispersal of invasive plant species in areas which were rural and previously unexposed to the species. Although this general pattern is understood, studies about urbanization and its specific influence on invasive plant species dispersal and establishment are few (Pimentel et al. 2000, Pizarro et al. 2010). Greene and Blossey (2014) observed the floodplain forests of the Piedmont region of South Carolina to evaluate the relationship between distribution and abundance of Chinese privet and urbanization. Greene and Blossey (2014) also reported that invasion in urban and forested watersheds could be slightly explained by increased developments. In addition to urbanization, agricultural activities, logging, and many other land uses may contribute to the dispersal of invasive plant species. Simply stated, the rapid results of long term and intensive land modifications are not amenable to recovery and in many cases cause irreversible alterations to landscapes and their habitats (McKinney 2006). This is potentially problematic because in the U.S. and other places in the world, populations are becoming increasingly urban and populations have increasingly migrated from rural to urban areas.

The consequences of land use alterations are varied and complex, but have been shown to cause many problems to local and global ecosystems (Kuhman 2009, Davis et al. 2000, Stohlgen 2003). The displacement of vegetation is one of the most significant results of land use change in urban areas. Another effect of land use change is invasive species infestation in native forests including riparian areas (Davis et al. 2000). Light availability, soil nutrition and infiltration rate, and litter structure in the forested area are often correlated with land use change, and these factors can increase forest susceptibility to invasion (Kuhman 2009).

Contemporary landscape studies demonstrate that land use patterns such as roads, residential development activities, and agricultural improvements cause rapid spread of invasive species propagules to nearby forest regions (McKinney 2006, Lundgren 2004, Davis et al. 2000). Moreover, fragmentation impacts wind and precipitation, and surface run-off in urban and agricultural landscapes, and propagules of invasive plants can more easily spread out and become established in new lands. Also, the seeds of invasive plants in urban and agricultural lands can be dispersed by birds more rapidly than rural landscapes because food for birds in urban areas is less than in rural forest lands (Heywood 1989).

Although land use disturbance has been shown to affect invasive species expansion, there may be variation as to how susceptible forested areas are to invasion. According to Stohlgen (2003), a high diversity in plant species might make a forest more resistant to invasion in urbanizing areas because the competition between species increases and some native plants cannot survive. Similarly, Hobbs and Huenneke (1992) noted that *''a diverse assemblage of plant species might reduce the potential impact of invasive specimens in an ecosystem despite the integral role of habitat disturbance on a community*''. They also claimed that most of the exotic plants that have been introduced to urban and rural areas in the U.S. have proliferated because of their resistance to constraining environmental conditions and their rapid adaptation capabilities. For instance, Rossman (2001) stated that woody vine kudzu (*Pueraria lobata* Wild.) which can thrive in harsh conditions, was introduced to U.S. for erosion control and ornamental aspects. Erosion control attributes can lead to greater invasibility of native plant communities in urban and rural areas (Rossman 2001). New settlements in urban areas specifically revealed that urbanization, and unexpected plant diversity in ecosystems caused infestation with many negative consequences in urban regions (Rossman 2001).

Riparian areas and Invasive Species

Riparian areas are one of the most diverse and beneficial ecosystems in the southern United States and link aquatic systems with terrestrial lands (Naiman and Decamps 1990). These areas are common throughout the region and it has been estimated that 663 million km of rivers and streams occupy the southeastern U.S. (EPA 2015). Riparian ecosystems are substantial sources of water, nutrients, sediments, and organic matter for adjacent floodplain ecosystems (Burton 2006). Because riparian areas border the stream channel and the terrestrial zone, it is often affected by the landscape features of surrounding lands. Thus, in order to assess the dispersal of flora within riparian corridors, the stream margins and surrounding uplands need to be considered (Pollock et al. 1993). Moreover, riparian areas are often corridors between diverse landscape zones such as urban and suburban areas and many diverse microhabitats can be located within riparian areas. Anthropogenic and natural disturbances, availability of moisture for proliferation of propagules, and a connected drainage network between upland and bottomland in riparian areas are some of the important factors that lead to invasive species occurrence and spread (Hood and Naiman 2000). Hood and Naiman (2000) indicated that construction near urban forest and streams, deforestation, seed dispersal by surface runoff and birds between upland and bottomland often facilitates recruitment of invasive species in riparian areas.

Many hypotheses have been proposed regarding the susceptibility of different communities to invasion and the research results derived from many field studies have shown varied results. Our understanding of invasive species and their occurrence is still insufficient to manage many nonindigenous plant specimens (Davis 2000). Although the specific mechanisms for the distribution of woody invasive species in riparian areas is often unknown, it is commonly correlated to changes in surrounding landscape, especially in previously unfragmented landscapes. Riparian areas can be more prone to be invaded because they function as corridors. Groffman et al. (2003) researched the importance of riparian corridors in the Baltimore ecosystem by analyzing water table, soil quality, and sediment deposition associated with agriculture and residential construction. It was shown that riparian corridors are prominent landscapes for invasive species which contribute to changes in riparian soils, vegetation, and microbial processes in urban areas. Since riparian areas often have high nutrient concentrations, they can be disposed to invasive plant seed establishment. As a result, the population density and cover of infested species is often significantly larger than in other ecosystems (Davis et al. 2000). The potential of nutrient concentrations in riparian areas is important because these nutrients provide pertinent conditions for competitive invasive plants to thrive and spread. Also, enriched soils may increase the resilience of invasive plants against competition with native species and harsh weather conditions (Predick and Turner 2008).

The invasion of exotic species in the region is increasing across the landscape, in spite of management practices have been applied by municipalities and managers (Wang 2009). According to Miller (2003), exotic alien plant species essentially have been checked and monitored by many researchers in the southern forests. To monitor the ecological destruction of invasive plant species, the U.S. Department of Agriculture (USDA) Forest Service's Forest Inventory and Analysis (FIA) program of the U.S. Forest Service's Southern Research Station initiated a multifaceted forest resource survey to identify the occurrence of invasive plant distribution in forest lands (Rudis et al. 2006). From plant database of USDA and NRCS in 2014, it was reported that Chinese privet (*Ligustrum sinense*) and Chinese tallow (*Triadica sebifera*) have been dispersing throughout the Southern U.S., including Alabama (Figure 1.1). A third species, silverthorn (*Elaeagnus pungens*), represents another invasive species that was commonly planted and has become naturalized in riparian zones throughout the southeast U.S.A description of each species is provided below.

Research Plant Species

Chinese privet (*Ligustrum sinense*)

Chinese privet (*Ligustrum sinense* Lour.) is a semi-evergreen to evergreen, shade tolerant shrub with growing up to 8.2 m height and 3.6 m width which has been invading riparian forests in southern United States, where it often forms monotypic stands displacing native plant communities (Hanula et al. 2009). It is a member of the *Oleaceae* family and it was first introduced into the southern United States from China in 1852 as an ornamental plant (Langeland and Burkes 1998). However, it has been escaping cultivation and has dramatically expanded into natural ecosystems (Brown and Pezeshki 2000). The dull green leaves of specimens are 0.5-6 cm long by 1.3-2.5 cm wide, and with entire or slightly wavy margins. The small, fragrant white flowers occur in clusters up to 10 cm long, and the leaves are light hairy on the underside (Grove and Clarkson 2005).

Chinese privet is capable of forming dense thickets in riparian forests by rapid colonization (Miller 2003). Miller also reported that Chinese privet is tolerant of low nutrition, high soil moisture and low light levels which makes it adaptable to canopied riparian forests. Chinese privet occupies approximately 3.5% of the total forested area in the southeastern U.S. by dominating many riparian ecosystems. The expansion of Chinese privet into new areas is correlated with seed dispersal by floodwaters, animals and birds (Ulyshen et al. 2010). Its propensity for vegetative propagation, shade tolerance, high growth rates, and copious seed production all contribute to the rapid spread of Chinese privet (Langeland and Burkes 1998).

Chinese privet displaces native plants from the understory and reduces overstory tree reproduction by competing for nutrients and light in forested lands (Kittel 2001). It is often found close to human settlement, across a wide variety of environmental conditions from dry to wet, and shady or open spaces (Grove and Clarkson 2005). Meriam and Feil (2002) researched the abundance of Chinese privet in forested lands by predicting the occurrence of privets in invaded areas to examine the relationship between invasive plants and number of native trees and herbaceous cover in floodplains. They also noted that Chinese privet significantly lowered the native plant diversity, within infested areas, biodiversity of herbaceous plants and trees were reduced by 33% of herbs and 25% respectively. Moreover, their research indicated that because of privet's capacity to out-compete and reduce native plant abundance, privet can cause significant, large scale ecosystem modifications such as changes in soil nitrogen, increase in susceptibility to fire, damaging nutrient cycles, increase in sedimentation and erosion. Greene and Blossey (2014) examined the relationship between urbanization and distribution and abundance of Chinese privet in Piedmont floodplain forests of South Carolina. They also explored the biotic and abiotic factors to determine the distribution and occurrence of Chinese privet and native plant communities. They concluded that there was a positive association between increased urbanization and Chinese privet but performance of privet was not related to urbanized watersheds or local edaphic features.

Chinese Tallow tree (*Triadica sebifera*)

Chinese Tallow tree or popcorn tree was first introduced from China to the U.S. in the 18th century in Savannah, Georgia and Charleston, South Carolina. It is a member of the

Euphorbiaceae family. During 18th century, the Foreign Plant Introduction Division of the Bureau of Plant Industry supported the planting of Tallow tree by land owners to support soap making from the sarcotesta (the seed coat) surrounding the seeds (Park et al. 2012). In China, this species has also been used for furniture, medicine, black dye, and wax for candles (Wang et al. 2009). Since its initial introduction, the range of Chinese tallow tree has expanded considerably. Bruce et al. (1997) reported that the species became dominant from east Texas to southern North Carolina, especially in coastal ecosystems. The seedlings of the tallow are capable of growing 50-70 cm in height per year, becoming mature within 3 or 4 years. Chinese tallow tree can reach 7 to 20 m in height under normal conditions (Zheng et al. 2005). It is a deciduous tree that has gray to whitish-gray bark with vertical cracks. It has leaves with smooth margins and shapes vary from circular to deltoid with 2.5-7 cm wide (Zheng et al. 2005). The 2-4 cm long flowers are monoecious, and the female flower is located in the pedicel. The species is tolerant of various soil types, as well as extremes in moisture regimes (Park et al. 2012).

In the U.S., Chinese tallow tree has been reported as an invasive tree species in the Gulf Coast states, South Carolina, North Carolina, southern Arkansas, Oklahoma, California, Tennessee, northern and central Florida, and is grown ornamentally in Arizona (Wang et al. 2009). According to Pattison and Mack (2008), the range of Chinese tallow tree could expand 500 km northward and inland of its current distribution in southeastern U.S.

Chinese tallow tree has shown mycorrhizal dependence where it occurs in contrast to native plant species (Zhang et al. 2013). It also competes with co-existing indigenous species that causes decline in the productivity of native species' seeds. (Niijer et al. 2007, Yang et al. 2013). In addition, Mccormick (2005) found that invasion by Chinese tallow reduced forest richness of native resident plants and invertebrates, and also decreased the ecosystem efficiency to productivity. In coastal prairies of southern U.S., it is considered a 'transformer' species due to its alterations of ecosystem nutrient cycling and increased primary productivity (Battaglia et al. 2009).

Silverthorn (*Elaeagnus pungens*)

Silverthorn or thorny olive is a non-native invasive species which was introduced to U.S. in 1830. Escaped populations have been observed in southern U.S. from Kentucky and Virginia south to Louisiana and Florida (Godfrey 1998, Gucker 2011). It has been commonly planted as an ornamental and in hedgerows while also used for highway plantings for almost two decades because it easily adapts to the harsh conditions in roadway medians of the southeast U.S. (Watts and Paxton 2000, Gucker 2011).

Silverthorn is a dense shrub with multiple stems, reaching 7.6 m tall and 4.6 m wide. It creates prolific and rapidly-growing stem sprouts which allow the species to grow onto adjacent plants (Miller 2003). Leaves of the species are evergreen, arranged alternately, and typically measure between 4 and10 cm long and less than half as wide. It also produces tubular form flowers that are about 1 cm long and occur in clusters of up to 3. Fruits are single-seeded drupes that are 1 to 1.5 cm long (Radford et al. 1968). In the U.S., silverthorn flowers in the fall and generates fruits in the spring season (Gucker 2011). Silverthorn is very easy to grow and adapts to most soil and growing conditions (Connie 2008). It also grows well under shade and tolerates salt spray and air pollution. It has also been planted on reclaimed mine sites because of its tolerance of harsh conditions (Gucker 2011).

Proposed Research

The spread of invasive species in riparian areas is an international problem which has resulted in significant loss of native species in riparian areas worldwide, including the east

Alabama region. Moreover, the municipalities in the area need reliable information regarding the occurrence, extent, and dispersal of invasive species and how land use change may increase the spread of these species. It is also essential to help land owners to identify emerging invasive species.

The objective of this study was to find the frequency of occurrence and dominance of these invasive shrubs by surveying the extent of *Ligustrum sinense* (Chinese privet), *Elaeagnus pungens* (Silverthorn), and *Tridica sebifera* (Chinese tallow tree) in riparian corridors across a land use gradient in east Alabama. Urbanization and historical land use may also be important to the current distribution of invasive plants. Additionally using Chinese privet (one of the region's most pervasive species), we explored potential relationships between historical land use and colonization of Chinese privet in east Alabama.

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Table 1.1: The fluctuation of population in the United States between 2000 and 2010 (USCB 2010).

Area	**Population**		**Percentage of Total Population**	
	2000	**2010**	**2000**	**2010**
United States	281,421,906	308,745,538	-	-
Urban	222,360,539	249,253,271	79.0%	80.7%
Rural	59,061,367	59,492,267	21.0%	19.3%

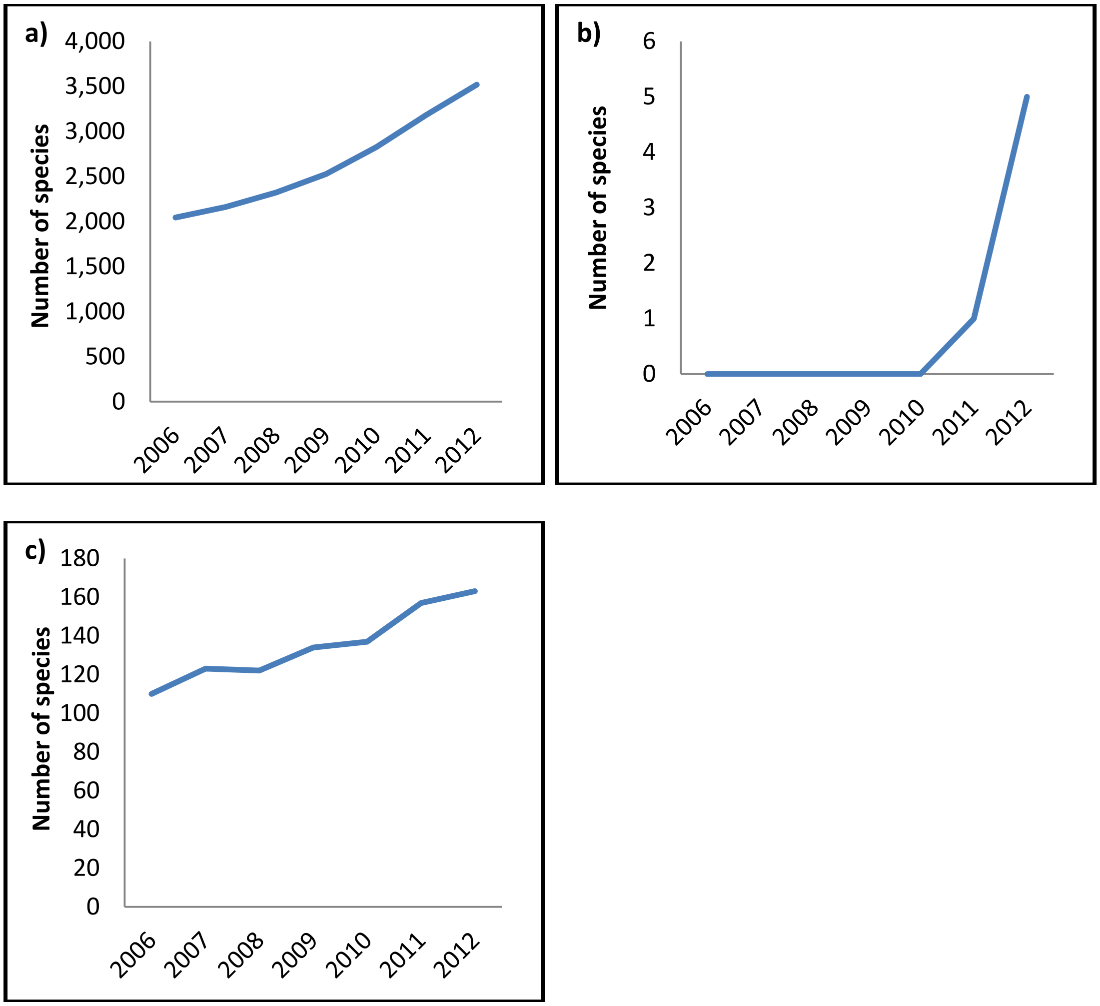

Figure 1.1: Cumulative number of a) Chinese privet, b) Silverthorn, and c) Chinese Tallow tree counts based from FIA survey data (results since 2001).

Chapter 2: EVALUATING THE DISTRIBUTION OF INVASIVE WOODY VEGETATION IN RELATION TO LAND USE IN AUBURN, ALABAMA

Abstract

Invasive species have been causing important and, under some circumstances, irreversible impacts to native species and communities, and related ecosystem services in most of the parts of the world. To address disturbances caused by invasive species occurrence, managing agencies and municipalities need reliable information regarding the occurrence, extent, and dispersal of invasive species and how land use may increase the spread of these species. It is also essential to inform and assist land owners to identify and report emerging invasive species. The objective of this study was to find the frequency and dominance of three invasive species common to riparian areas of east Alabama: *Ligustrum sinense* (Chinese privet), *Elaeagnus pungens* (silverthorn), and *Triadica sebifera* (Chinese tallow tree). Surveys of these species in riparian forests in and around Auburn, Alabama were conducted to show the relative extent of these shrubs and their relation to urban land use. It was expected to see the highest levels of invasive species in the city center with decreasing levels radiating outward into rural areas. The purpose of this research was to evaluate the current distribution of Chinese privet, Chinese tallow tree, and silverthorn along riparian areas and how urban land use may affect the presence-absence and prevalence of these non-native plant species within study sites. Further out from the city center and suburban lands, cover of both Chinese privet and silverthorn tended to decrease. In contrast, Chinese tallow tree density percent cover showed an opposite trend with landscapes close to city center often having slightly less cover. This study shows that urban land use and specifically housing density may be an important association with invasive plant species in east Alabama.

INTRODUCTION

Invasive species have been causing important and, under some circumstances, irreversible impacts to native species and communities, and related ecosystem services in most of the parts of the world (Wang and Grant 2012). They are shown to reduce ecosystem functions by degrading forest lands, wetlands, and agricultural habitats. They also replace the native vegetation and reduce biodiversity, forest productivity, and suitable wildlife habitat. Once these species are successful in colonizing a landscape, it is often considerably expensive and difficult to remove them (Shuster et al. 2005). It has been estimated that over 5000 nonnative plant species have infested native forest and shrub ecosystems in the United States (Parker et al. 2009).

Over the last several decades, according to Gordon (1998), the characteristics of exotic invasive species have received broadening interest from conservationists, ecologists, and land managers. Many invasive plants have been shown to negatively affect important ecosystem properties by replacing native species. Forest biodiversity is clearly impacted by increased infestation by invasive species (Brantley 2008, Davis et al. 2000, Stohlgen 2003, Kuhman 2009). It was reported that the shifts in community dominance and increasing colonization of non-native plant specimens have been recognized as a primary threat to local species presence and persistence. The infestation of invasive species has been shown to significantly threaten plant species listed under the Endangered Species Act in almost half of United States ecosystems (Wilcove et al. 1998).

According to Ehrenfield and Stander (2010), riparian zones are one of the most prevalent areas which contain invasive species. Riparian forests are forested corridors which act as important transformers and filters of nutrients and sediments from urban and agricultural runoff (Malanson 1993), and facilitate flood control following high rain events (Piemental et al. 2000).

The biotic communities in riparian areas are often influenced by urban land use surrounding streams (Ehrenfield and Stander 2010) and the effects can be important for sustaining aquatic resources in urban and rural landscapes.

Urban land use can have a substantial impact on riparian ecosystems, by changing habitats, altering hydrology, and increasing the abundance of invasive species (Mack and Erneberg 2002, Loewenstein and Loewenstein 2005, Wear and Greis 2013). Groffman et al. (2003) stated that urbanization was one of the most threatening factors affecting ecosystems, and despite evidence showing its impact to waterways and floodplains, there are still important knowledge gaps related to urban impacts. Urban land use can lead to biotic changes to riparian areas such as habitat degradation and hydrologic changes related to surface runoff (Jennings and Jarnagin 2002). Urbanization has also been shown to fragment the forests surrounding riparian zones that can change habitat conditions (e.g., soil conditions, light penetration) that increase the likelihood of invasive species (Tererai et al. 2015). For example, Caughlin et al. (2012) observed the abundance of an invasive plant, *Ficus microcarpa* in the South Florida, U.S. and it was demonstrated that fragmented forests increased seed dispersal and colonization of this invasive plant.

The spread of invasive species in riparian areas is an international problem and has resulted in significant loss of native species and forest function in riparian areas worldwide, including the east Alabama region. In the U.S., 216 million ha of forest resources in 13 southern states provide 62% of timber production in the United States (Battaglia et al. 2009). These resources also serve as areas of important wildlife habitat, water conservation, biodiversity, and recreational activities (Merriam and Feil 2002; Battaglia et al. 2009). To address disturbances caused by invasive species occurrence, managing agencies and municipalities need reliable information regarding the occurrence, extent, and dispersal of invasive species and how land use may increase the spread of these species. It is also essential to inform and assist land owners to identify and report emerging invasive species (Pizarro et al. 2010). The objective of this study was to find the frequency and dominance of three invasive species common to riparian areas of east Alabama: *Ligustrum sinense* (Chinese privet), *Elaeagnus pungens* (silverthorn), and *Triadica sebifera* (Chinese tallow tree). Surveys of these species in riparian forests in and around Auburn, Alabama were conducted to show the relative extent of these shrubs and their relation to urban land use. We expected to see the highest levels of invasive species in the city center with decreasing levels radiating outward into rural areas. The purpose of this research was to evaluate the current distribution of Chinese privet, Chinese tallow tree, and silverthorn along riparian areas and how urban land use may affect the presence-absence and prevalence of these non-native plant species within study sites.

METHODS

Study site selection

This study was conducted in 46 riparian forests adjacent to first and second order streams throughout Auburn, Alabama (Figure 2.1). Auburn has a total land area of 150.4 km^2 with a population of 53,393 (USCB 2010). The climate is humid, with a mean annual precipitation and snowfall of 134.6 cm and 2.5 cm, respectively (NOAA 2015).

To find potential riparian areas, prospective study sites were initially identified using current aerial photographs and stream maps attained from the City of Auburn (2014). To be considered for this study, stream riparian areas had to be forested with a minimum stream length of 200 m. Because this study emphasized urban and suburban land use, emphasis was also placed on sampling sites throughout the city proper and including residential areas representing a wide range of ages. Also, sites tended to be located in common areas or areas that were publically accessible. Each stream was visited prior to field sampling to verify conditions, determine accessibility and confirm overall suitability. Of the approximately 70 streams initially visited, 46 were deemed suitable for sampling.

Riparian surveys

Because of the large number streams to be sampled, a survey method was developed to rapidly assess the prevalence of the target invasive shrubs within each riparian area while also assessing forest conditions in the plot and the surrounding area. The survey developed was based on other plant community rapid assessment surveys (Bellemare et al. 2002, Lundgren et al. 2004, Forsyth et al. 2004) and designed to collect data on invasive species cover, growth stage and density using a modified version of the Braun-Blanquet method (Bellemare et al. 2002) (see Figure 2.1). For each plot, a survey sheet was completed for each of the three target invasive species. As detailed below, data collected included target species total cover by growth stage and density. Supporting data included surrounding land cover (immediate 200 m surrounding the area), forest community form of each plot, cover by strata in each plot (understory herbaceous, shrub/sapling, subcanopy/canopy trees), visual indication of flooding (extremely rare, occasional flooding, regular flooding), and the diameter at breast height (DBH) of the largest Chinese privet in each plot. The cover of each plot was measured as canopy cover for all study sites.

Surveys were conducted between May and September of 2014. For each stream, a transect was extended along the total stream length. Transects were arbitrarily started within the available riparian zones depending on accessibility of the area. For each transect, 10 sampling plots (10 x 10 m quadrats) were established in the riparian zone. Plots were evenly spaced along the total transect. The length of transects ranged between 200 m and 700 m depending on the length of stream available at each study site. The width of the buffer zones was 200 m (100-m extending in both directions perpendicular to the stream). Whenever feasible, plots alternated sides of the stream so that riparian conditions were evenly sampled on both sides. Plots were established so that one side of the quadrat was aligned with the edge of the stream channel. Plot quadrats were set up by using a field tape and the corners of each quadrat were established by using marking flags to create boundaries of a plot. For each target invasive species, a separate survey sheet was filled by walking in the plot and visually estimating the cover, number of stems, and the growth stage of the species (seedlings, saplings, mature and old growth). The criteria for growth stages were determined for seedlings (< 1.5 m tall), saplings (< 2.5 cm DBH), mature (2.5-10 cm DBH), and old growth stage (> 10 cm DBH). It should be noted that no effort was made to discern seedlings and sprouts. Percent cover ranges were recorded as trace (less than 1%), low (1 to 5%), moderate (6 to 25%), high (26 to 50%) and majority (51 to 100%) (Table 2.1). Cover per plot was calculated using range medians (e.g., a moderate cover [6 to 25%] was calculated as 15%) and averaged across all sampled plots for the transect. Plots were evaluated to determine the history of flooding occurrence based on visual evidence (moist soils, sediment deposits, flow paths). Site flooding was estimated as extremely rare, occasionally flooded, and regularly flooded/ saturated. For all options, like community form, a dummy variable was created which represented the presence (1) or absence (0) of the flood condition and the proportion of each category was calculated. The frequency (number of plots per transect) occupied by each target species was also calculated using collected data. Finally, the DBH (1.5 m) size of the largest Chinese privet in each was measured using a tree caliper.

Forest strata in each plot was evaluated by looking at the proportion of cover provided by mature trees (canopy trees >10 cm DBH), immature trees (subcanopy 2.5-10 cm DBH), shrubs/saplings (<2.5 cm DBH), herbaceous/ruderal, and lawn/pasture in the plot. For each category, a dummy variable was recorded which showed if it was present in a plot (1), otherwise it was zero. To establish a visual record, photographs of each plot were taken to document conditions in the field.

For all transects, a digital 200 m buffer zone was created and overlaid on each stream. The length of the buffer zones varied depending on stream lengths. Additional information about adjacent site conditions and land cover (extending into the buffer) were recorded on the sheets. To determine the differences between sapling/shrub, immature forest and mature forest, visual evaluations were conducted by observing canopy cover, DBH of trees (see categories above), and surface cover extending into the buffer. Cover in the 200 m buffer zone was also confirmed by examining aerial photographs of study sites. Approximately 4 hours was required to establish a transect and survey a study site.

City of Auburn spatial analysis

A spatial analysis was conducted to evaluate the distribution of invasive species within the study area. The location of each study site was indicated on digital maps, and the 200 m buffer zone surrounding the transect was delineated (Figure 2.2). The digital maps were created in ESRI ArcGIS version 10.2 based on riparian surveys of this study, and the location of stream transects were attained by using GPS during the surveys.

For illustrating recent study site conditions, a 2011 land use/land cover (LULC) map was used by overlaying first and second order streams with major roads in the Auburn, metropolitan area (Figure 2.3). The GIS data were derived from AlabamaView website in 2014 (alabamaview.org). The digital LULC map indicated the following land cover types: a) urban, b) agriculture and pasture, c) industrial, d) forested and non-forested, e) wetlands, and f) transportation. For each study site, land use type within the 200 m buffer zone was determined to illustrate the most current land use conditions. Also, for each study site buffer area, the number of houses and road length was calculated and normalized per ha. Gravel, paved, and driveways of houses were considered as roads. The most current digital aerial photographs were derived from city of Auburn website in 2014 (webgis.auburnalabama.org), were used to confirm land use and determine if any recent land use changes had occurred within the 200 m buffer zones.

Each of the target species observed in the 46 study sites were mapped on digital distribution maps in GIS. The cover percentages of each species were indicated using dots of different colors. The distribution and cover of each species was evaluated for trends related to proximity to the urban core (more densely populated regions) and the more exurban, peripheral areas of the city.

Data analysis

To evaluate the relationship among variables from the riparian surveys, Pearson correlation and regression analyses were applied to the data using SAS 9.3. Correlations were tested for the three target species separately and used to examine for potential relationships between cumulative total invasive shrub cover and the various shrub growth stage covers. These results were also used to evaluate the tendency for different growth forms to co-occur within riparian areas. If the Pearson correlation coefficient (r) was >0.5 (or <-0.5), then it was considered a strong relationship between two variables.

Linear regression analysis was used to find the potential relationship between urban land use (road density and house density) in the surrounding 200 m buffer and the various measures of target species cover and frequency. Similarly, the amount of forest cover in the 200 m buffer was also used as an independent regression variable to determine potential relationships between forest cover and invasive species. Total species cover and cover by growth stage of each target species were examined as dependent variables for all three species. Regression results were considered significant if the p-value of the parameter was significant at $p<0.10$. A $p<0.05$ was considered highly significant. Also, the R-square (R^2) of regression showed the explanatory power of the independent variables (forest cover, house density, road density) on measures of invasive shrub cover and growth stage.

RESULTS

Riparian conditions and surrounding lands

Riparian sites used for this study were all forested and based on survey data, average forest cover in plots was 39.6±11.1%. Average cover by sapling/shrubs and understory growth was 42.6 ±12.3 and 38.8 ±9.1 respectively. Most riparian sites showed some indication of either occasional or regular flooding (Table 2.3). Within the 200 m buffer zone, land cover contained both mature (24.5±12.3%) and immature (15±7.4%) forests (Table 2.3). Cover by shrubs and sapling was high at 42.6±12.2%. The dominant vegetative cover type within surrounding was shrubs and saplings. The size of the buffer zone sizes ranged between 4.0 and 14.0 ha depending on stream length sampled (Table 2.1).

Invasive species surveys

<u>Chinese privet</u>

Chinese privet was found in all 46 study sites and was the most prevalent of the target invasive species. Chinese privet density was high on most plots. Half of the plots contained 5-20 specimens and 48% had >20 stems per plot. Multi-stems were handled by number of total stems for each species. Chinese privet also contributed substantial cover. On average, total Chinese privet cover was 28.8 ±12.0 %. On most of the sites, privet cover averaged between 20-40%, however, several sites had cover of >80% privet (Figure 2.5). The average cover of the different growth stages were 12.7±5.3% for seedlings, 8.0±2.8% for saplings, and 6.1±3.5% for mature growth, and 1.7±1.5% for old growth forms (Figure 2.6).

Pearson correlation results showed a strong and positive correlation between Chinese privet cover and the mature growth stages of privets (r=0.75), indicating that this form was mostly associated with study sites that had high cover of Chinese privet. There was a positive relationship between privet density and mature cover (r=0.65). Regression analysis was used to assess relationships between surrounding land use and Chinese privet cover (Table 2.7). Only housing density had a statistically significant and positive relationship with total Chinese privet cover (Figure 2.8). Housing density also had a positive and significant relationship with sapling form (Figure 2.11-b) and mature form cover (Figure 2.11-c). There was a significant relationship between road density and percent cover of Chinese privet seedlings (Figure 2.11-a). Cover of old growth privet was positively related to all land use variables (Figure 2.11-d,e,f).

Silverthorn

Silverthorn was the second most prevalent target species in this study (Table 2.4). Like Chinese privet, all 46 study sites contained some silverthorn. Average total cover by silverthorn was 17.5 ±11.9%. Cover by the various growth stages were 6.6±3.3% for seedlings, 4.3±2.5% for saplings, 3.7±2.4% for mature, and 3.0±2.4% for old growth cover (Figure 2.6). The number of silverthorn per plot ranged from 5-20 specimens for most of the study sites (63%). Densities of >20 per plot were detected at 24% of the plots and density of 1-5 specimens was recorded at 13% of the sites (Figure 2.7).

Correlations between total cover and various growth forms of silverthorn were similar to results from Chinese privet. Total cover by silverthorn was positively correlated with mature and old-growth cover with a Pearson correlation coefficients of r=0.55 and r=0.56, respectively. No correlations were detected between % cover of the different growth stages. Regression analyses were conducted to examine silverthorn cover in relation to land use (Table 2.8). Housing density in the 200 m buffer exhibited a positive and statistically significant relationship with total cover and cover by all the various growth stages (Figure 2.9, Table 2.8). House density had positive and significant relationship with seedling, mature and old growth cover of silverthorn (Figure 2.12-a,c,d). There was also a positive relationship between old growth stage and total forest cover (Figure 2.12-e). The old-growth form had the highest R^2 (0.23) among other dependent variables of silverthorn (Table 2.8).

Chinese tallow tree

Chinese tallow tree was the least common target species recorded in this study and on average, Chinese tallow cover averaged 1.6± 2.8% across the study sites. While Chinese tallow generally had less cover throughout Auburn, seedling cover was found at 63% of the study sites (Figure 2.5). Cover by the various growth stages of Chinese tallow was averaged 0.9±0.5% for seedlings, 0.2±0.3% for saplings, 0.1±0.2% for mature, and 0.4±0.4% for old growth specimens (Figure 2.6). Density of Chinese tallow was low across all sites with most sites (89%) containing 1-5 specimens, and 3% of study sites did not have any Chinese tallow tree.

Total percent cover of Chinese tallow tree had a positive correlation with cover of sapling (r=0.67) and mature (r=0.63) stages. No other substantial correlations were detected among growth stages. Based on regression analysis, a significant relationship was detected between housing density and total percent cover of Chinese tallow and percent cover of old growth tallow tree (Table 2.9). Significant positive relationships were detected between cover by old growth Chinese tallow and housing density (Figure 2.13-b). A positive relationship was also detected between sapling cover, old growth cover and road density (Figure 2.13-a,c). There was a positive association between old growth cover and total forest cover (Figure 2.13-d).

City of Auburn spatial analysis

Overall, of the three target species, Chinese privet was the most dominant species with the greatest average percent cover (26-50%) (Figure 2.14). Most incidences of high cover (>26%) were located in the more densely populated city core and surrounding suburban areas. Sites outside the urban core were also populated with Chinese privet but the percent cover in these areas were typically <26%. Two exceptions to this trend were a site outside of Auburn but close to the adjacent City of Opelika, and a second site that was located at the northern extent of the study range.

The highest percent cover of silverthorn was also recorded on average in the 6- 25% range (Figure 2.15). Assessment of silverthorn cover across the Auburn area showed that most incidence of high cover (26-50%) were located in the more densely populated city core. Four exceptions to this trend were two sites with low % cover and located at the northern extent of the study range and two sites located in suburban areas. Sites outside the urban core were also occupied by silverthorn but the percent cover in these areas tend to be <25%.

In contrast, Chinese tallow tree was the least dispersed species (Figure 2.16). Unlike the other species, the occurrence of Chinese tallow tree was more common in sites outside the Auburn urban core. Evaluation of Chinese tallow tree cover across the Auburn area indicated that most incidence of low cover (<1%) were located in the more densely populated city core. Two exceptions to this trend were one site at northwest and a second site at southeast extent of the study range. Sites outside the urban core showed more percent cover (1-5%). Also, three sites located at suburban lands were populated by Chinese tallow tree at high cover range (6-25%).

DISCUSSION

The objective of this study was to determine the association between distribution and dominance of three common invasive species (Chinese privet, silverthorn, and Chinese tallow tree) and urban land use in riparian corridors in and around Auburn in east Alabama. Occurrence of Chinese privet, silverthorn, and Chinese tallow tree in riparian areas was associated with certain measures of urban land use. There was a positive relationship between housing density (#/ha) and total percent cover of Chinese privet (Table 2.7, Figure 2.8). Silverthorn abundance was also positively related to housing density, and silverthorn seedling cover was also positively related to road density (Table 2.8). The least prevalent species was Chinese tallow tree, but it was also positively related with housing density (Table 2.9). These results are consistent with others that have detected a positive relationship between urban land use and distribution of invasive species (Borgmann and Rodewald 2005; Parker et al. 2009; Pizarro et al. 2010; Terereai et al. 2015). Pizarro et al. (2010) applied similar independent variables by using regression analysis on correlated variables (road density, house density, and invasive species richness) to test the relationship between invasive species richness and housing growth in New England, USA. As a result of their analyses, they found that housing variables were significantly associated with abundance and spread of invasive species.

Housing density can increase the occurrence of fragmentation in urban areas, and this may allow increased spread of invasive plant species which are adaptive to forest edge, disturbance, and propagule pressure (Hobbs 2001). Mehrhoff et al. (2003) reported that invasive shrubs have been planted for horticultural and ornamental aspects around houses which facilitates the spread of invasive plants and increases spread of invasive plant species around urban landscapes. Also, disturbances associated with housing construction (soil exposure, edges, trails, among other factors) can generate microhabitats which favor establishment of invasive plants in urban areas (Wania et al. 2006). In the southeastern U.S., Chinese privet can thrive in multiple soil types along the stream banks, and residents are still using privet in home landscapes which results with escaping to roadsides, invading forest edges and riparian areas (Zhao et al. 2013).

The relationship among the growth rates of Chinese privet can be used to detect patterns of colonization. For instance, the growth stage contributing most to privet cover was old growth (Figure 2.11-d). Housing density had a positive relationship with all stages of Chinese privet except seedlings (Table 2.7, Figure 2.11-b,c), but the relationship wasn't particularly strong. Only total forest cover was correlated with the old growth stage of Chinese privet (2.11-f). Road density only had a positive relationship with seedling and old growth stages of privet (Figure 2.11-a,e). Road density and housing density did not exhibit identical patterns for privet. This result may have occurred because houses may have been established before road networks developed around farmlands and rural areas. Also, many riparian corridors may have occurred on lands that were distant to major roads.

The most prevalent form of silverthorn was seedlings. Housing density was positively correlated with seedling and old growth of silverthorn (Figure 2.12-a,d). However, road density was found positively related with only seedling form (Figure 2.12-b). Like Chinese privet, only total forest cover was correlated with old growth stage of silverthorn (Figure 2.12-e). Based on density, the most commonly surveyed form of Chinese tallow tree was seedlings. Road density was positively related with sapling and old growth stages, however the relationship was fairly weak (R^2=0.08 and 0.09, Figure 2.13-a,c). Housing density was positively correlated only with old growth stage of Chinese tallow tree (Figure 2.13-b) however again there was a weak relationship between old growth stage and total forest cover (Figure 2.13-d). This relationship may be found because Chinese tallow tree density was low in total surveyed riparian transects and old growth stages were particularly observed with high forest cover. Pysek and Pysek (1995) observed the habitat preference and spread of the invasive plant, *Heracleum mantegazzianum* (giant hogweed) related to roads in the Czech Republic. For their study, 14 habitat types were surveyed and majority of vegetation types were evaluated. It was concluded that invasive plants were most abundant adjacent to roads and railways due to rapid seed dispersal around transportation ways. Parendes and Jones (2000) investigated the relationship between spread mechanism of invasive species and road and stream segments in Oregon, U.S. They found that roads and fragmented forests had a positive relationship with spread of invasive plants along the riparian streams. In our study, road density was just slightly correlated with total percent cover and growth stages of all target invasive species along the riparian areas. This result might be because some of the suburban house lands in the 200 m zone were divided into small parecls with fewer roads connected to them compared to other urban areas. Some of the roads might have been less detectable after agricultural lands were abandoned and became reforested.

Considering the density of urban and suburban lands represented in Table 2.1, streams in 200 m buffers with high housing density were most correlated with abundance of target invasive species. Similarly, Borgmann and Rodewald (2005) researched the relationship between urban land use and exotic shrub species in Ohio, U.S.A. They specifically evaluated the correlation of percent cover rates of residential and commercial land use with surveyed invasive shrubs. They found that percent cover of invasive species was significantly associated with urban riparian areas. They also found that forested landscapes with low canopy cover were more infested than landscapes with low housing density. In this study, analysis showed that there was a positive correlation with housing density and invasive plant species cover and housing density could predict the density and spread of target invasive plants in the surveyed riparian streams of Auburn. Also, results of this study are consistent with Greene and Blossey (2014) reported that urban development and distance to developed land could predict the percent cover of Chinese privet in South Carolina.

Urbanization could be a key factor that is associated with the increasing prevalence of invasive species cover in riparian areas. In coastal regions of southeastern United States, population growth leads to increased residential development and forest loss (Barksdale and Anderson 2014). Similarly, urban land use leads to greater impervious surface area around riparian areas which can result in low infiltration rates, high surface run off, and increased erosion (Paul and Mayer 2001; Chadwick et al. 2006). The extent of these environmental alterations and increased disturbance associated with urbanization may help to determine the spread of invasive shrubs in urban landscapes. Determining the relationship between the abundance of invasive plants and urban land use may provide useful information to managers and private land owners seeking to effectively manage invasive shrubs.

Our results show some differences regarding urban land use variables and their relation to distribution of invasive plants in riparian areas. Although housing density had the highest explanatory power related to percent cover of target species in this study, there are also other factors that explain invasive plant species. Some other factors that we did not measure such as soil disturbance and microbial communities, light exposure, non-native predators, and native plant richness could be also associated with urbanization and its relation to invasive plants. Sung et al. (2011) examined the relationship between watershed urbanization and its relation with woody invasive plants in Austin, TX. They measured multiple variables associated with environmental conditions of floodplains such as impervious surface adjacent to watersheds, flooding, soil disturbance, and period of dry season. They concluded that urban land use promotes invasion by invasive plants by leading hydrologic drought in riparian areas.

The Auburn spatial analysis results of this study showed that LULC change in total surveyed riparian streams was related to abundance and spread of target invasive shrubs. For instance, the density of Chinese privet showed some evidence of a positive trend with residential and suburban land use (Figure 2.14). Likewise, the abundance of silverthorn exhibited some trends with suburban and urban lands in total surveyed riparian sites (Figure 2.15). Both Chinese privet and silverthorn were significantly abundant within landscapes closer to the city center. Further out from the city center and suburban lands, cover of both Chinese privet and silverthorn tended to decrease. In contrast, Chinese tallow tree density percent cover showed an opposite trend with landscapes close to city center often having slightly less cover (Figure 2.16). This result may be because tallow tree has been introduced more recently than other target species within study sites. Also, it is worth noting that the regression results for tallow were likely driven by two sites of high cover in the high housing density. This result may suggest that new developments may have been established close to riparian areas already with high cover of Chinese tallow tree. Without these two sites, there would probably not be a relationship between housing density and Chinese tallow tree cover. By looking at LULC maps, riparian areas close to urban developments were more susceptible to invasion by the target invasive species. Parker et al. (2009) observed the correlation between native and exotic species in Edgewater, MD. They analyzed the species richness and patterns of diversity in deforested lands. In the same study, it was concluded that young forests had 41% more exotic species than older forest lands, and there was a positive relationship between the abundance of native and exotic invasive species richness in the Chesapeake Bay area. As a result, it is assumed that urban and agriculture land use was highly associated with richness and prevalence of invasive shrubs within riparian areas of this study.

According to Siebenthaler (2014), until the 20^{th} century, agricultural products such as corn, cotton, and cattle were the primary economic resources for Auburn, yet after 1940s, farm owners started to abandon agricultural lands due to industrial development in the city. Pandi et al. (2014) assessed the persistence and prevalence of invasive plants by surveying 190 farmsteads in central Hungary. In the same research, it was concluded that rural depopulation and abandoning farm lands can facilitate invasive species to remain for decades within agricultural landscapes. Mattingly and Orrock (2013) examined the influence of historical land use on distribution of invasive Lespedeza plants in North Carolina, USA. They found that soil disturbance and increase in historical land use change were highly associated with invasive species introduction and expansion in landscapes throughout North Carolina. Similarly, Auburn is surrounded by abandoned agricultural lands which were possibly occupied by invasive plant species in the past and may have been areas of introduction and spread of target species surveyed in this study.

This study shows that urban land use and specifically housing density may be an important association with invasive shrubs species in east Alabama. One improvement to this study could have been to use more riparian streams for the surveys. The number of riparian sites were limited by accessibility and landowner permission. Thus, this study may be a benchmark for future studies where additional data might be collected. If more riparian surveys are implemented, it can possibly increase the explanatory power of the survey.

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Table 2.1: Housing and road density in the 200 m buffer zone for each study site.

site#	SITES	Buffer Size (ha)	Road Length (m)	Road Density (m/ha)	# of houses	Housing Density (#/ha)
1	East Samford Avenue	10.6	456.0	43.0	1	0.1
2	Champions Blvd.	6.9	326.2	47.5	1	0.1
3	Donahue Dr.	4.0	110.0	30.5	10	2.8
4	McMillan St.	7.8	580.0	74.3	9	1.2
5	McMillan St. 2	14.0	241.6	17.3	12	0.9
6	Kiesel Park	4.1	469.4	121.2	0	0.0
7	Waterstone Cir.	7.0	527.8	75.6	57	8.2
8	Heywood St.	12.1	456.6	37.9	51	4.2
9	Sam Harris park	8.8	133.0	15.2	12	1.4
10	Sanders Creek	4.2	293.8	69.9	18	4.3
11	Drake Middle school	10.3	771.1	75.1	56	5.5
12	Drake2 Westview cemetery	7.9	265.7	33.8	15	1.9
13	Church	8.6	417.8	48.5	58	6.7
14	Town Creek Park	9.6	215.2	22.5	29	3.0
15	Cary Woods Creek	8.2	296.4	36.0	46	5.6
16	Camden Ridge Creek	10.1	610.9	60.3	117	11.6
17	Tuscany Village	6.0	186.2	30.9	24	4.0
18	Preserve Drive	9.2	322.6	35.0	20	2.2
19	Ellington Way	7.9	618.3	78.4	57	7.2
20	Watercrest Boulevard	11.2	275.1	24.7	18	1.6
21	Heath Road	10.2	133.7	13.1	1	0.1
22	Academy Drive	9.2	187.1	20.4	10	1.1
23	Mall Parkway	8.0	334.5	41.8	40	5.0
24	Northridge Street	11.6	232.8	20.0	28	2.4
25	Chewacla Dr.	9.4	365.4	38.8	82	8.7
26	E Thach Ave.	10.0	290.9	29.0	53	5.3
27	E Samford Ave.	7.0	222.2	31.8	27	3.9

Table 2.1: Continue

28	Lee Road 72	7.8	238.0	30.4	8	1.0
29	Lee Road 72-2	5.8	48.5	8.4	4	0.7
30	East Farmville Road	7.2	178.6	24.7	3	0.4
31	East Farmville Road-2	12.2	235.8	19.3	9	0.7
32	Mimms Ln.	5.9	436.3	74.5	9	1.5
33	Beechbrook Dr.	6.9	128.8	18.6	10	1.4
34	Wooden Bridge	10.1	104.8	10.4	12	1.2
35	Aberdeen Ln.	8.0	120.9	15.2	26	3.3
36	Ogletree Rd.	8.9	103.9	11.7	10	1.1
37	Chewacla State Park 1	10.2	102.6	10.1	8	0.8
38	Chewacla State Park 2	7.0	85.2	12.2	1	0.1
39	East University Drive	8.2	204.1	24.8	11	1.3
40	Stoneridge Drive	6.9	309.3	44.5	18	2.6
41	Duck Samford Park	5.5	227.1	41.1	5	0.9
42	Reynolds Drive	4.7	107.7	22.8	9	1.9
43	Fisheries 1	9.3	118.1	12.8	0	0.0
44	Fisheries 2	7.8	130.3	16.8	2	0.3
45	Fisheries 3	7.0	0.0	0.0	0	0.0
46	Town Creek 2	8.0	138.1	17.3	6	0.8
	Mean ±SE	8.3±2.2	268.7±170	34.5±24.4	21.8±24.7	2.6±2.5

Table 2.2: Land cover in the 200 m buffer zone of each study site.

		Land Cover (%)				
Site #	**Site**	**Mature Forest**	**Immature Forest**	**Shrubs /Saplings**	**Herbaceous/ Ruderal**	**Lawn/ Pasture**
1	East Samford Avenue	9.0	11.5	61.0	18.5	0.0
2	Champions Blvd.	50.0	0.0	31.5	18.5	0.0
3	Donahue Dr.	15.5	27.5	26.5	30.5	0.0
4	McMillan St.	27.0	11.0	47.0	9.0	6.0
5	McMillan St. 2	8.0	26.0	43.0	22.0	1.0
6	Kiesel Park	7.0	30.0	38.0	24.5	0.5
7	Waterstone Cir.	1.0	14.5	69.5	15.0	0.0
8	Heywood St.	2.0	21.5	67.5	9.0	0.0
9	Sam Harris park	18.0	9.0	47.5	25.5	0.0
10	Sanders Creek	19.0	19.0	41.0	20.0	1.0
11	Drake middle school	12.0	17.0	47.0	24.0	0.0
12	Drake2West-view cemetry	11.5	11.5	59.0	18.0	0.0
13	Church	13.5	13.5	46.5	26.5	0.0
14	Town Creek Park	13.5	33.0	30.5	23.0	0.0
15	Cary Woods Creek	11.0	12.0	62.5	13.5	1.0
16	Camden Ridge Creek	14.0	12.0	66.0	5.5	2.5
17	Tuscany Village	25.5	24.0	41.5	9.0	0.0
18	Preserve Drive	35.0	14.5	35.5	14.0	1.0
19	Ellington Way	19.0	10.0	44.0	27.0	0.0
20	Watercrest Boulevard	21.0	6.0	47.0	26.0	0.0
21	Heath Road	38.0	9.5	30.5	22.0	0.0
22	Academy Drive	36.5	4.0	38.5	21.0	0.0
23	Mall Parkway	37.0	8.5	40.0	14.5	0.0
24	Northridge Street	31.0	10.0	43.5	15.5	0.0
25	Chewacla Dr.	29.5	19.0	25.5	26.0	0.0

Table 2.2: Continue

26	E Thach Ave.	21.0	13.5	44.0	21.5	0.0
27	E Samford Ave.	28.0	11.0	40.0	21.0	0.0
28	Lee Road 72	28.0	8.0	29.0	35.0	0.0
29	Lee Road 72-2	27.0	9.0	43.0	14.0	6.0
30	EastFarmville Road	55.5	13.5	20.5	10.5	0.0
31	EastFarmville Road-2	47.0	18.5	21.0	13.5	0.0
32	Mimms Ln.	25.5	21.5	31.5	21.5	0.0
33	Beechbrook Dr.	27.0	14.0	36.0	23.0	0.0
34	Wooden Bridge	29.0	16.5	32.5	22.0	0.0
35	Aberdeen Ln.	24.0	10.5	42.5	23.0	0.0
36	Ogletree Rd.	15.0	24.0	30.0	31.0	0.0
37	Chewacla State Park 1	27.5	13.0	28.0	31.5	0.0
38	Chewacla State Park 2	24.5	16.5	32.0	27.0	0.0
39	EastUniversity Drive	20.0	22.0	29.0	27.0	2.0
40	Stoneridge Drive	34.0	5.0	42.0	19.0	0.0
41	Duck Samford Park	43.0	10.5	32.0	13.5	1.0
42	Reynolds Drive	33.0	11.0	45.0	11.0	0.0
43	Fisheries 1	42.5	13.0	27.0	17.5	0.0
44	Fisheries 2	27.0	23.5	29.0	20.5	0.0
45	Fisheries 3	29.0	11.0	28.5	31.5	0.0
46	Town Creek 2	16.0	31.0	43.5	9.5	0.0
Mean ±SE		24.3±12.3	15.0±7.3	39.9±12.07	20±7.05	0.5±1.3

Table 2.3: Proportion of study site plots with indication of flood tendency and mean vegetation strata cover for each study site.

		Flood Indicators			Vegetation cover (%)		
#	**Sites**	**Extremely rare**	**Occasion al**	**Regularly**	**Understory**	**Shrub**	**Canopy**
1	East Samford Avenue	0.0	0.4	0.6	19.6	12.5	14.8
2	Champions Blvrd.	0.3	0.2	0.5	26.8	17.1	36.4
3	Donahue Dr.	0.0	0.3	0.7	33.9	33.6	40.8
4	McMillan St.	0.1	0.5	0.4	37.4	43.8	42.4
5	McMillan St. 2	0.0	0.4	0.6	38.5	42.2	45.9
6	Kiesel Park	0.2	0.2	0.6	27.6	32.5	29.1
7	Waterstone Cir.	0.0	0.1	0.9	39.9	37.6	32.8
8	Heywood St.	0.0	0.3	0.7	29.0	27.5	26.7
9	Sam Harris park	0.0	0.6	0.4	38.5	29.0	34.1
10	Sanders Creek	0.0	0.3	0.7	31.1	39.9	54.4
11	Drake middle school	0.0	0.2	0.8	40.8	50.7	57.9
12	Drake2Westview cemetry	0.0	0.1	0.9	36.2	63.9	31.4
13	Church	0.1	0.3	0.6	29.0	37.3	50.5
14	Town Creek Park	0.0	0.2	0.8	42.2	38.7	51.9
15	Cary Woods Creek	0.0	0.2	0.8	44.5	54.2	49.6
16	Camden Ridge Creek	0.0	0.3	0.7	44.5	56.5	38.8
17	Tuscany Village	0.0	0.2	0.8	23.1	39.7	39.9
18	Preserve Drive	0.0	0.1	0.9	38.5	35.0	57.0
19	Ellington Way	0.0	0.2	0.8	34.8	43.6	49.6
20	Watercrest Boulevard	0.0	0.2	0.8	53.0	54.2	31.4
21	Heath Road	0.0	0.2	0.8	48.2	44.5	53.3
22	Academy Drive	0.0	0.2	0.8	45.6	35.0	48.2
23	Mall Parkway	0.0	0.3	0.7	47.0	50.7	46.8
24	Northridge Street	0.0	0.2	0.8	42.0	49.4	50.7
25	Chewacla Dr.	0.0	0.3	0.7	56.5	56.5	39.9

Table 2.3: Continue

26	E Thach Ave.	0.0	0.3	0.7	56.5	67.6	29.0
27	E Samford Ave.	0.2	0.3	0.5	45.9	48.4	42.4
28	Lee Road 72	0.0	0.3	0.7	39.6	48.2	44.7
29	Lee Road 72-2	0.2	0.5	0.3	31.4	36.5	36.4
30	EastFarmville Road	0.0	0.2	0.8	31.3	30.0	54.2
31	EastFarmville Road-2	0.0	0.2	0.8	24.2	20.6	69.0
32	Mimms Ln.	0.0	0.2	0.8	43.1	60.2	57.9
33	Beechbrook Dr.	0.0	0.3	0.7	46.8	60.2	50.5
34	Wooden Bridge	0.0	0.3	0.7	42.2	44.7	42.2
35	Aberdeen Ln.	0.0	0.4	0.6	36.2	44.5	45.9
36	Ogletree Rd.	0.0	0.1	0.9	50.5	60.2	55.6
37	Chewacla State Park 1	0.0	0.3	0.7	43.1	39.9	43.1
38	Chewacla State Park 2	0.0	0.2	0.8	38.5	42.2	43.1
39	EastUniversity Drive	0.0	0.4	0.6	29.3	40.8	49.1
40	Stoneridge Drive	0.0	0.1	0.9	32.7	40.8	57.9
41	Duck Samford Park	0.0	0.1	0.9	42.2	32.7	61.6
42	Reynolds Drive	0.0	0.2	0.8	47.0	54.2	46.8
43	Fisheries 1	0.0	0.3	0.7	43.4	33.6	42.2
44	Fisheries 2	0.0	0.1	0.9	23.0	27.6	42.2
45	Fisheries 3	0.0	0.5	0.5	36.0	44.7	51.9
46	Town Creek 2	0.0	0.3	0.7	56.5	57.9	55.6
	Mean±SE	0.0±0.1	0.3±0.1	0.7±0.2	38.8±9.1	42.6±12.2	45.1±10.5

Table 2.4: Mean total cumulative cover and cover of various growth stages for Chinese privet at each study site.

			Cover (%) of Growth Stages			
#	**Sites**	**Total Cover (%)**	**Seedling**	**Sapling**	**Mature**	**Old-growth**
1	East Samford Ave.	13.4	11.5	1.3	0.5	0.1
2	Champ. Blvrd	21.7	15.5	5.0	1.2	0.0
3	south Donahue	33.7	8.9	8.2	16.2	0.3
4	McMillan	29.1	10.0	9.0	6.7	3.3
5	McMillan 2	35.0	3.2	13.5	15.2	2.8
6	Kiesel Park	11.0	3.3	4.5	1.1	1.0
7	Waterst. Cr.	24.2	12.4	7.4	3.1	1.2
8	Heywood	18.3	10.0	5.9	2.3	0.2
9	Sam Harris Park	30.2	6.2	12.5	10.7	0.6
10	Sanders Creek	21.7	13.9	4.9	2.7	0.2
11	Drake Middle Sch.	44.7	18.1	9.4	15.2	2.0
12	Drake 2	33.4	18.4	7.4	7.4	0.3
13	Church	37.3	13.6	13.8	7.8	2.1
14	Town Creek Park	47.0	8.9	7.0	24.9	6.1
15	Cary Woods	25.3	11.2	8.5	4.8	0.8
16	Camden Ridge	36.2	18.5	10.1	6.2	1.5
17	Tuscany Village	15.2	9.6	2.9	1.1	0.2
18	Preserve Dr.	23.3	10.8	9.7	2.6	0.2
19	Ellington Way	27.9	11.0	9.2	7.1	0.6
20	Watercrest Dr.	36.1	12.6	11.2	8.5	0.2
21	Health Road	43.3	18.2	12.3	11.0	1.7
22	Academy Dr.	34.7	9.4	13.9	6.9	1.0
23	Mall Parkway	33.3	14.6	5.7	9.3	0.3
24	Northridge	43.0	20.2	10.5	11.4	0.9
25	Chewacla Dr.	11.5	5.2	3.7	1.4	0.0
26	East Thach Avenue	48.2	12.8	12.5	15.9	7.0
27	East Samfard	25.9	10.7	6.2	5.6	0.8
28	Lee Road 72	31.1	16.0	6.7	5.3	0.0
29	Lee Road	21.8	9.3	6.7	3.4	2.5
30	East Farm. Rd.	12.2	6.9	1.2	0.5	0.0
31	East Farmville	11.0	6.1	0.6	1.0	0.0
32	Mimms Ln.	34.7	15.3	8.5	6.8	0.7
33	Beech Brook Dr.	38.4	17.1	11.5	5.8	1.3
34	Wooden bridge	27.5	12.4	6.7	5.6	0.0
35	Aberdeen Ln.	27.4	15.6	2.7	5.1	1.2
36	Ogletree Road	28.7	8.9	8.5	7.2	1.3
37	Chewacla 1	17.9	6.3	5.6	4.3	0.0
38	Chewacla 2	19.2	8.6	6.7	3.8	0.0
39	East Univ. Dr.	46.9	13.8	11.7	15.7	0.9
40	Stone Ridge	32.0	10.9	5.3	7.5	5.1
41	Duck Samfard Park	17.6	7.7	3.9	2.4	0.2
42	Reynolds Dr.	34.8	12.3	12.7	9.0	0.7
43	Fisheries 1	21.4	8.1	7.8	3.3	0.0
44	Fisheries 2	6.6	3.2	1.5	0.6	0.0
45	Fisheries 3	8.1	2.8	2.0	1.7	0.0
46	Town Creek 2	60.2	11.7	7.8	29.2	11.4
	Mean ±SE	28.3±11.9	12.0±4.4	7.5±3.7	6.1±6.2	1.0±2.2

Table 2.5: Mean total cumulative cover and cover of various growth stages for silverthorn at each study site.

			Cover (%) of Growth Stages			
#	CREEKS	Total Cover (%)	Seedling	Sapling	Mature	Old-growth
1	East Samford Ave.	0.4	0.3	0.0	0.0	0.0
2	Champ. Blvd	2.4	0.5	0.9	0.0	0.0
3	south Donahue	9.8	2.0	4.1	3.0	0.8
4	McMillan	6.1	3.0	1.2	0.2	1.6
5	McMillan 2	0.9	0.2	0.1	0.1	0.0
6	Kiesel Park	9.2	2.3	3.2	1.8	0.9
7	Waterst. Cr.	9.9	3.1	1.3	1.2	0.3
8	Heywood	3.6	0.7	1.3	0.1	0.0
9	Sam Harris Park	14.6	3.8	8.1	2.7	0.0
10	Sanders Creek	47.0	17.4	0.0	22.5	7.0
11	Drake Middle Sch.	29.7	12.8	4.2	8.2	1.6
12	Drake 2	35.0	11.5	11.9	5.4	6.1
13	Church	26.5	8.2	10.6	3.7	4.0
14	Town Creek Park	12.1	5.5	4.5	1.1	1.0
15	Cary Woods	5.5	2.1	1.0	0.8	0.0
16	Camden Ridge	15.2	8.0	3.3	0.0	0.8
17	Tuscany Village	16.9	10.1	3.6	1.5	1.7
18	Preserve Dr.	21.2	12.7	2.6	2.2	1.5
19	Ellington Way	11.6	3.5	3.5	1.0	1.2
20	Watercrest Dr.	13.6	6.3	1.2	2.6	2.2
21	Health Road	20.7	5.9	3.4	1.9	3.3
22	Academy Dr.	11.1	3.0	1.9	0.9	0.9
23	Mall Parkway	18.6	5.2	2.9	3.4	1.5
24	Northridge	13.6	3.2	1.8	1.4	3.1
25	Chewacla Dr.	45.9	9.6	6.7	8.3	21.3
26	East Thach Avenue	9.1	1.2	2.6	2.2	1.3
27	East Samford	19.9	3.5	5.0	2.1	5.4
28	Lee Road 72	26.0	3.6	4.8	9.3	3.0
29	Lee Road	6.1	3.5	0.9	0.2	1.6
30	East Farm. Rd.	2.7	1.4	0.5	0.1	0.1
31	East Farmville	3.7	0.7	1.2	0.7	0.0
32	Mimms Ln.	28.7	6.4	5.2	10.0	4.2
33	Beech Brook Dr.	7.6	2.0	0.3	1.4	0.9
34	Wooden bridge	23.4	8.0	2.8	3.9	4.1
35	Aberdeen Ln.	22.6	6.8	4.0	5.3	4.3
36	Ogletree Road	19.8	4.9	2.2	2.2	6.5
37	Chewacla 1	20.5	3.3	4.3	3.9	7.0
38	Chewacla 2	22.8	1.1	2.7	7.8	8.9
39	East Univ. Dr.	20.8	6.2	4.8	1.3	2.2
40	Stone Ridge	17.0	0.7	2.4	5.3	1.6
41	Duck Samfard Park	50.4	7.8	7.8	15.1	14.6
42	Reynolds Dr.	15.2	2.7	2.5	6.0	0.9
43	Fisheries 1	30.5	6.5	3.8	5.3	5.6
44	Fisheries 2	16.6	4.7	1.7	4.2	4.3
45	Fisheries 3	32.0	8.5	2.7	9.1	8.5
46	Town Creek 2	9.4	5.3	2.5	1.6	0.0
	Mean ±SE	17.5±11.9	5.5±3.8	3.5±2.6	3.0±4.4	2.4±4.1

Table 2.6: Mean total cumulative cover and cover of various growth stages for Chinese tallow tree at each study site.

			Cover (%) of Growth Stages			
#	**CREEKS**	**Total Cover (%)**	**Seedling**	**Sapling**	**Mature**	**Old-growth**
1	East Samford Ave.	0.9	0.5	0.1	0.0	0.0
2	Champ. Blvrd	0.9	0.2	0.2	0.2	0.0
3	south Donahue	0.0	0.0	0.0	0.0	0.0
4	McMillan	0.2	0.0	0.0	0.0	0.0
5	McMillan 2	15.5	4.2	4.8	3.3	3.1
6	Kiesel Park	2.7	0.8	0.4	0.4	0.5
7	Waterst. Cr.	2.3	0.5	0.0	0.0	0.7
8	Heywood	2.2	1.1	0.0	0.2	0.0
9	Sam Harris Park	1.6	1.1	0.2	0.0	0.0
10	Sanders Creek	0.2	0.1	0.0	0.0	0.0
11	Drake Middle Sch.	0.2	0.1	0.0	0.0	0.0
12	Drake 2	0.6	0.2	0.0	0.0	0.1
13	Church	0.9	0.4	0.1	0.1	0.0
14	Town Creek Park	0.0	0.0	0.0	0.0	0.0
15	Cary Woods	0.1	0.0	0.0	0.0	0.0
16	Camden Ridge	0.4	0.0	0.0	0.0	0.1
17	Tuscany Village	0.1	0.0	0.0	0.0	0.0
18	Preserve Dr.	8.3	2.8	2.2	0.6	1.8
19	Ellington Way	0.7	0.2	0.0	0.0	0.1
20	Watercrest Dr.	0.3	0.0	0.0	0.0	0.0
21	Health Road	0.6	0.1	0.1	0.0	0.1
22	Academy Dr.	0.5	0.1	0.0	0.0	0.1
23	Mall Parkway	0.5	0.0	0.0	0.0	0.1
24	Northridge	3.4	0.6	0.1	0.4	0.9
25	Chewacla Dr.	0.3	0.1	0.0	0.0	0.0
26	East Thach Avenue	0.2	0.1	0.0	0.0	0.0
27	East Samfard	1.7	0.5	0.0	0.1	0.2
28	Lee Road 72	1.7	0.5	0.1	0.0	0.2
29	Lee Road	0.2	0.0	0.0	0.0	0.0
30	East Farm. Rd.	0.1	0.0	0.0	0.0	0.0
31	East Farmville	0.1	0.0	0.0	0.0	0.0
32	Mimms Ln.	1.7	0.3	0.0	0.0	0.4
33	Beech Brook Dr.	3.4	0.9	0.0	0.3	0.9
34	Wooden bridge	2.0	0.7	0.1	0.0	0.4
35	Aberdeen Ln.	2.0	0.5	0.0	0.0	0.6
36	Ogletree Road	8.4	2.4	0.3	0.8	2.5
37	Chewacla 1	0.1	0.0	0.0	0.0	0.0
38	Chewacla 2	0.1	0.0	0.0	0.0	0.0
39	East Univ. Dr.	3.2	1.0	0.2	0.2	0.5
40	Stone Ridge	2.0	0.4	0.0	0.2	0.6
41	Duck Samfard Park	4.6	1.3	0.6	0.3	0.1
42	Reynolds Dr.	0.7	0.1	0.0	0.1	0.2
43	Fisheries 1	0.1	0.0	0.0	0.0	0.0
44	Fisheries 2	0.1	0.0	0.0	0.0	0.0
45	Fisheries 3	0.1	0.0	0.0	0.0	0.0
46	Town Creek 2	0.2	0.0	0.0	0.0	0.0
	Mean ±SE	1.6±2.8	0.5±0.8	0.2±0.8	0.2±0.5	0.3±0.6

Table 2.7: Results of regression analyses for relationship between Chinese privet cover variables and land use in the 200 m buffer.

Total cover % (n=46)					
Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr > |t|
Housing Density (#/ha)	0.285	1.41**	0.57	2.473	0.001
Road Density(m/ha)	0.042	0.002	0.074	0.027	0.684
Total forest cover (%)	0.032	0.095	0.168	0.570	0.278
Seedling cover % (n=46)					
Housing Density (#/ha)	0.101	0.581	0.692	0.839	0.113
Road Density(m/ha)	0.087	0.049*	0.042	1.167	0.067
Total forest cover (%)	0.061	0.047	0.056	0.055	0.470
Sapling cover % (n=46)					
Housing Density (#/ha)	0.143	0.602**	0.210	2.867	0.019
Road Density(m/ha)	0.032	0.004	0.022	0.182	0.536
Total forest cover (%)	0.086	0.051	0.071	0.718	0.195
Mature growth cover % (n=46)					
Housing Density (#/ha)	0.127	0.835**	0.583	1.432	0.009
Road Density(m/ha)	0.060	0.009	0.027	0.343	0.330
Total forest cover (%)	0.078	0.049	0.057	0.850	0.191
Old-Growth cover % (n=46)					
Housing Density (#/ha)	0.221	0.385**	0.179	2.151	0.014
Road Density(m/ha)	0.094	0.027*	0.019	1.401	0.086
Total forest cover (%)	0.102	0.062*	0.041	1.512	0.058

**significant at $p<0.05$ *significant at $p<0.10$

Table 2.8: Results of regression analyses for relationships between silverthorn cover variables and land use in the 200 m buffer.

Total cover % (n=46)					
Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr > |t|
Housing Density (#/ha)	0.189	1.867**	0.840	2.223	0.004
Road Density(m/ha)	0.062	-0.008	0.147	-0.054	0.605
Total forest cover (%)	0.058	0.180	0.225	0.800	0.433
Seedling cover % (n=46)					
Housing Density (#/ha)	0.138	0.526*	0.388	1.356	0.051
Road Density(m/ha)	0.089	0.046*	0.043	1.070	0.064
Total forest cover (%)	0.074	-0.019	0.053	-0.358	0.725
Sapling cover % (n=46)					
Housing Density (#/ha)	0.930	0.108	0.841	0.128	0.140
Road Density(m/ha)	0.032	0.024	0.047	0.511	0.396
Total forest cover (%)	0.031	0.041	0.059	0.695	0.408
Mature growth cover % (n=46)					
Housing Density (#/ha)	0.101	0.564*	0.441	1.279	0.077
Road Density(m/ha)	0.054	0.008	0.032	0.250	0.797
Total forest cover (%)	0.037	0.034	0.049	0.694	0.495
Old-Growth cover % (n=46)					
Housing Density (#/ha)	0.231	0.748**	0.456	1.640	0.034
Road Density(m/ha)	0.086	0.061	0.134	0.455	0.248
Total forest cover (%)	0.110	0.035*	0.033	1.061	0.083

**significant at $p<0.05$ *significant at $p<0.10$

Table 2.9: Results of regression analyses for relationship between Chinese tallow tree cover variables and land use in the 200 m buffer.

Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr > |t|
Total cover % (n=46)					
Housing Density (#/ha)	0.134	0.387**	0.193	2.005	0.049
Road Density(m/ha)	0.035	0.027	0.096	0.281	0.451
Total forest cover (%)	0.051	0.096	0.103	0.932	0.276
Seedling cover % (n=46)					
Housing Density (#/ha)	0.061	-0.008	0.021	-0.381	0.260
Road Density(m/ha)	0.077	0.003	0.015	0.220	0.844
Total forest cover (%)	0.038	0.011	0.024	0.458	0.654
Sapling cover % (n=46)					
Housing Density (#/ha)	0.119	0.004	0.011	0.364	0.314
Road Density(m/ha)	0.083	0.009*	0.010	0.929	0.078
Total forest cover (%)	0.031	0.015	0.062	0.242	0.563
Mature growth cover % (n=46)					
Housing Density (#/ha)	0.127	0.019	0.028	0.679	0.210
Road Density(m/ha)	0.041	0.001	0.003	0.333	0.419
Total forest cover (%)	0.062	0.017	0.025	0.680	0.184
Old-Growth cover % (n=46)					
Housing Density (#/ha)	0.172	0.096*	0.080	1.203	0.061
Road Density(m/ha)	0.094	0.019*	0.021	0.905	0.097
Total forest cover (%)	0.105	0.047*	0.044	1.068	0.065

**significant at p<0.05 *significant at p<0.10

SURVEYING THE EAST ALABAMA-DISPERSAL OF INVASIVE SHRUBS **Data Sheet**	
Site	**Date**
Plot	**Time**
Target Species	**Investigator**
I)Target Species Plot Information	**II)Surrounding Land Information**
Q1-Cover of Plot (%)	*Q4-Immediate Area LULC (Surrounding 200 m)*
a) Trace (less than 1%) b) Low (1 to 5%) c) Moderate (6 to 25%) d) High (26 to 50 %) e) Majority (51 to 100%)	a) Mature Forest ______ % of total b) Shrubs/Sapling ______ % of total c) Herbaceous/Ruderal ______ % of total d) Lawn/Pasture/Cropland ______ % of total e) Immature Forest ______ % of total
Q2-Growth Stages	**III) Community Plot Information**
a) Sprout and Seedling (<1.5 m. tall) ______ % of total b) Sapling (<2.5cm dbh) ______ % of total c) Mature (2.5-10cm dbh) ______ % of total d) Old-growth (>10cm dbh) ______ % of total	*Q5-Community Form of Plot*
Q3-Target Species Density (#/Plot)	a) Mature Forest b) Shrubs/Sapling c) Herbaceous/Ruderal d) Lawn/Pasture/Cropland e) Immature Forest
a) 0 b) 1-5 c) 5-20 d) >20	*Q6-Hydrology* a) Extremely Rare b)Occasional Flooding c) Regular Flooding/Soil Saturation
DBH Size of Largest Chinese Privet:	*Q7-Understory Cover*
	a) Trace (less than 1%) b) Low (1 to 5%) c) Moderate (6 to 25%) d) High (26 to 50 %) e) Majority (51 to 100%)
Notes:	*Q8-Shrub Cover* a)Trace (less than 1%) b) Low (1 to 5%) c) Moderate (6 to 25%) d) High (26 to 50 %) e) Majority (51 to 100%)
	Q9-Canopy Cover a) Trace (less than 1%) b) Low (1 to 5%) c) Moderate (6 to 25%) d) High (26 to 50 %) e) Majority (51 to 100%)

Figure 2.1**:** Sample of the riparian survey field sheet.

LOCATIONS OF STUDY SITES AND AERIAL PHOTOGRAPHY OF AUBURN (2013)

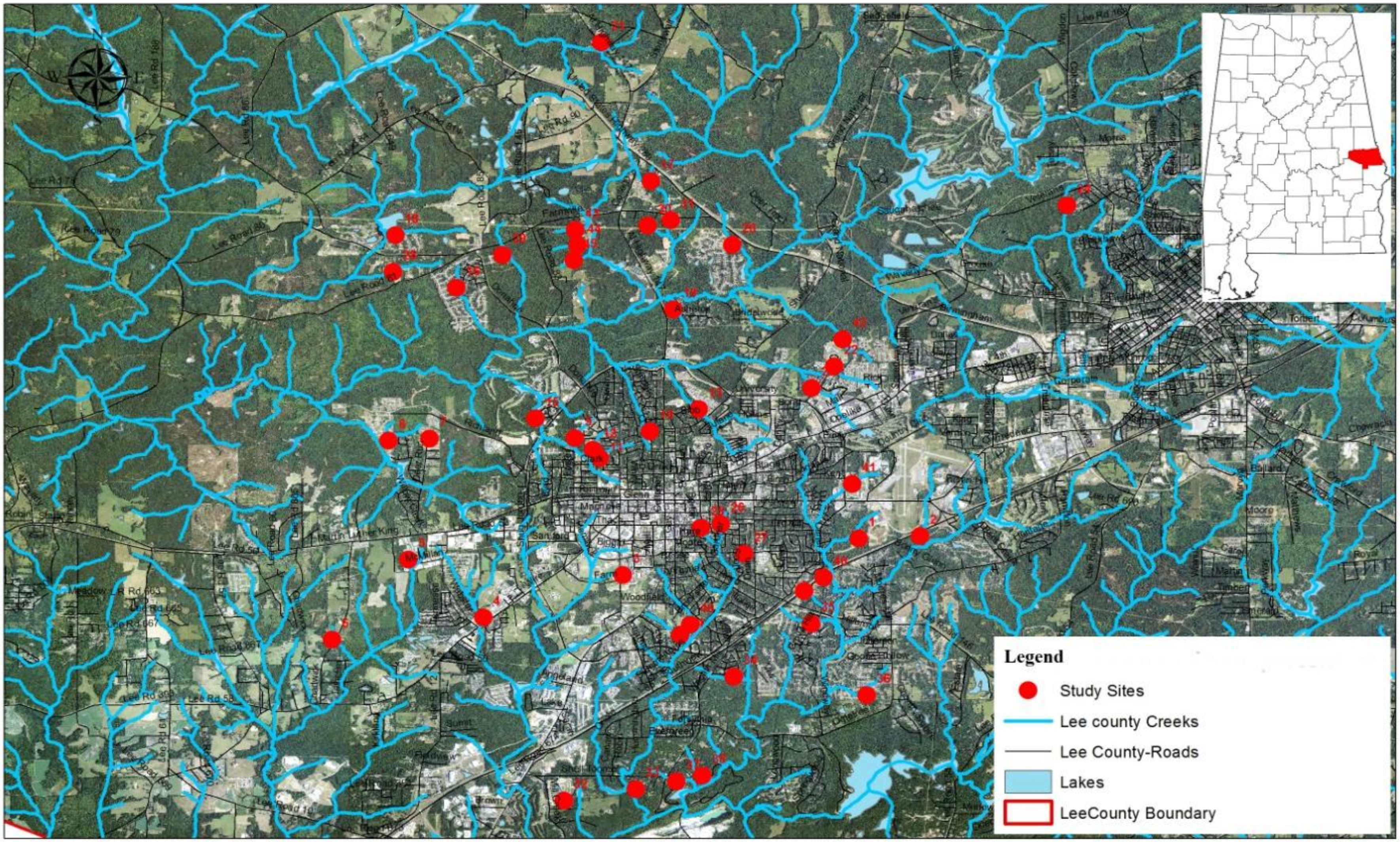

Figure 2.2: Map of 46 riparian study sites surveyed for invasive shrub species.

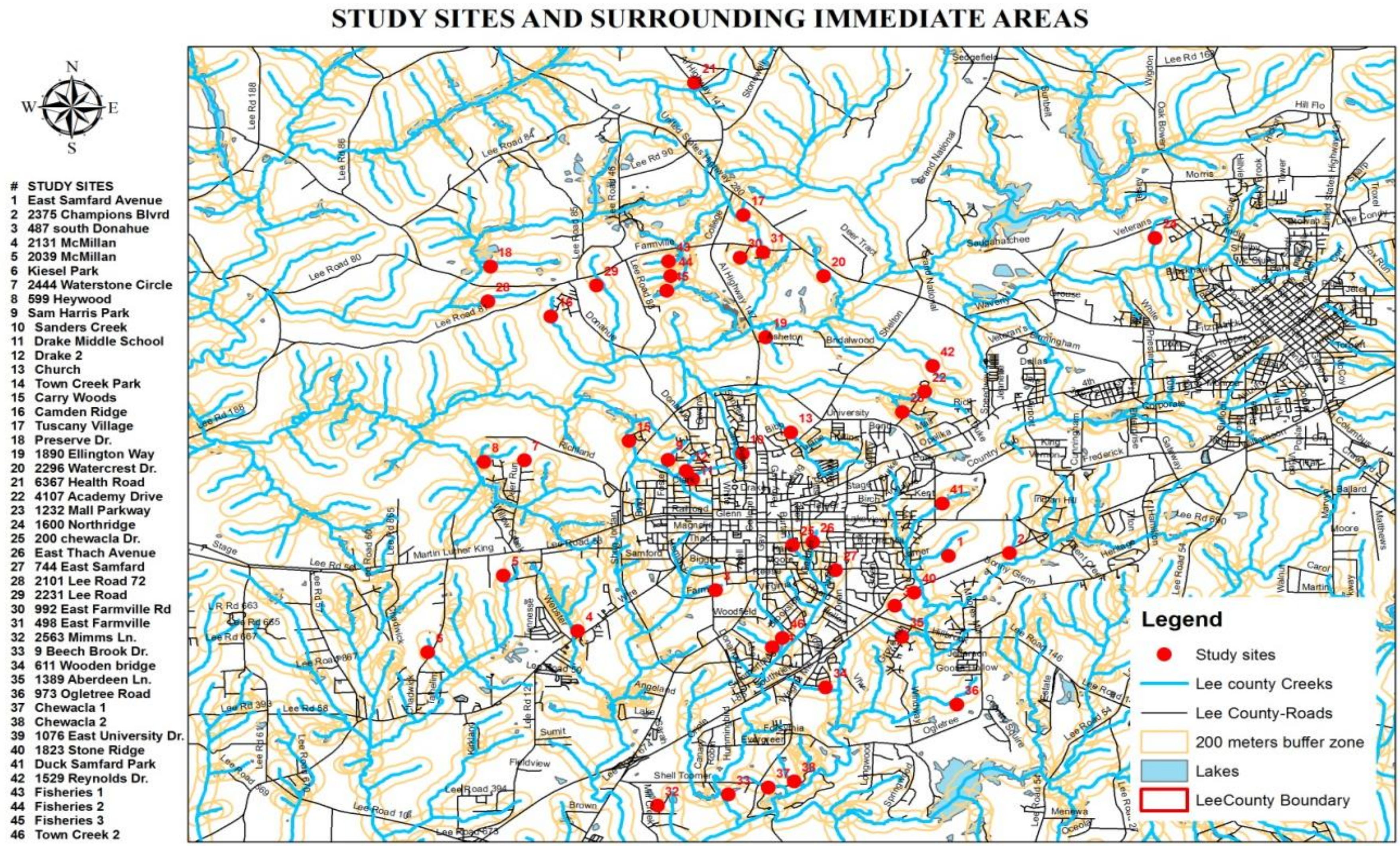

Figure 2.3: The location of surveyed study sites and 200 m. immediate buffer zones.

LAND USE/LAND COVER MAP OF AUBURN (2011)

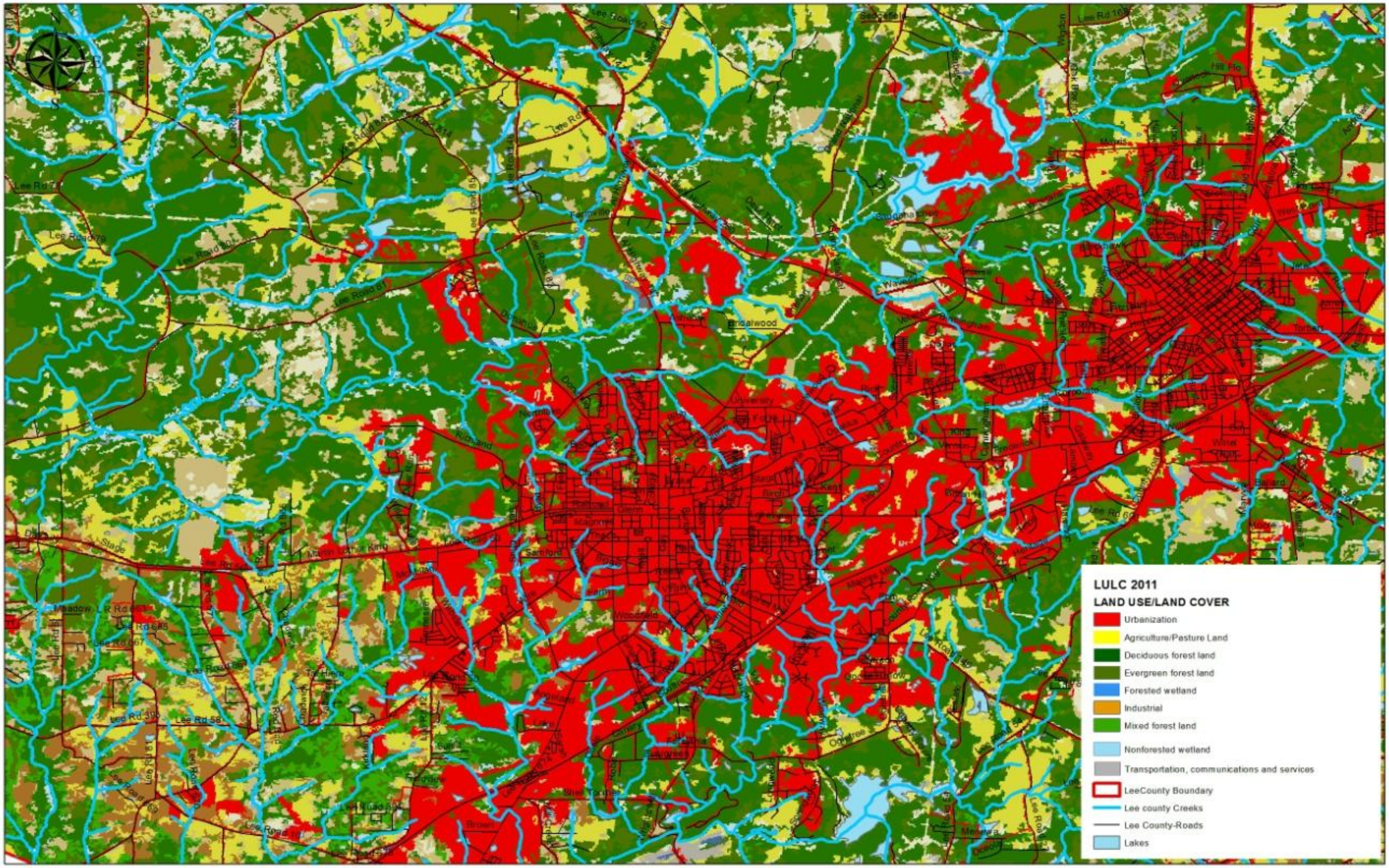

Figure 2.4: LULC map of Auburn metropolitan area with streams.

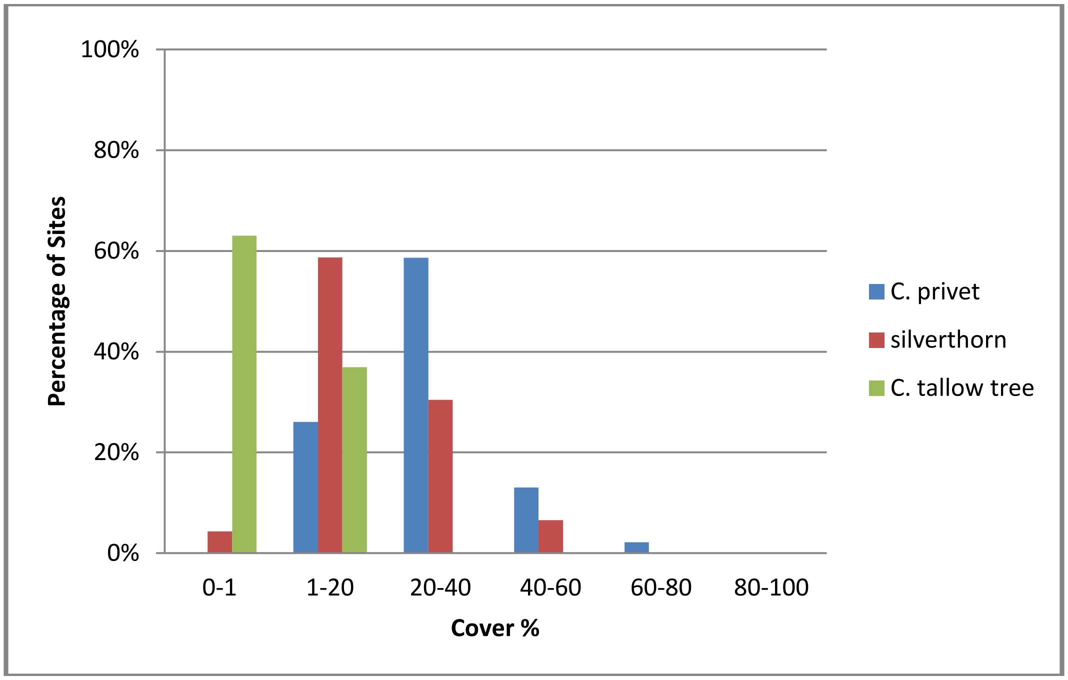

Figure 2.5: Frequency of mean total cover ranges by target species for the study sites (n=46).

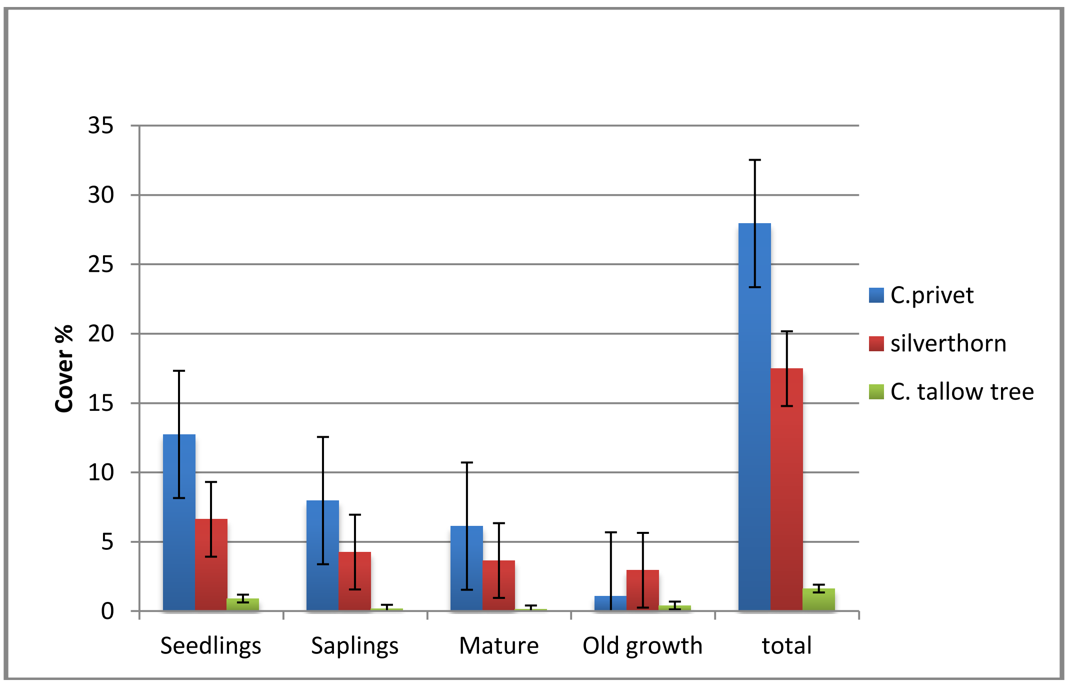

Figure 2.6: Mean (±SE) total percent cover and cover by the various growth stages of surveyed target species in riparian study sites (n=46).

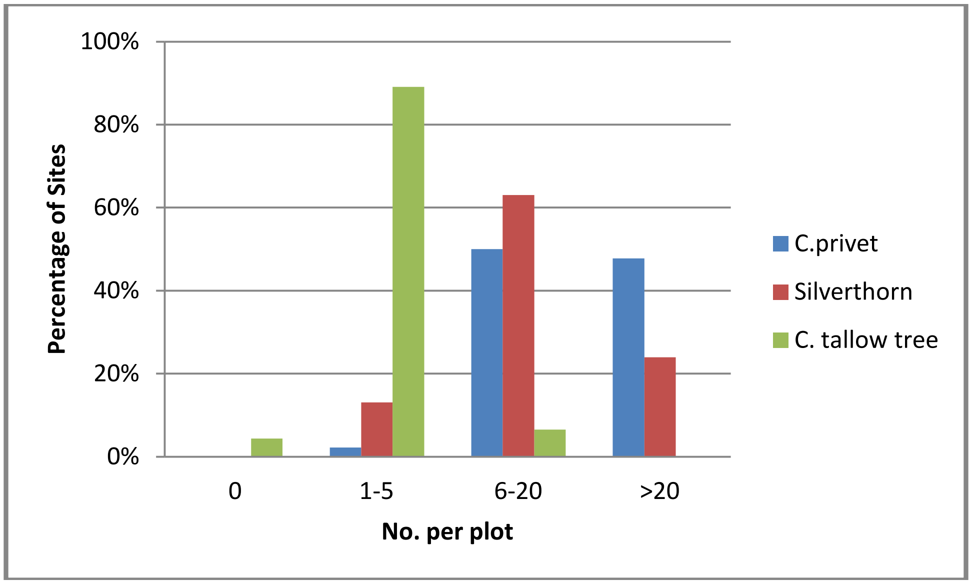

Figure 2.7: Frequency of target species stem density counted in plots from each study sites.

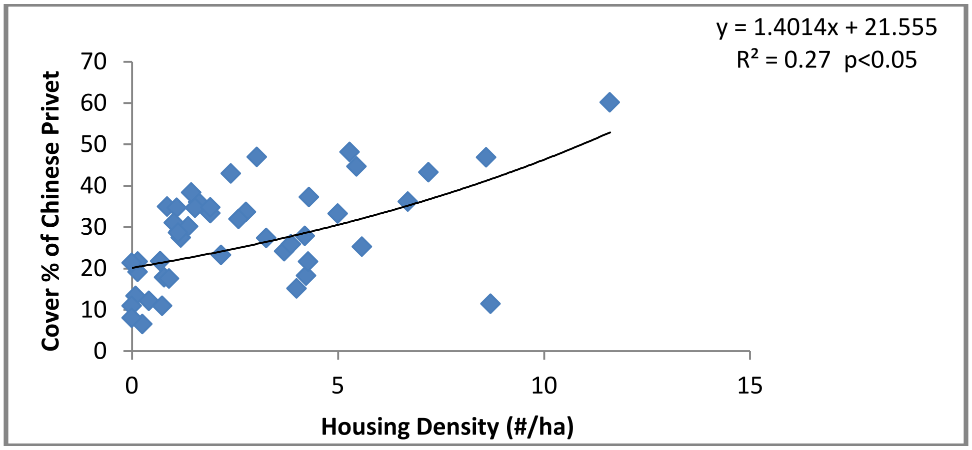

Figure 2.8: Regression results and scatter plot for total cover % of Chinese privet and housing density in the 200 m buffer zone.

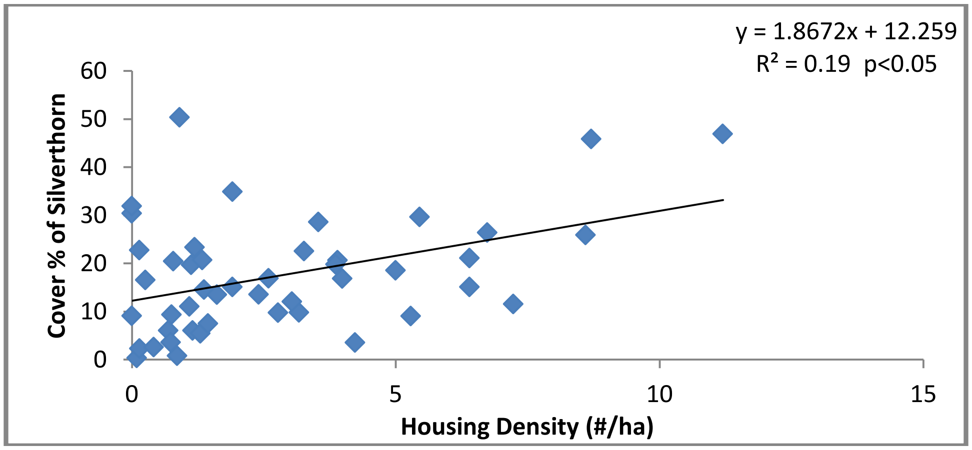

Figure 2.9 : Regression results and scatter plot for total cover % of silverthorn and housing density in the 200 m buffer zone.

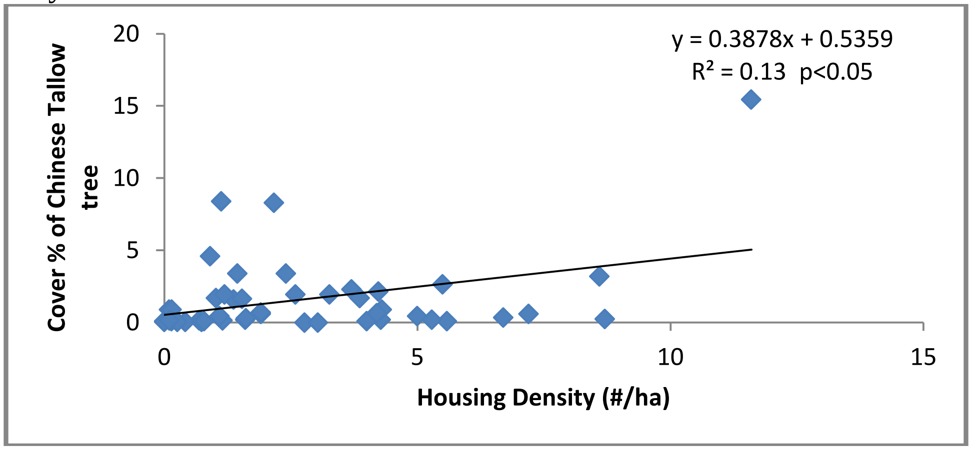

Figure 2.10: Regression results and scatter plot for total cover % of Chinese tallow tree and housing density in the 200 m buffer zone.

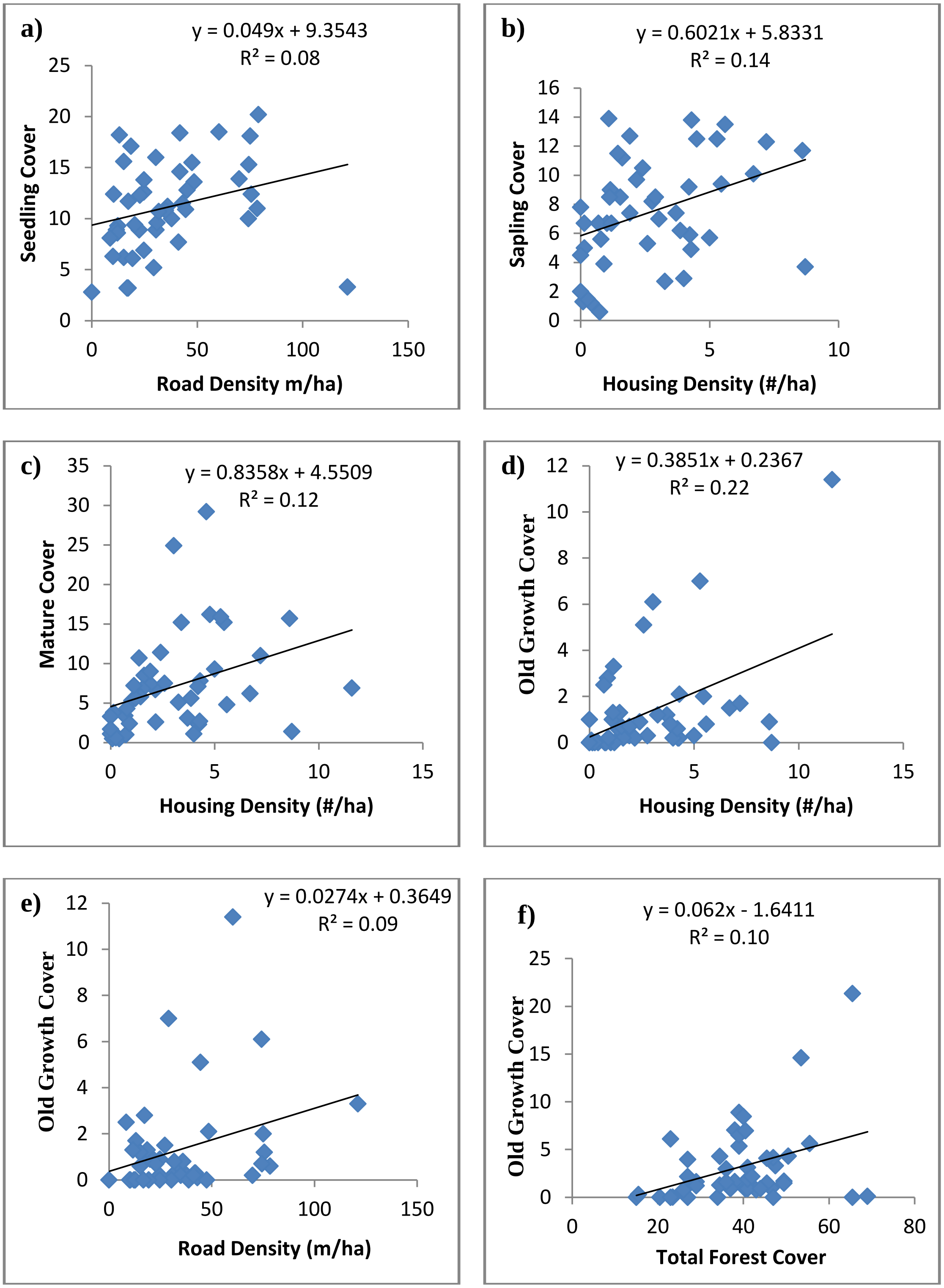

Figure 2.11 : The relationship between a) seedling cover and road density, b) sapling cover and housing density, c) mature cover and housing density, d) old growth cover and housing density, e) old growth cover and road density, f) old growth cover and total forest cover for Chinese privet ($p<0.05$).

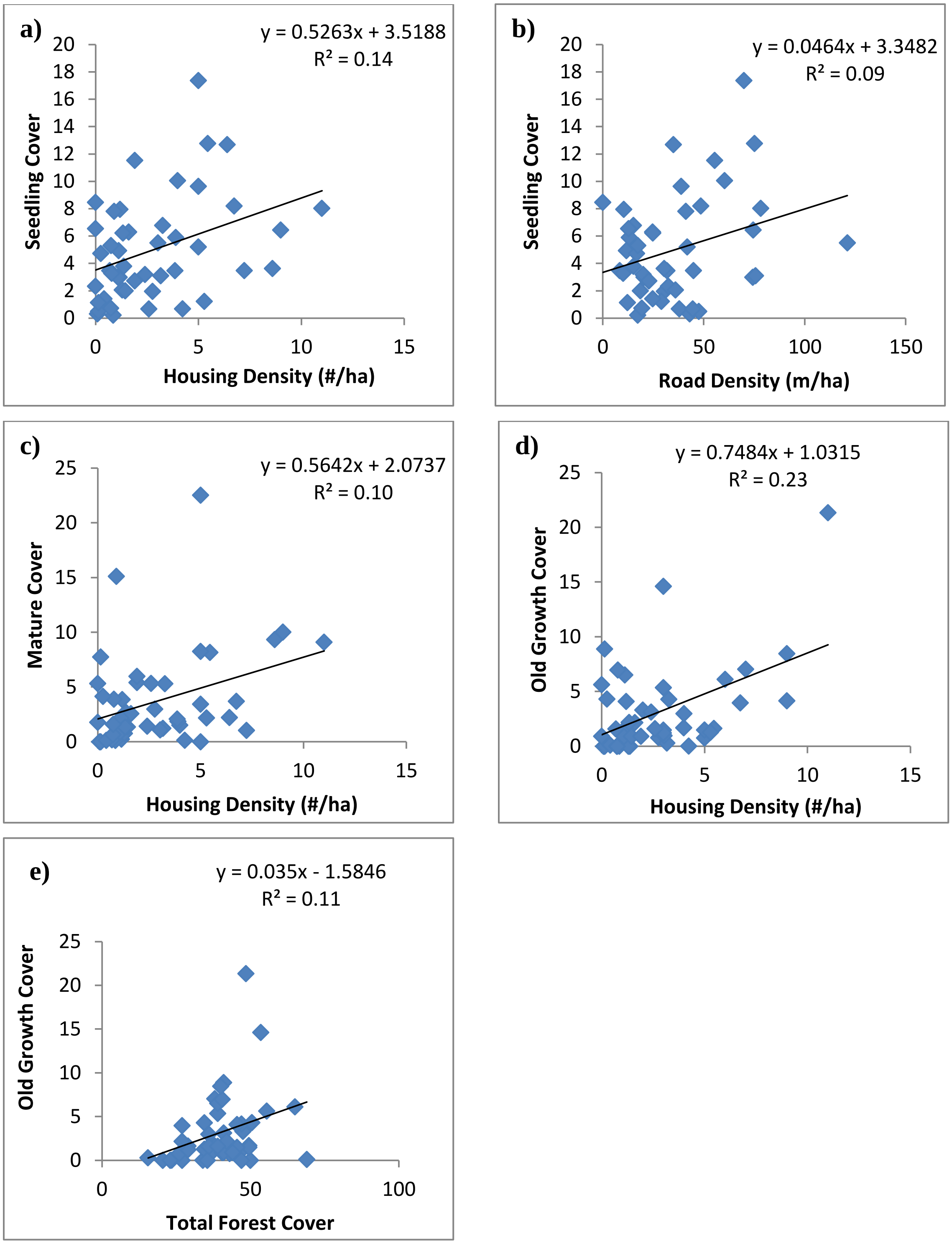

Figure 2.12: The relationship between a) seedling cover and housing density, b) seedling cover and road density, c) mature cover and housing density, d) old growth cover and housing density, e) old growth cover and total forest cover for silverthorn ($p<0.05$).

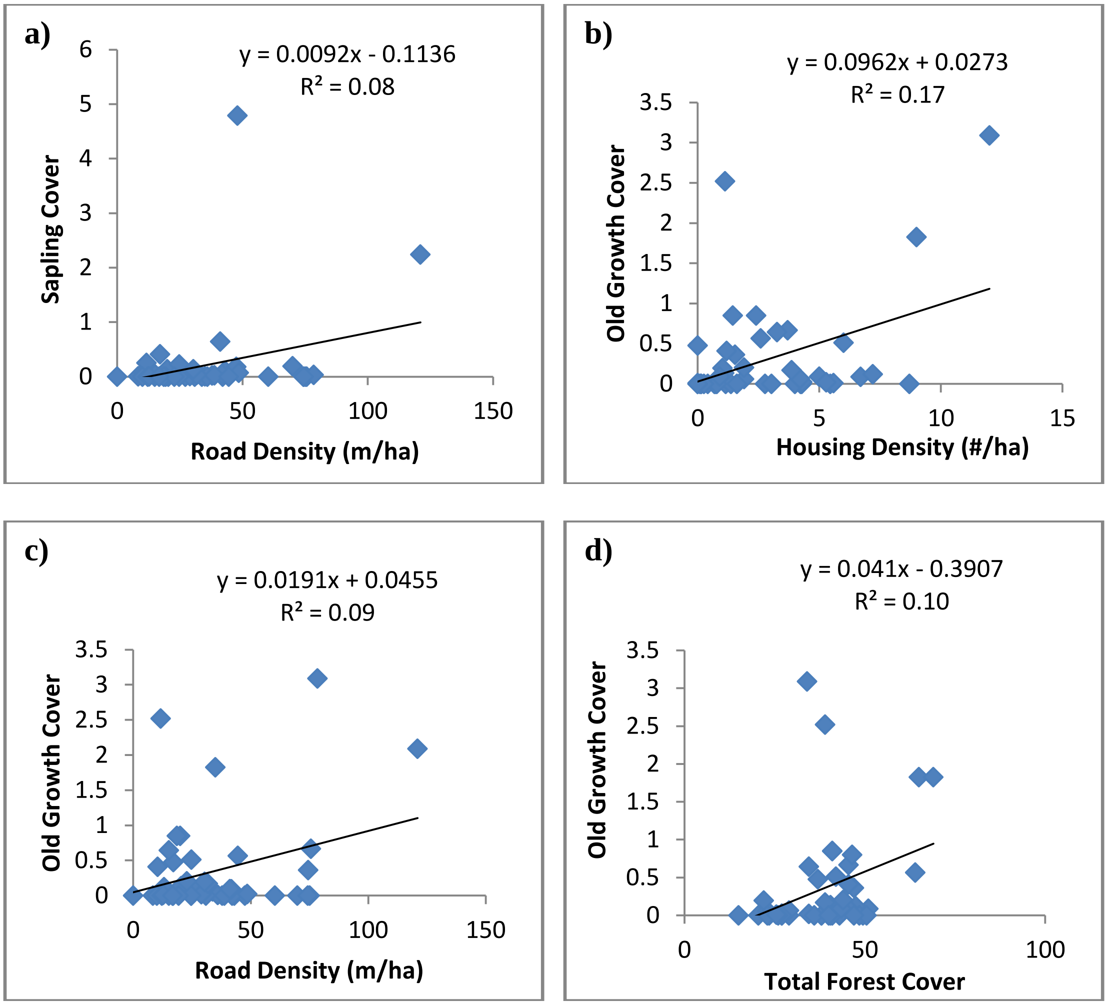

Figure 2.13: The relationship between a) sapling cover and road density, b) old growth cover and housing density, c) old growth cover and road density, d) old growth cover and total forest cover for Chinese tallow tree ($p<0.05$).

DISTRIBUTION OF CHINESE PRIVET AND 2011 LULC MAP OF AUBURN

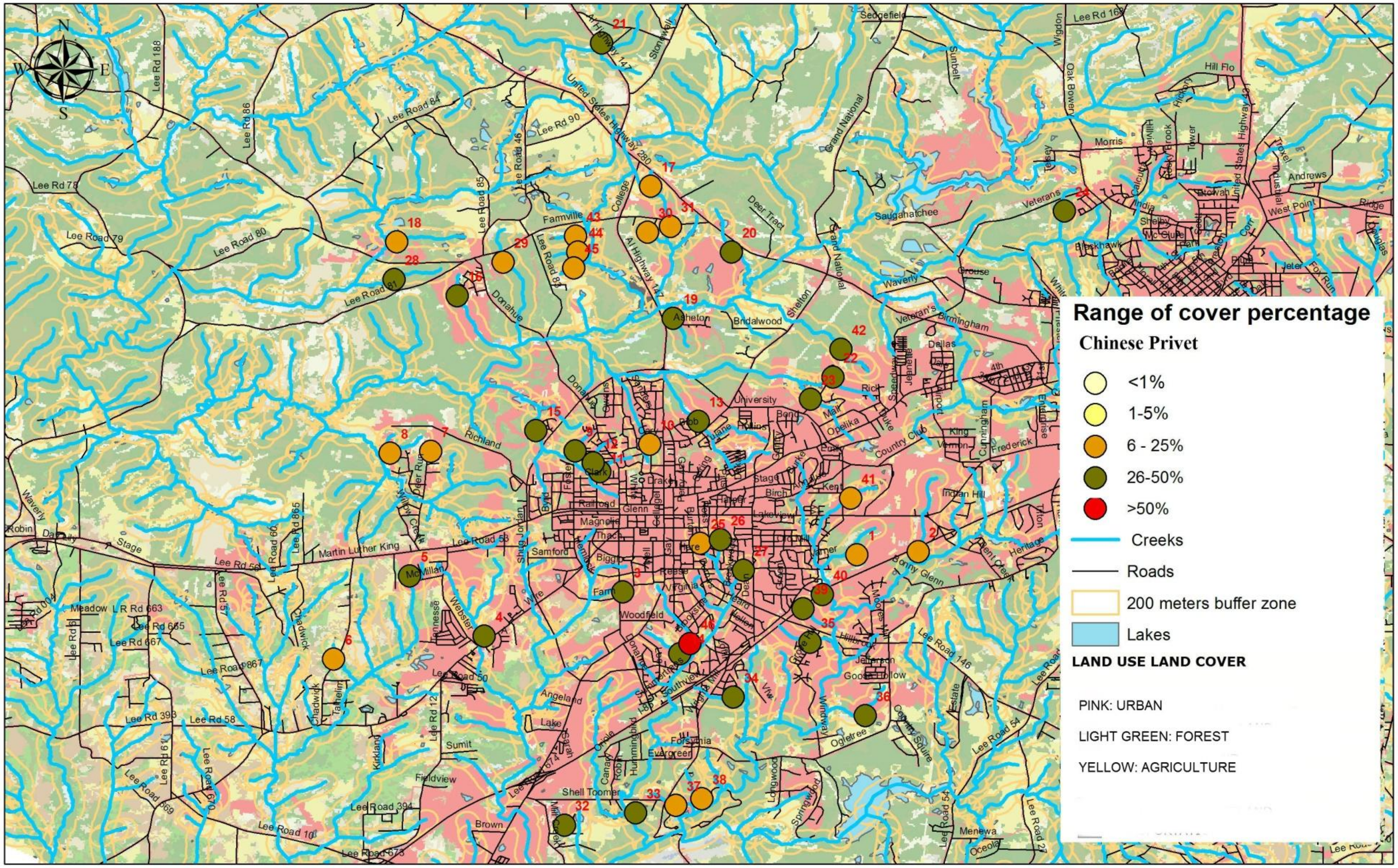

Figure 2.14: Locations and total cover ranges of surveyed Chinese privet species and LULC in Auburn.

DISTRIBUTION OF SILVERTHORN AND 2011 LULC MAP OF AUBURN

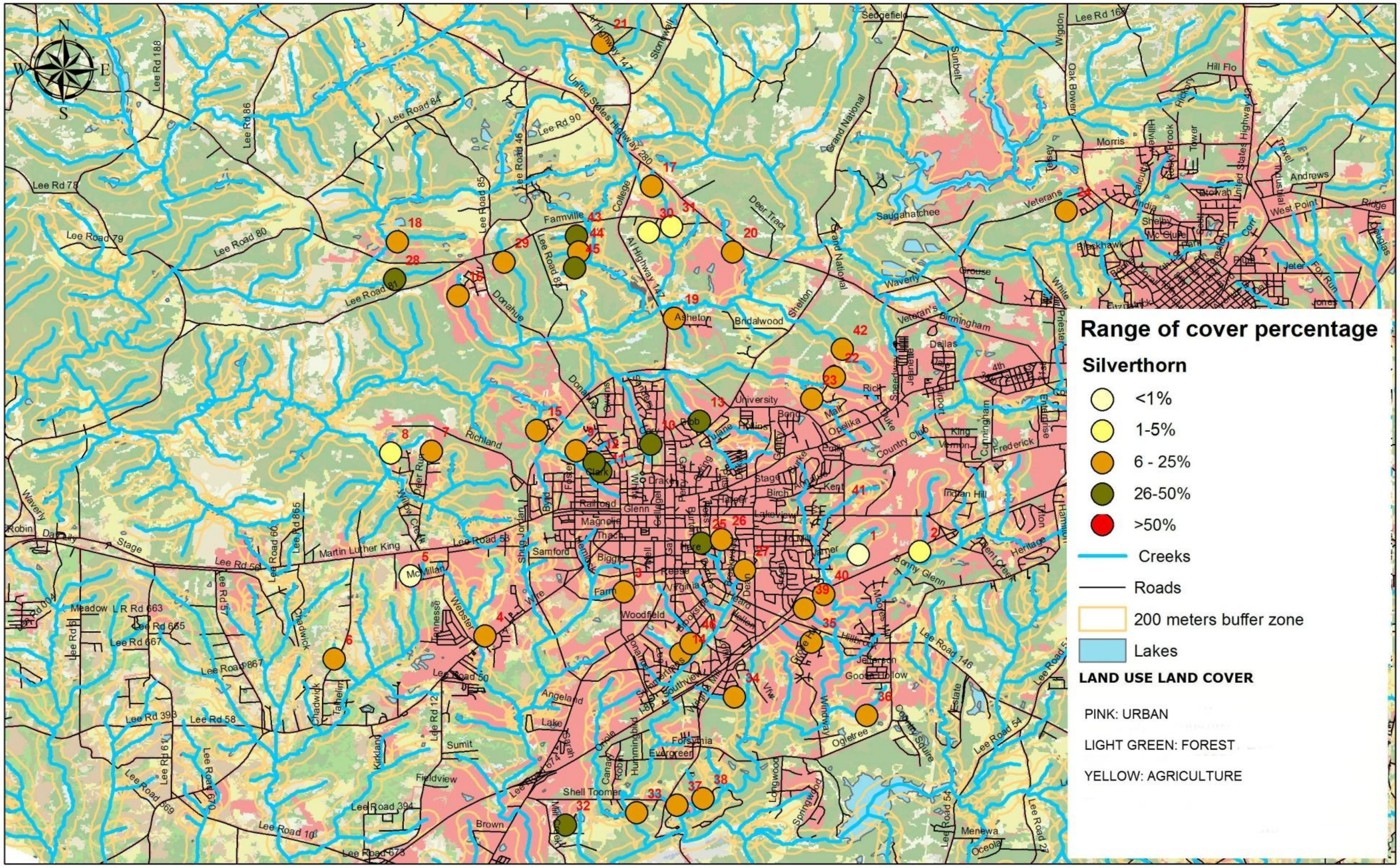

Figure 2.15: Locations and total cover ranges of surveyed silverthorn species and LULC in Auburn.

DISTRIBUTION OF CHINESE TALLOW TREE AND 2011 LULC MAP OF AUBURN

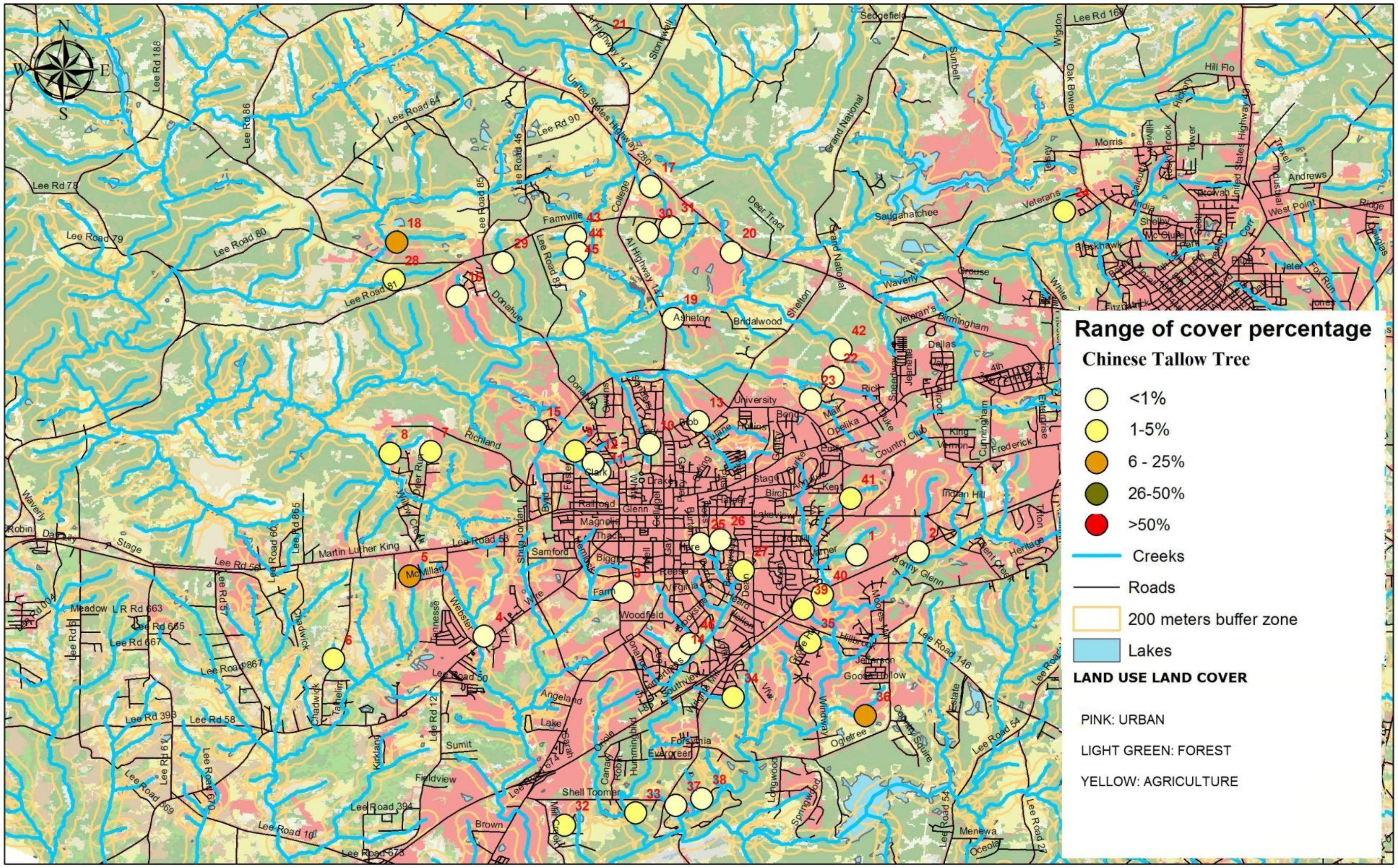

Figure 2.16: Locations and total cover ranges of surveyed Chinese tallow tree species and LULC in Auburn.

Chapter 3: EXAMINING THE DISTRIBUTION AND DISPERSAL OF CHINESE PRIVET (*Ligustrum sinense*) IN RELATION TO HISTORICAL LAND USE

Abstract

The spread of invasive species in riparian areas is an international problem which has resulted in significant loss of native species in riparian areas worldwide, including the east Alabama region. Because of the association between land use and invasive species, the distribution of invasive species may provide some indication of historical land change. Historical LULC has been shown to influence and better explain current environmental features on the landscape, including invasive species occurrence. Knowing how long invasive plants have occupied an area can provide further understanding of their dynamics and persistence. Land use and land cover patterns are expected to continue to affect the distribution of invasive species, specifically Chinese privet in southeastern United States; however, historical LULC may be important as well. This study was conducted in riparian areas of Auburn, Alabama. It was expected that the current and historical LULC may have influenced the occurrence of native and non-native plant species. In this study, we expected to find the oldest populations of Chinese privet coincided with older urban land use change, and the youngest populations near lands historically less disturbed and maintained as forest, or only recently disturbed. Another goal of this study was also to predict the succession trajectory of Chinese privet throughout the surveyed riparian areas. There was a positive relationship detected between percent cover of Chinese privet and housing density (houses/ha) of study sites. Additionally, there was a positive relationship between size and cover by all growth stages of Chinese privet. The results also indicated that the age of Chinese privet had stronger relationships with housing density around the city center and suburban areas than agriculture and forested lands in Auburn, Alabama.

INTRODUCTION

During the last century, the environment has been substantially impacted by human activity and land use change in natural landscapes. Worldwide, human population is almost 250% greater than it was in 1950 (Cohen 2003). Population change is often accompanied by substantial change in land cover. For instance, since the 19^{th} century, there have been four main time periods of prevailing disturbance and recovery associated with land use change in the southeast U.S. These have included agricultural conversion from the 17^{th} century to the 19^{th} century, timber exploitation in the early 20^{th} century, forest restoration and recovery, and urbanization starting in the late 20^{th} (Rusch et al. 2003). Each of these changes in land use/land cover (LULC) have changed the native habitat, altered hydrology, displaced native vegetation, and disturbed soils- all conditions which may ultimately contribute to increased invasive species on the landscape (McKinney 2006).

Urbanization can have a substantial impact on ecosystems by increasing the abundance of invasive species (Cohen 2003; Loewenstein and Loewenstein 2005; Pandi et al. 2014). Hobbs (2001) stated that *"landscape shifts such as fragmentation of native communities and local conditions result in a synergistic effect that causes biotic invasions."* For example, reduced tree cover in fragmented areas can increase surface run off and the dispersal of invasive plants seeds (Hobbs 2001). McDonald et al. (2008) showed that fragmentation driven by residential and commercial development can cause more forest light penetration and increase the occurrence of weedy habitats. Studying different scales of fragmentation on the landscape and local features plays an important role when determining the effects of LULC change such as urbanization. Due to new development, renewal of urban settlement, change in ownership or land management activities, as well as improvements in infrastructure applications, urban land use change has become increasingly prevalent around the world (Grimm et al. 2000).

Because of the association between land use and invasive species, the distribution of invasive species may provide some indication of historical land change. Johnson et al. (2006) showed that the demography (the age) of dispersed invasive shrubs correlated with historical patterns of urbanization. Mattingly and Orrock (2013) demonstrated that historical LULC change such as agricultural conversion may have caused soil disturbance which resulted in the current increase of invasive plant introduction and reduction of native plant communities. Since the 20th century, native habitats have been disturbed by urban development and agriculture which has caused an abundance of invasive plants to be introduced to riparian areas (Motzkin and Foster 2002). Moffatt et al. (2004) stated that urban development and establishments of new settlements close to riparian forest lands associated with a decrease in tree basal area, vegetation diversity, soil nutrient levels, and water quality in the U.S. Similarly, Lemke et al. (2012) reported that mine sites in the southern region of U.S. have been altering native habitat by displacing vegetative cover with construction, and facilitating early successional habitats for decades. They claimed that some counties in the southern Piedmont of the southeast U.S. have been occupied by invasive plants because these areas were historically maintained as mining regions and long term human habitation. Historical LULC has been shown to influence and better explain current environmental features on the landscape, including invasive species occurrence.

Knowing how long invasive plants have occupied an area can provide further understanding of their dynamics and persistence (Perkins et al. 2006). By examining the age of plants, it was possible to calculate the propagation and survival potential of invasive species (Perkins et al. 2006). Dietz and Schweingruber (2002) analyzed annual growth rings of woody vegetation to determine age and allow them to predict the tendency for invasion in a landscape. By examining the age of woody vegetation, the approximate time of introduction and establishment of invasive plants can be estimated and compared with related disturbance levels in vegetation and habitat to examine the history of alterations on a landscape (Dietz and Schweingruber 2002). According to Flory and Clay (2006), by looking at the characteristics and ages of invasive plants, historical land alterations can be examined and future invasions may be predicted. They suggested that the ages of invasive species could be correlated with land use change over time.

According to Mack (2001), one of the main issues caused by invasive plant species is economic loss. There have been some management activities implemented to control invasive species, but improved strategies are severely needed. In addition, a significant budget is often allocated for controlling invasive shrubs regardless of new introductions in the U.S. (Hartman and McCarthy 2007). Therefore, understanding the relationship between land alterations by humans, the effect it has on invasive species infestations, and the impacts of infestation are considerably crucial to effectively manage for these species.

Chinese privet (*Ligustrum sinense* Lour.) is an evergreen, invasive shrub which has invaded riparian forests throughout the southern United States (Figure 3.1), where it can form nearly monotypic stands and displace native plant communities (Hanula et al. 2009). It is a member of the *Oleaceae* family and it was first introduced into the United States from China in 1852 as an ornamental plant (Langeland and Burkes 1998). According to The National Resources Conservation Service (NRCS 2014), it is one of the most widely spreading and threatening invasive species in the region (http://www.nrcs.usda.gov). Cuda and Zeller (2000) reported that Chinese privet escaped from cultivation by 1932 and adapted in states such as north

Florida, Georgia, Alabama, Kentucky, North and South Carolina, Tennessee and Mississippi. Brantley (2008) reported that it is a shade tolerant and aggressive shrub that disturbs forestlands by displacing native plant specimens in the understory. Also, Chinese privet has occurred more than 1 million ha in riparian forests of U.S. and the relationship between land use and growth range of privet is still ambiguous (Green and Blossey 2014). Riparian areas are commonly infested by Chinese privet due to its regenerative characteristics which provides rapid spread and colonization (Figure 3.2 and Figure 3.3), and these shrubs can easily adapt to a wide range of conditions including floodplains of the southeastern United States.

Although current land uses can influence Chinese privet in riparian ecosystems, historical land use change and landscape characteristics may further explain the current distribution pattern of these shrubs. For instance, historical land use can relate to increased pH level of soils and rates of net nitrification which likely promoted Chinese privet proliferation and loss of native vegetation (Holle and Motzkin 2007). Also, rural depopulation and forest clearing activities around farmsteads resulted in loss of vegetative cover and the introduction of several non-native plants such as Japanese barberry (*Berberis thunbergii*) and leatherleaf mahonia (*Mahonia bealei*) which were often used for short-term restoration and land stabilization by local managers (Ward 2002). According to Lundgren et al. (2004), historical land use may play an important role in determining where higher nutrition levels occur in the soil which allows privet to outcompete native vegetative cover in riparian areas.

Land use and land cover patterns are expected to continue to affect the distribution of invasive species, specifically Chinese privet in southeastern United States; however, historical LULC may be important as well. This study was conducted in riparian areas of Auburn, Alabama where there have been substantial changes in LULC over the last 50 years. It was expected that the current and historical LULC may have influenced the occurrence of native and non-native plant species. In this study, we expected to find the oldest populations of Chinese privet coincided with older urban land use change, and the youngest populations near lands historically less disturbed and maintained as forest, or only recently disturbed. Another goal of this study was also to predict the succession trajectory of Chinese privet throughout the surveyed riparian areas. This study will be beneficial for land owners and managers to derive information, such as how long Chinese privet has been colonized, the hotspots regarding initial colonization, and if historical patterns of land use and privet occurrence match current patterns.

METHODS

Invasive species survey

As described in Chapter 2, riparian surveys were conducted in 2014 using a plant community rapid assessment survey (see examples by Bellemare et al. 2002, Lundgren et al. 2004, Forsyth et al. 2004) designed to collect extensive data on invasive species cover, growth form and density. Study sites were 46 riparian areas located throughout Auburn, AL (Figure 2.2, Chapter 2). Because this study emphasized urban land use, emphasis was placed on sampling sites throughout the city proper and sampling riparian streams in residential areas represented by a wide range of development ages. Details regarding field surveys and data collection were provided in Chapter 2. As part of this effort, the largest Chinese privet specimen was determined for each plot (10 per study site) by measuring its diameter at breast height (DBH). For multi-stemmed Chinese privet species, DBH size of the largest stem was measured. As also indicated in Chapter 2, total privet cover and cover by various growth stages were also collected. The criteria for privet growth stages were for seedlings (< 1.5 m tall), saplings (< 2.5 cm dbh), mature

(2.5-10 cm dbh), and old growth stage (> 10 cm dbh). We did not discern the differences between seedlings and sprouts for surveys.

Dendrochronological analysis

In order to estimate the age of Chinese privet, dendrochronological analysis was used to determine the relationship between DBH and shrub age. Twenty stem sections were collected from Chinese privet specimens in Town Creek Park in Auburn, AL, between June-August 2014. Sections represented a wide range trunk diameters (3.7-15.5 cm). The stem sections of larger Chinese privet (DBH >10cm) were collected using a tree increment borer (36.5 cm. length and 1.5 cm. caliper). For smaller privet (DBH <10cm), a handsaw was used to carefully cut the shrub and a cross section of the trunk at DBH height. Cross sections were transported in freezer bags and the tree ring cores in plastic straws. All were labeled with the measured DBH sizes, and stored in a refrigerator at $4^{o}C$ until processing.

To minimize fungal growth and desiccation, the stem sections were examined within 2 days of being collected in the field. Increment cores were glued on wooden boards and labeled by the DBH size of Chinese privet that they were attained from. Paynter et al. (2003) detailed that analyzing woody invasive shrubs cross sections are often problematic due to diffuse tree rings that can become less perceptible over time. To improve visual detection of growth rings, the fluorescent method has been suggested to eliminate diffuse porous and detect annual growth rings (Lussier et al. 2004). For this method, the cross section of stems and cores were sharply cut and shaved using a hand planer. The surface of each stem sample was painted using a fluorescent-yellow marker to obtain an optimal surface wetness. By applying a white fluorescent light on the yellow marked stem sections, the growth rings of each sample were more perceptible and counted under a magnifying scope. The approximate ages based on the number of apparent tree rings of each specimen were recorded.

Using the annual rings and DBH for each sampled Chinese privet, a best-fit regression analysis was conducted to determine the mathematical relationships between the variables (Figure 3.6). The oldest sample was 46 years old and was obtained from a 51.68 cm DBH size stem, while the youngest privet was 3 years old with a DBH of 3.05 cm. It should be noted that ages were considered approximations because of several issues inherent to dendrochronology and these methods. This method did not take into account the number of years for the Chinese privet to grow to the height of DBH at which the corresponding rings were pertinent to be detected. Other inaccuracies aging trees can result from false rings and missing rings (Schweingruber and Poschlod 2005). It should also be noted that the area that the samples collected in Town Creek Park was different than surveyed riparian forests in the park.

Ages of the largest privet on each of the riparian sites was estimated using the mathematical relationship. The oldest specimen from each study site was used as an approximate age of Chinese privet establishment for this site. Based on the regression equation, the oldest Chinese privet was found in Town Creek Park (site #14) at an approximate age of 24 years. The youngest specimen was derived as approximately 7 years old in Fisheries 3 (site #45). This study site was different from the area that samples were taken. The mean age across all study sites was found 16.4±3.6 years old (Table 3.3).

Historical LULC change analysis

To evaluate historical LULC throughout the study sites, aerial photographs (in years 1966, 1986, and 2011 respectively) of each study site and its 200 m buffer (see Chapter 2) were manually digitized using ArcGIS 10.2. Within each of the buffer zones, the size of buffer zones (ha), the number of residential houses, and the road length (m) were measured by using aerial photographs of Auburn for years 1966, 1986, and 2011 (Table 3.1). Gravel, paved, and driveways were considered as roads in the measurement. The housing density (houses/ha) and road density (m/ha) were calculated for each study site for 1966, 1986, and 2011. The rate of increase in housing density between the 3 specific years was calculated as follows:

$$H = \frac{housing\ density_t - housing\ density_i}{t-i}$$

where *H* is the yearly increase in housing density, *t*, *i* are the specific years (2011-1966, 2011-1986, and 1986-1966), and denominator shows the number of years between dates. The same formula was modified and applied for calculation of yearly increase in road density derived from digitized maps (Table 3.2).

Dominant historical and current land use and land cover for each site was estimated by looking at LULC maps of Auburn obtained from the City of Auburn website (http://webgis.auburn alabama.org). Maps included the extent of urban, agriculture/pasture, industrial, wetland, forested and non-forested land use changes surrounding Auburn, AL for 1976 and 2011 (Figure 3.4 and Figure 3.5). To determine if study sites historically occurred within agriculture, forest, or urban lands (the three dominant types), sites were overlaid on 1976 LULC maps. Surrounding land use/land cover classifications for each site were then designated for 1966 and 1986 by checking for consistency with land uses on the 1966 and 1986 aerial photographs and reclassifying sites if necessary. Historical land conditions for each site were used to better interpret relationships between LULC and Chinese privet through data analyses as described below.

To determine the spatial distribution of oldest Chinese privet at each study site, sites were overlaid on the 1976 and 2011 LULC map and shown with oldest privet age. In this map, the oldest Chinese privet at each study site was identified using a yellow color which was used to show youngest privets (5-10 years old) up to a red color indicating the oldest privet species detected (20-25 years old).

Data analyses

To evaluate the relationship between historical urban measures (housing density (houses/ha) and road density (m/ha) in 1966, 1986, and 2011) and cover % of Chinese privet, best-fit regression analysis was applied using SAS 9.3. Using all 46 sites, regression analysis was used to find the potential relationships between percent cover (2014) of Chinese privet and urban measures (road and housing density) for 1966, 1986, and 2011. A similar analysis was conducted between privet stand age (2014) and historical urban measures for 1966, 1986 and 2011. To evaluate for potential relationships specific to sites from individual LULC categories (urban, forest, agriculture), regressions between 2014 privet cover and house/road density (in 1966, 1986 and 2011) were stratified and analyzed based on LULC category. In some years, there was not enough study sites (n<12) to conduct reliable regression analyses and for these scenarios regression analyses were omitted.

To determine the potential role of historical colonization of privet, best-fit regression was also used to examine the potential relationship of housing density (houses/ha) and road density (m/ha) (in 1966, 1986 and 2011) on the age of the oldest Chinese privet at each site. In both cases, the R-square (R^2) of each regression was compared between years to determine if there was improvement in the explanatory power of the independent variables (housing density, road density) on measures of percent cover and the stand age of Chinese privet variables per each study site. Finally, to examine successional trends related to privet colonization, best-fit regression analysis was used to examine the potential relationship between age of oldest privet and total cover of different growth stages. All regression results were considered significant at $p<0.05$ and $p<0.10$.

RESULTS

LULC change analysis

The 1976 LULC maps showed that 45.6% of the study sites were located within agricultural lands, 27.2% within urban, and 27.2% within forested lands in 1976 (Figure 3.7, Figure 3.9). Review of aerial photographs from 1966 showed that these LULC designations for the study sites were unchanged from 1966 to 1976. By 1986, 30.4% of the study sites were located within agricultural lands, 39.1% were urban, and 30.4% were within forested lands. By 2011, 86.9% of study sites were urban, 6.6% were agriculture, and 6.6% were forest lands (Figure 3.8, Figure 3.9). Comparison of LULC maps between 1976 and 2011 confirmed that urban land use became more prevalent by 2011 than agriculture and pasture land use in 1976. Since the majority of study sites were close to urban core, land use change was estimated as more prevalent for landscapes close to urban core and suburban lands surrounding Auburn, AL (Figure 3.8, Figure 3.9).

Annual increase rate and average housing density and road density were determined for each study site (Table 3.2). Average housing density (houses/ha) was 0.9±1.2 and the annual increase rate was 0.02± 0.03 houses/ha between 1966 and 2011. However, mean housing density increased by 0.4±0.7 houses/ha and the annual increase rate was 0.02± 0.03 houses/ha in 1966-

1986 period. Average housing density was 1.1±1.2 houses/ha and annually increased by 0.03±0.04 houses/ha between 1986 and 2011. Mean road density increased 17.3±24.2 m/ha at an annual rate of 0.4±0.5 m/ha yearly in the period of 1966 to 2011, while it increased 16.9±15.2 m/ha at an annual rate of 0.8±0.8 m/ha between 1966 and 1986. Average road density also increased by 34.1±24.3 m/ha between 1986 and 2011. Like housing density (houses/ha), the most increase in road density was found between 1986 and 2011.

Relationships between historical urban measures and Chinese privet

Historical measures obtained from aerial photographs for 1966, 1986, and 2011, and Chinese privet percent cover in Chapter 2 were used for various regression analyses. There was a positive relationship between cover % of Chinese privet and housing density (houses/ha) of study sites for all three observation years although relationships were fairly weak (Table 3.4). The highest R^2 for housing density (1986) and total privet cover % was found at $R^2 = 0.32$. The $R^2 = 0.27$ was found for housing density (2011) and privet cover %, where it was the lowest for housing density (1966) at $R^2 = 0.19$ (Figure 3.9). Road density (1986) only had a positive and significant relationship at the 10% significance level (p=0.08) and $R^2 = 0.12$ (Table 3.4; Figure 3.9).

Examining trends using sites only mapped urban sites for each year, there was a positive relationship between urban measures (road and house density) and percent cover of Chinese privet (Figure 3.10). The highest R^2 for housing density (1986) and total privet cover % was found at $R^2 = 0.56$ (Figure 3.10-b), where R^2 was 0.52 between housing density (2011) and cover % of privet (Figure 3.10-a). There was also a positive relationship between road density (1966, 1986, and 2011) and cover % of Chinese privet (Figure 3.10-d,e,f).

Examining trends using sites only mapped as forested in 1966 and 1986 (there were insufficient sites to run regression for forested sites in 2011), the highest R^2 was between housing density in 1966 and total privet cover % (R^2=0.27) (Figure 3.11-b). There was also a positive relationship between housing density (1986) and cover % of privet at R^2 =0.26 (3.11-a). However, both these relationships are suspect as they are driven by one or two points with excessive leverage. Relationships between road density and cover % of privet was insignificant (3.11-c,d).

For sites mapped as agriculture in 1966, 1986 and 2011 the number of sites mapped as agriculture in 2011 was insufficient (n<12) for regression so regressions were restricted to agriculture sites in 1966 and 1986. For the other years, the highest R^2 for housing density (1966) and total privet cover % was found at R^2=0.32 (Figure 3.12-b). There was also a positive relationship between housing density (1986) and cover % of privet at R^2=0.28 (Figure 3.12-a). Considering road density for 1966 and 1986, they were positively associated with cover % at R^2=0.27 and R^2=0.20, respectively (Figure 3.12-d,c). Although these sites had a positive and significant relationship, it was noted that the highest explanatory power was found for mapped urban sites.

Regression analysis was also used to evaluate the relationship between historical urban measures in 1966, 1986, and 2011, and the estimated age of the largest Chinese privet for all study sites in 2014. For this analysis, the highest R^2 between housing density and the oldest Chinese privet age was found in 1986 at R^2=0.24. The R^2=0.17 between housing density (1966) and the oldest Chinese privet age, while the R^2 = 0.19 for oldest Chinese privet age and housing density (2011) (Table 3.4; Figure 3.13). Considering only sites that were mapped as urban, the highest R^2 for housing density and the oldest Chinese privet age was also in 1986 (R^2 = 0.44)

(Figure 3.14-b). There was a positive relationship between housing density (2011), housing density (1966), and privet age at R^2=0.27, R^2=0.26, respectively (3.14-a,c). Road density (1986) and road density (2011) were also positively associated with the oldest Chinese privet age at R^2=0.27 and R^2=0.19, respectively (Figure 3.14-d,e). Like housing density, the relationship between road density (1966) and privet age was also weak (Figure 3.14-f). For only sites that were mapped as agricultural land use, the highest R^2 for housing density (1986) and the oldest Chinese privet age was found in 1986 (R^2=0.22) (Figure 3.15-a), where R^2=0.22 between housing density (1966) and privet age (Figure 3.15-b). There was a positive relationship between road density (1986) and the oldest Chinese privet age at R^2=0.17 (Figure 3.15-c) however the relationship was relatively weak (Figure 3.15-d).

Relationship between Chinese privet age and cover

There was a positive relationship between size and cover by all growth forms of privet ($p<0.05$). The highest explanatory power was found for the oldest privet age and old-growth stage at R^2=0.49 (Figure 3.16). Also, the oldest privet age had a higher explanatory (R^2 =0.31) power on mature form, while the R^2=0.21 for seedling form of privet (Figure 3.16). A significant relationship ($p<0.05$) was also detected between the estimated age of the largest Chinese privet and total cover by Chinese privet (R^2 = 0.22).

Chinese privet and historical LULC relationship

By categorizing sites based on their historical conversion between years (e.g., forest in 1986 to urban in 2011) or lack of change (e.g., urban in 1986 to urban in 2011) it was possible to further examine the relationship between privet cover and LULC. The results of the relationship between LULC change over time and mean % cover of Chinese privet are represented in Table 3.8. Since 1966, the highest average % cover was found for sites with land use change from agriculture in 1986 to urban in 2011 (n=17). These sites had a mean privet cover of 44.9±16.4% between 1986 and 2011. In the same period, 22 sites changed from agriculture to urban land use and had an average 33.6±11.8% cover of Chinese privet, where there were only 3 sites found which converted from agriculture to urban at average 30.5±8.4 % cover of privet (Table 3.8). Sites which stayed urban over time also had higher privet density than sites stayed forest and agriculture between 1966 and 2011. Average percent cover of Chinese privet was higher in 1986 and 2011 for all land use change classes than 1966-2011 and 1966-1986.

For each transect, the age data set of Chinese privet were combined with 1976 and 2011 LULC maps (Figure 3.7 and Figure 3.8) of study sites. Overlaying privet stand age on the 1976 LULC map of Auburn illustrated that land cover with agriculture, urban, and suburban lands tended to coincide with the oldest Chinese privets (for approximately 80% of study sites). In contrast, Chinese privet stand age was overlaid on 2011 LULC map and it was estimated that approximately 45% of study sites which were demonstrated on 1976 LULC map as agriculture and nearly 17% of forest areas, illustrated as urban lands on 2011 LULC map (Figure 3.8, Figure 3.9). It was found that areas converted to urban tended to be occupied by the oldest Chinese privets. Also, for both the 1976 and 2011 LULC maps, estimated stand age were significantly lower in older forest lands than urban landscapes of Auburn, AL. Chinese privet occupied more study sites around the city center and suburban areas than current forested lands.

DISCUSSION

In this study, using Chinese privet (one of the region's most common invasive species), we explored potential relationships between historical land use and colonization of Chinese privet in Auburn, AL. The goal of this study was to find the oldest populations of Chinese privet and determine if Chinese privet species coincided with historical land use change. We also expected that potential relationships between age of the infestation and growth stages of privet may provide insight regarding the successional trajectory of these invasive species throughout the surveyed riparian areas. For the past 45 years, the highest annual increase rate of housing density was estimated in 1986-2011 (Table 3.2). There was a positive relationship between total percent cover of Chinese privet and housing density of study sites for all three observation years (Table 3.4, Figure 3.10-a,b,c). Specifically, urban sites had the highest explanatory power for the relationship between urban measures and total percent cover of privet (Figure 3.10). This relationship may show that increase in urban sites can have an important influence on distribution of Chinese privet in riparian forests by causing surface runoff, flooding, and fragmentation which can lead privet seeds to disperse and recolonization of species over time. Also, sites historically maintained as agriculture in 1966 also were positively associated between housing density and percent cover of privet (Figure 3.12-b). Even if there were less house occurrences detected for some agriculture sites (Figure 3.12), percent cover of Chinese privet was high. This could be because privet might also be used as ornamental aspects around the farmsteads and after abandoning agricultural lands, Chinese privet could have maintained reproduction and spread over time. This may suggest that increase in urban land use is positively associated with abundance of Chinese privet in riparian forests. The consequences of urban disturbance such as increase in impervious surface, distortion of waterways, flooding, and soil degradation can create harsh conditions which may also promote privet to thrive. It was interesting to note that relationships between urban measures (house and road density) and percent cover by privet were improved when sites were separated based on their mapped land use. This would suggest that the response of privet to housing and roads may follow different patterns depending on surrounding LULC. These patterns were certainly less clear when sites from different LULC categories were lumped together (Figure 3.9).

Since the oldest Chinese privet detected in our study sites was approximately 23 years old (Table 3.3), this suggests that housing density around 1986 may partially explain early Chinese privet colonization and spread within riparian areas of this study. Also, there has been significant development in Auburn since 1986 which likely increased the introduction of Chinese privet as an ornamental species used for hedge planting. Both housing and road density scatterplots showed that study sites with less than 20% total cover by Chinese privet in 2014 were all either areas that were agricultural or forested in 1966 (Figure 3.9). It should be noted that some agricultural lands and forested lands also had higher cover. Looking at more recent land uses (2011), where nearly all sites have become urbanized, urban areas now capture a full range of Chinese privet cover. Newer developments likely caused more disturbances such as fragmentation, hydrologic distortion, and open space landscapes that may have influenced the spread of seeds and further colonization of Chinese privet in Auburn, AL.

For this study, there were some factors associated with estimating the stand age and DBH size of Chinese privet that should be considered. The use of DBH size for estimating stem age of privets can be somewhat controversial because single stem privets can have more annual growth rings than multi-stem privets, and two samples with the same DBH size can have different number of annual growth rings. Also, there could be a difference in growth depending on landscape position and stem re-growth which were not tested here. For growth stages of Chinese privet, seedling cover can also have sprouts which we did not discern the difference between these two groups. Others have used privet stem size and density to make estimates of colonization. Greene and Blossey (2014) used the largest individuals of Chinese privet as the oldest for looking at the invasion history, and reported that stem density and percent cover could be predictor of explaining invasion success. They also claimed that single stem size can have higher explanatory power than multi-stemmed privets to predict the distribution.

Looking at the age of Chinese privet in relation to historical land use maps, sites that had been occupied the shortest time by privet (i.e., the most recently colonized) were often on agricultural and forested lands. Housing and road density in 1986 were the best predictors of privet age (Figure 3.13, Figure 3.14, Figure 3.15). This suggests that sites with urban and agriculture land use may have been more densely occupied by oldest Chinese privet than forested sites (Figure 3.14 and Figure 3.15). The reason of this could be that some urban sites which were historically used as agriculture may have been invaded by privet and colonization has increased over time. Determining how long invasive plants have occupied an area can facilitate some understanding of their long term effects to ecosystem and biodiversity and patterns of dispersal. For instance, Dietz and Schweingruber (2002) examined the growth rings of 60 invasive species representing 23 plant families in lower Michigan, U.S. to find growth rings of both native and non-native plants. They concluded that frequently disturbed areas such as meadows, and other ruderal sites were frequently occupied by older invasive plants, and semi-disturbed habitats were infested by young specimens. Pizarro et al. (2012) monitored the spread and invasion of glossy privet (*Ligustrum lucidum*) by using tree core analysis and mapping the distribution in the Sierras Chicas, Argentina. In this study, privet invasion and its relationship with urban development were examined, and it was concluded that glossy privet abundance was positively affected by expansion rates of urban land use and disturbance. They found that transects close to new developments and clusters of houses were highly disturbed and dominated by glossy privet. The results of these studies align with our results in which the oldest privets were analyzed and compared with historical land use of Auburn, AL.

Regression results also indicated that the age of Chinese privet had stronger relationships with housing density around the city center and suburban areas than agriculture and forested lands (Figure 3.13). Overlaying the 1976 LULC map of Auburn and the oldest Chinese privets of each riparian stream showed that 80% of study sites which were urban and agricultural lands were occupied by the oldest Chinese privets. These findings were similar to Mattingly and Orrock (2013) which examined the influence of historical land use on distribution of invasive Lespedeza plants in North Carolina, USA. They found that soil disturbance and historical land use change were highly associated with invasive species introduction and expansion in landscapes throughout North Carolina. Therefore, the relationship between historical land use and invasion of invasive shrubs can facilitate understanding the driving forces of non-native plant introduction and dispersion.

Much of the agricultural lands surrounding Auburn were abandoned in the late 1950s and reverted to forest or were developed to residential areas (Siebenthaler 2014). Based on overlaying study sites with 1976 and 2011 LULC map, it can be estimated that the oldest populations of Chinese privet could coincide with land use change for some of the study sites. Also, sites with agricultural land use in 1976 and old privet might be because most of the lands have been used for agriculture until 1950s (Siebenthaler 2014), and invasive plants could be introduced to some of these farmsteads to establish rapid growing vegetation. Also non-native species were cheap to establish and grow in the 1900s which increased their occupancy and colonization rates (Hanula et al. 2009). Therefore, it may be assumed that occupied agriculture lands might have contributed to the infestation by Chinese privet due to emerging of seeds and remnant species from the history of Auburn.

Not surprising, regression results showed a positive relationship between the oldest privet age and percent cover by old-growth form (Figure 3.16). It was noteworthy that all the other growth stages also had a significant and positive relationship between the oldest privet age and percent cover (3.11). Considering these results, it can be surmised that mature and old growth Chinese privet occur and proliferate together and lead to more seed dispersal and increasing colonization. This was consistent with patterns related to Chinese privet by Greene and Blossey (2014) who identified propagule pressure as an important factor in invasion success of this species. In this study, all growth stages had a positive relationship with estimated stand age, and it can be estimated that density of seedling and sapling forms can persist and increase under a canopy of mature and old growth forms. This supports other observations that Chinese privet is a shade tolerant species and adaptable to harsh conditions (Grove and Clarkson 2005, Hanula et al. 2009, Pizarro et al. 2012). The correlations between different growth stages in Chapter 2 indicated there was a positive correlation between mature and old growth forms of Chinese privet for all study sites. As privet gets older and colonization exceeds 20 years, it would be expected that the density of this species will increase substantially as younger growth stages persist under mature and old growth privets (Figure 3.16), and Chinese privet begins to dominate the riparian forest. The information provided here can be used to predict the trajectory of Chinese privet growth and cover in riparian areas over time. Our observations are consistent with Johnson et al. (2006) who reported that early successional habitats were more prone to be occupied by an increasing density of invasive shrubs over time due to expanding development and limited forest managements. Our results are also consistent with Greene and Blossey (2014) reported that there was a positive relationship between Chinese privet density and maximum DBH with privet and its mean percent cover.

This research highlights the potential relationships between historical land use, development, and colonization of Chinese privet in east Alabama. Important implications of this study are potential distribution and density of Chinese privet and specifically how disturbance associated with land use change has increased privet abundance in study sites. The negative effects of agriculture and urban land use might be a major threat that alters the ecosystem and its functions over time. Chinese privet has been established and dispersed more rapidly and abundantly within riparian areas adjacent to disturbed habitats. The results of this study may be beneficial for managers and land owners to determine initial colonization and spread of Chinese privet and create control points for management activities in further riparian areas of Auburn, AL. Also, this study can be also applied on similar riparian areas which have been under invasion of Chinese privet over time.

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Table 3.1: Results of historical and current residential development rates of total riparian streams within 200 m buffer zone (R.L.= road length, R.D.= road density, #of H.= number of houses, and H.D.= housing density).

Site #	SITES	1966				1986				2011			
		R. L (m)	R.D (m/ha)	#of H.	H. D (#/ha)	R.L (m)	R.D. (m/ha)	#of H.	H.D (#/ha)	R.L (m)	R. D. (m/ha)	#of H.	H.D (#/ha)
1	East Samford Avenue	456.0	43.0	0	0.0	568.0	53.6	0	0.0	456.0	43.0	1	0.1
2	Champions Blvrd.	96.3	14.0	0	0.0	86.7	12.6	0	0.0	326.2	47.5	1	0.1
3	S Donahue Dr	110.6	30.7	4	1.1	131.0	36.3	4	1.1	110.0	30.5	10	2.8
4	McMillan St	75.9	9.7	1	0.1	261.8	33.5	3	0.4	580.0	74.3	9	1.2
5	McMillan St-2	123.4	8.8	2	0.1	143.5	10.3	2	0.1	241.6	17.3	12	0.9
6	Kiesel Park	49.5	12.8	0	0.0	44.6	11.5	0	0.0	469.4	121.2	0	0.0
7	Waterstone Cir	15.2	2.2	0	0.0	13.7	2.0	3	0.4	527.8	75.6	57	8.2
8	Heywood St	166.6	13.8	1	0.1	260.1	21.6	2	0.2	456.6	37.9	51	4.2
9	Sam Harris park	272.6	31.1	0	0.0	245.4	28.0	0	0.0	133.0	15.2	12	1.4
10	Sanders Creek	214.9	51.1	8	1.9	330.6	78.6	13	3.1	293.8	69.9	18	4.3
11	Drake middle school	613.1	59.7	21	2.0	718.9	70.0	41	4.0	771.1	75.1	56	5.5
12	Drake2Westviewcemetry	332.1	42.2	11	1.4	368.0	46.8	13	1.7	265.7	33.8	15	1.9
13	Church	347.5	40.4	1	0.1	387.5	45.0	33	3.8	417.8	48.5	58	6.7
14	Town Creek Park	65.8	6.9	0	0.0	278.8	29.2	21	2.2	215.2	22.5	29	3.0
15	Cary woods creek	0.0	0.0	0	0.0	415.9	50.5	2	0.2	296.4	36.0	46	5.6
16	Camden Ridge Creek	0.0	0.0	0	0.0	14.6	1.4	1	0.1	610.9	60.3	117	11.6
17	Tuscany Village	177.8	29.6	0	0.0	151.7	25.2	0	0.0	186.2	30.9	24	4.0
18	Preserve Drive	122.8	13.3	0	0.0	79.2	8.6	0	0.0	322.6	35.0	20	2.2
19	Ellington Way	295.2	37.4	0	0.0	190.4	24.2	3	0.4	618.3	78.4	57	7.2
20	Watercrest Boulv.	100.2	9.0	0	0.0	64.6	5.8	0	0.0	275.1	24.7	18	1.6
21	Heath Road	218.1	21.4	0	0.0	140.7	13.8	0	0.0	133.7	13.1	1	0.1
22	Academy Drive	0.0	0.0	0	0.0	0.0	0.0	0	0.0	187.1	20.4	10	1.1
23	Mall Parkway	244.2	30.5	2	0.2	283.0	35.4	6	0.7	334.5	41.8	40	5.0

Table 3.1: Continue

24	Northridge St.	0.0	0.0	0	0.0	0.0	0.0	0	0.0	232.8	20.0	28	2.4
25	Chewacla Dr	436.5	46.4	48	5.1	281.6	29.9	62	6.6	365.4	38.8	82	8.7
26	EThachAve	379.5	37.9	28	2.8	244.8	24.4	35	3.5	290.9	29.0	53	5.3
27	E Samford Ave	98.2	14.0	13	1.9	233.8	33.5	17	2.4	222.2	31.8	27	3.9
28	Lee Road 72	104.0	13.3	0	0.0	138.0	17.6	3	0.4	238.0	30.4	8	1.0
29	Lee Road 72-2	0.0	0.0	0	0.0	0.0	0.0	0	0.0	48.5	8.4	4	0.7
30	East Farm. Rd	123.4	17.1	0	0.0	147.3	20.4	0	0.0	178.6	24.7	3	0.4
31	East Farm. Rd	106.6	8.7	0	0.0	124.4	10.2	0	0.0	235.8	19.3	9	0.7
32	Mimms Ln	50.8	8.7	3	0.5	59.3	10.1	3	0.5	436.3	74.5	9	1.5
33	Beechbrook Dr	0.0	0.0	0	0.0	81.6	11.8	1	0.1	128.8	18.6	10	1.4
34	Wooden Brid	139.0	13.8	1	0.1	96.5	9.6	7	0.7	104.8	10.4	12	1.2
35	Aberdeen Ln	23.4	2.9	1	0.1	27.3	3.4	3	0.4	120.9	15.2	26	3.3
36	Ogletree Rd	139.2	15.7	0	0.0	163.0	18.3	2	0.2	103.9	11.7	10	1.1
37	Chewacla State Park (1)	81.0	8.0	3	0.3	94.5	9.3	3	0.3	102.6	10.1	8	0.8
38	Chewacla State Park (2)	69.6	10.0	0	0.0	114.1	16.3	1	0.1	85.2	12.2	1	0.1
39	East University Dr	131.6	16.0	1	0.1	153.5	18.6	2	0.2	204.1	24.8	11	1.3
40	Stoneridge Drive	172.8	24.9	0	0.0	201.6	29.0	1	0.1	309.3	44.5	18	2.6
41	Duck Samford Park	124.8	22.6	0	0.0	145.6	26.4	1	0.2	227.1	41.1	5	0.9
42	Reynolds Drive	0.0	0.0	0	0.0	0.0	0.0	0	0.0	107.7	22.8	9	1.9
43	Fisheries 1	0.0	0.0	0	0.0	101.5	11.0	0	0.0	118.1	12.8	0	0.0
44	Fisheries 2	112.2	14.4	0	0.0	130.9	16.8	0	0.0	130.3	16.8	2	0.3
45	Fisheries 3	0.0	0.0	0	0.0	0.0	0.0	0	0.0	0.0	0.0	0	0.0
46	Town Creek 2	97.1	12.2	1	0.1	129.2	16.2	2	0.3	138.1	17.3	6	0.8
	Mean ± SE	141.0± 138.9	17.3± 15.7	3.3± 8.8	0.4± 1.0	170.6± 149.7	21.2± 18.1	6.3± 12.7	0.8± 1.4	268.7 ±170	34.5± 24.4	21.8± 24.7	2.6± 2.5

Table 3.2: Calculated change over 3 time periods and annual increase rates of total study streams within the 200 m buffer (R.L.= road length, R.D.= road density, #of H.= number of houses, and H.D.= housing density).

		1966-2011				1966-1986				1986-2011			
	SITES	**R. L (m)**	**R.D (m/ha)**	**#of H.**	**H. D (#/ha)**	**R. L (m)**	**R.D (m/ha)**	**#of H.**	**H. D (#/ha)**	**R. L (m)**	**R.D (m/ha)**	**#of H.**	**H. D (#/ha)**
1	East Samford Avenue	0.0	0.0	1	0.0	112.0	10.6	0	0.0	-112.0	43.0	1	0.0
2	Champions Blvrd	229.9	33.5	1	0.1	-9.6	-1.4	0	0.0	239.5	47.5	1	0.1
3	S Donahue Dr	-0.6	-0.2	6	0.0	20.3	5.6	0	0.0	-21.0	29.4	6	0.0
4	McMillan St	504.1	64.6	8	0.6	185.9	23.8	2	0.3	318.2	74.1	6	0.3
5	McMillan St-2	118.2	8.5	10	0.6	20.0	1.4	0	0.0	98.2	17.2	10	0.6
6	Kiesel Park	419.9	108.4	0	0.0	-5.0	-1.3	0	0.0	424.8	121.2	0	0.0
7	Waterstone Cir	512.6	73.5	57	4.0	-1.5	-0.2	3	0.4	514.1	75.6	54	3.6
8	Heywood St	290.0	24.1	50	2.1	93.5	7.8	1	0.1	196.5	37.8	49	2.0
9	Sam Harris park	-139.6	-15.9	12	1.3	-27.3	-3.1	0	0.0	-112.4	15.2	12	1.3
10	Sanders Creek	78.9	18.8	10	0.0	115.7	27.5	5	1.2	-36.7	68.0	5	-1.2
11	Drake middle school	158.0	15.4	35	1.5	105.8	10.3	20	1.9	52.3	73.1	15	-0.5
12	Drake2	-66.4	-8.4	4	-0.5	35.9	4.6	2	0.3	-102.3	32.4	2	-0.7
13	Church	70.3	8.2	57	1.7	40.0	4.6	32	3.7	30.3	48.4	25	-2.0
14	Town Creek Park	149.4	15.6	29	1.7	213.1	22.3	21	2.2	-63.7	22.5	8	-0.5
15	Cary woods creek	296.4	36.0	46	2.1	415.9	50.5	2	0.2	-119.5	36.0	44	1.8
16	Camden Ridge Creek	610.9	60.3	117	6.5	14.6	1.4	1	0.1	596.3	60.3	116	6.4
17	Tuscany Village	8.3	1.4	24	0.9	-26.2	-4.3	0	0.0	34.5	30.9	24	0.9
18	Preserve Drive	199.8	21.6	20	1.6	-43.6	-4.7	0	0.0	243.4	35.0	20	1.6
19	Ellington Way	323.1	41.0	57	1.7	-104.7	-13.3	3	0.4	427.8	78.4	54	1.4
20	Watercrest Boulv.	174.9	15.7	18	1.0	-35.5	-3.2	0	0.0	210.5	24.7	18	1.0
21	Heath Road	-84.4	-8.3	1	0.1	-77.4	-7.6	0	0.0	-7.0	13.1	1	0.1
22	Academy Drive	187.1	20.4	10	0.8	0.0	0.0	0	0.0	187.1	20.4	10	0.8
23	Mall Parkway	90.3	11.3	38	3.0	38.8	4.9	4	0.5	51.5	41.6	34	2.5
24	Northridge St	232.8	20.0	28	1.1	0.0	0.0	0	0.0	232.8	20.0	28	1.1
25	Chewacla Dr	-71.1	-7.6	34	1.8	-154.9	-16.4	14	1.5	83.8	33.7	20	0.3

Table 3.2: Continue

26	East Thach Ave	-88.5	-8.8	25	1.4	-134.6	-13.4	7	0.7	46.1	26.2	18	0.7
27	E Samford Ave	123.9	17.7	14	0.5	135.6	19.4	4	0.6	-11.7	29.9	10	-0.1
28	Lee Road 72	134.1	17.1	8	0.4	34.0	4.3	3	0.4	100.0	30.4	5	0.0
29	Lee Road 72-2	48.5	8.4	4	0.3	0.0	0.0	0	0.0	48.5	8.4	4	0.3
30	East Farm Road	55.2	7.6	3	0.3	24.0	3.3	0	0.0	31.3	24.7	3	0.3
31	East Farm. Rd	129.2	10.6	9	0.5	17.8	1.5	0	0.0	111.4	19.3	9	0.5
32	Mimms Ln	385.5	65.8	6	-0.1	8.5	1.4	0	0.0	377.0	74.0	6	-0.1
33	Beechbrook Dr	128.8	18.6	10	1.5	81.6	11.8	1	0.1	47.2	18.6	9	1.4
34	Wooden Bridge	-34.2	-3.4	11	0.8	-42.4	-4.2	6	0.6	8.3	10.3	5	0.2
35	Aberdeen Ln	97.5	12.2	25	2.4	3.9	0.5	2	0.3	93.6	15.1	23	2.1
36	Ogletree Rd	-35.3	-4.0	10	0.7	23.8	2.7	2	0.2	-59.1	11.7	8	0.5
37	Chewacla State Park (1)	21.6	2.1	5	-0.2	13.5	1.3	0	0.0	8.1	9.8	5	-0.2
38	Chewacla State Park (2)	15.7	2.2	1	0.1	44.6	6.4	1	0.1	-28.9	12.2	0	-0.1
39	East University Dr.	72.5	8.8	10	0.5	21.9	2.7	1	0.1	50.6	24.6	9	0.4
40	Stoneridge Dr.	136.4	19.6	18	1.6	28.8	4.1	1	0.1	107.6	44.5	17	1.5
41	Duck Samford Park	102.2	18.5	5	0.4	20.8	3.8	1	0.2	81.4	41.1	4	0.2
42	Reynolds Drive	107.7	22.8	9	0.8	0.0	0.0	0	0.0	107.7	22.8	9	0.8
43	Fisheries 1	118.1	12.8	0	0.0	101.5	11.0	0	0.0	16.7	12.8	0	0.0
44	Fisheries 2	18.1	2.3	2	0.1	18.7	2.4	0	0.0	-0.6	16.8	2	0.1
45	Fisheries 3	0.0	0.0	0	0.0	0.0	0.0	0	0.0	0.0	0.0	0	0.0
46	Town Creek 2	41.0	5.2	5	0.1	32.1	-12.1	1	0.1	8.9	17.2	4	0.0
Mean ± SE		127.6± 145.2	17.3± 24.2	18.5± 22.2	1.1± 1.2	29.6± 75.2	16.9± 15.2	3.1±6 .4	0.4± 0.7	98.1± 120.3	34.1± 24.3	15.5± 20.8	1.1± 1.2
Mean Annual Increase Rate ± SE		2.8± 3.2	0.4± 0.5	0.4± 0.5	0.02±0 .03	1.5± 3.8	0.8± 0.8	0.2±0 .3	0.02±0 .03	3.9± 4.8	1.3± 0.9	0.6± 0.8	0.03±0 .04

Table 3.3: The results of application of age equation on the oldest Chinese privet data set (DBH: Diameter at Breast Height, 1.40 m).

Site #	**SITES**	**DBH (cm)**	**Oldest Privet (years)**
1	East Samford Avenue	10.5	17.0
2	Champions Blvrd	10.1	16.3
3	South Donahue	11	17.7
4	McMillan	11	17.7
5	McMillan-2	10.6	17.2
6	Kiesel Park	12.7	19.9
7	Waterstone Circle	10.2	16.6
8	Heywood	11	17.7
9	Sam Harris Park	11.2	18.0
10	Sanders Creek	11.5	18.4
11	Drake Middle School	11	17.7
12	Drake 2	10.2	16.6
13	Church	10.5	17.0
14	Town Creek Park	15.5	23.4
15	Cary Woods	10.5	17.0
16	Camden Ridge	11.5	18.4
17	Tuscany Village	11.2	18.0
18	Preserve Dr.	10	16.3
19	Ellington Way	11.4	18.2
20	Watercrest Dr.	12.2	19.3
21	Health Road	11	17.7
22	Academy Drive	10.5	17.0
23	Mall Parkway	10.6	17.2
24	Northridge	10	16.3
25	Chewacla Dr.	5.2	9.2
26	East Thach Avenue	12.5	19.7
27	East Samford	10.7	17.3
28	Lee Road 72	7.8	13.2
29	Lee Road	11.5	18.4
30	East Farmville Rd	5.2	9.2
31	East Farmville	5.6	9.9
32	Mimms Ln.	12.2	19.3
33	Beech Brook Dr.	12.8	20.1
34	Wooden bridge	8.6	14.4
35	Aberdeen Ln.	11.8	18.8
36	Ogletree Road	11.6	18.5
37	Chewacla 1	7.5	12.8
38	Chewacla 2	6.2	10.8
39	East University Dr.	12.6	19.8
40	Stone Ridge	10.5	17.0
41	Duck Samford Park	10	16.3
42	Reynolds Dr.	10.75	17.4
43	Fisheries 1	4.2	7.7
44	Fisheries 2	6.5	11.3
45	Fisheries 3	3.7	6.9
46	Town Creek 2	13.5	21.0
	Mean ± SE	10.1±2.5	16.4±3.6

Table 3.4: Results of regression analyses for relationship between cover % of Chinese privet, oldest Chinese privet, housing density (houses/ha) and road density (m/ha) for total study sites.

Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr > |t|
Total cover % (n=46)					
Housing Density 2011 (houses/ha)	0.271	0.683**	0.342	1.997	0.019
Housing Density 1986 (houses/ha)	0.327	4.710*	2.982	1.579	0.069
Housing Density 1966 (houses/ha)	0.195	5.235	6.001	0.872	0.304
Total cover % (n=46)					
Road Density 2011 (m/ha)	0.062	0.037	0.053	0.698	0.384
Road Density 1986 (m/ha)	0.120	0.239*	0.205	1.166	0.082
Road Density 1966 (m/ha)	0.089	0.204	0.748	0.273	0.631
Oldest Chinese Privet (n=46)					
Housing Density 2011 (houses/ha)	0.194	0.617**	0.245	2.518	0.004
Housing Density 1986 (houses/ha)	0.241	1.355**	0.776	1.746	0.032
Housing Density 1966 (houses/ha)	0.169	1.341	1.984	0.676	0.318
Oldest Chinese Privet (n=46)					
Road Density 2011 (m/ha)	0.095	0.041	0.104	0.389	0.582
Road Density 1986 (m/ha)	0.148	0.085*	0.073	1.163	0.072
Road Density 1966 (m/ha)	0.084	0.068	0.109	0.624	0.514

**significant at $p<0.05$ *significant at $p<0.10$.

Table 3.5: Results of regression analyses for relationship between cover %, oldest privet age, and growth forms of Chinese privet for total study sites.

Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr > |t|
Seedling					
Oldest C. Privet	0.213	0.062**	0.169	3.091	0.003
Sapling					
Oldest C. Privet	0.271	0.010**	0.136	3.466	0.001
Mature					
Oldest C. Privet	0.314	0.151**	0.219	4.289	0.001
Old Growth					
Oldest C. Privet	0.321	0.321**	0.074	4.315	0.001
Cover (%)					
Oldest C. Privet	0.221	0.041**	0.429	3.863	0.001

**significant at $p<0.05$.

Table 3.6: Results of regression analyses for relationship between cover % of Chinese privet in sites designated as urban, forest, and agricultural land use (based on LULC maps) and housing density/ road density.

Total Cover % of Urban Sites					
Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr>|t|
Housing Density 2011 (houses/ha) N=39	0.52	3.030**	0.460	6.590	<.0001
Housing Density1986 (houses/ha) N=15	0.56	6.190**	1.483	4.280	0.001
Housing Density1966 (houses/ha) N=9	0.24	4.710*	2.196	1.900	0.089
Road Density2011 (m/ha) N=39	0.21	0.224**	0.072	3.110	0.004
Road Density1986 (m/ha) N=15	0.24	0.291*	0.165	1.850	0.090
Road Density1966 (m/ha) N=9	0.22	0.227*	0.121	1.920	0.089
Total Cover % of Forested Sites					
Housing Density1986 (houses/ha) N=11	0.25	7.588*	4.003	1.960	0.078
Housing Density1966 (houses/ha) N=13	0.27	5.771*	3.027	1.870	0.091
Road Density1986 (m/ha) N=11	0.15	0.273	0.208	1.340	0.211
Road Density1966 (m/ha) N=13	0.13	0.431	0.350	1.250	0.238
Total Cover % of Agricultural Sites					
Housing Density1986 (houses/ha) N=20	0.28	5.061**	2.011	2.170	0.044
Housing Density1966 (houses/ha) N=24	0.32	17.596**	6.291	2.870	0.010
Road Density1986 (m/ha) N=20	0.21	0.596	0.262	2.320	0.032
Road Density1966 (m/ha) N=24	0.27	0.622	0.243	2.540	0.021

*Significant at $p<0.10$, **Significant at $p<0.05$.

Table 3.7: Results of regression analyses for relationship between oldest Chinese privet age in sites designated as urban and agricultural land use (based on LULC maps) and housing density/ road density.

Oldest Chinese Privet-Urban Sites					
Variable	R-Square (R^2)	Parameter Estimate	Standard Error	t Value	Pr>|t|
Housing Density 2011 (houses/ha) N=39	0.28	0.668**	0.184	3.099	0.000
Housing Density1986 (houses/ha) N=15	0.43	1.179**	0.391	2.930	0.013
Housing Density1966 (houses/ha) N=9	0.25	1.051*	0.515	2.040	0.064
Road Density2011 (m/ha) N=39	0.19	0.072**	0.024	2.890	0.006
Road Density1986 (m/ha) N=15	0.26	0.069*	0.032	2.160	0.052
Road Density1966 (m/ha) N=9	0.20	0.077	0.043	1.780	0.101
Oldest Chinese Privet- Agricultural Sites					
Housing Density1986 (houses/ha) N=20	0.22	4.055*	2.013	1.900	0.074
Housing Density1966 (houses/ha) N=24	0.14	1.711*	0.872	1.961	0.086
Road Density1986 (m/ha) N=20	0.21	0.217*	0.106	2.040	0.057
Road Density1966 (m/ha) N=24	0.13	0.148	0.089	1.700	0.107

*Significant at p<0.10, **Significant at p<0.05.

Table 3.8: The mean (±SE) % cover of Chinese privet in 2014 based on land use change categories between years (F→F is forest to forest, A→A is agriculture to agriculture, U→U is urban to urban, F→U is forest to urban, and A→U is agriculture to urban land use change for (total n=46).

Years	**F→F**	**A→A**	**U→U**	**F→U**	**A→U**
1966-2011	23.8±18.4 (n=4)	18.6±0.9 (n=3)	30.4±12.3 (n=7)	22.5±9.1 (n=10)	33.6±11.8 (n=22)
1966-1986	22.2±11.2 (n=11)	28.4±10.1 (n=20)	30.4±12.3 (n=9)	21.6±9.2 (n=3)	30.5±8.4 (n=3)
1986-2011	23.8±18.4 (n=4)	18.6±0.9 (n=3)	33.4±13.2 (n=13)	29.2±9.5 (n=9)	44.9±16.4 (n=17)

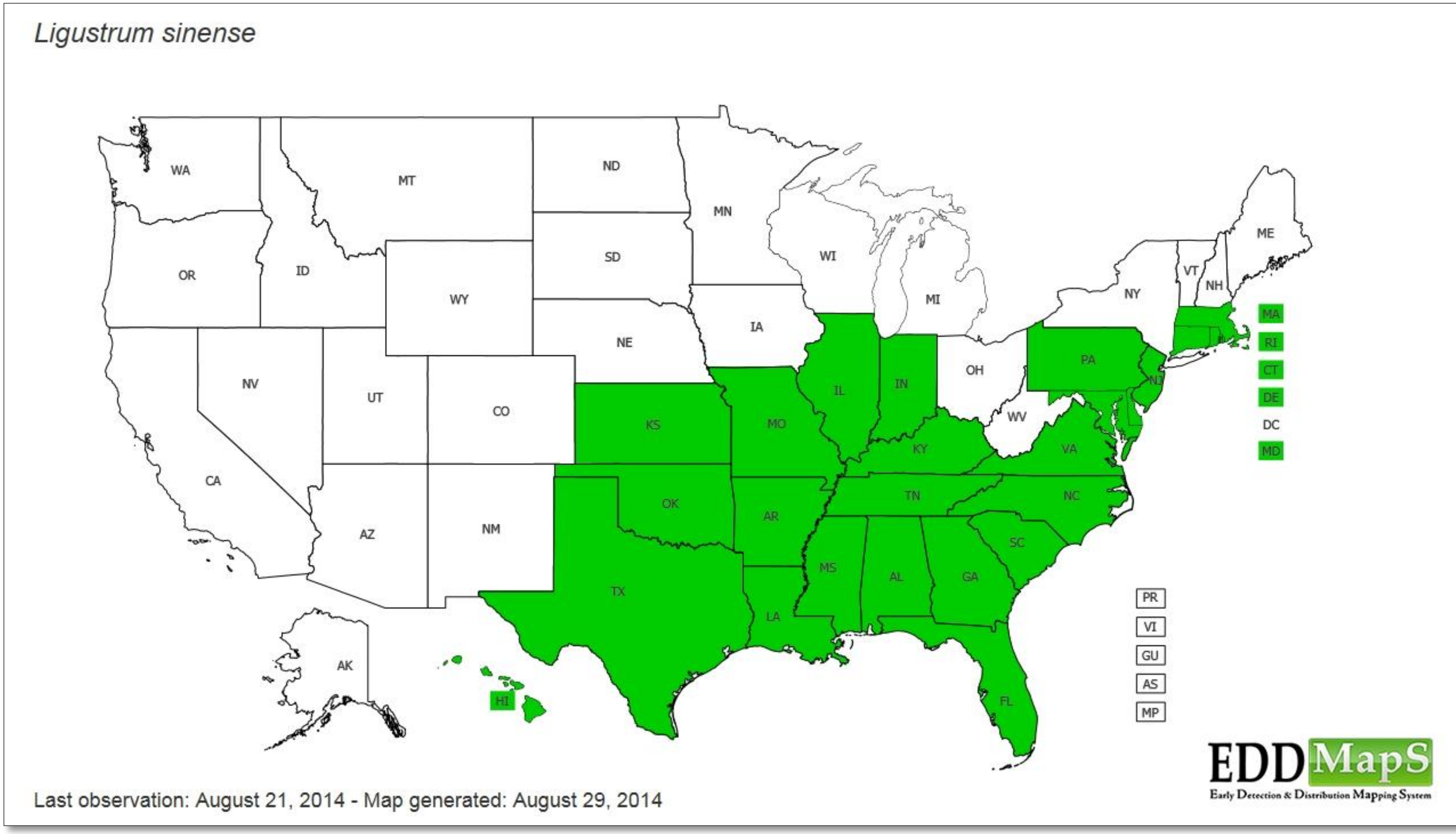

Figure 3.1: The spread of Chinese privet (*Ligustrum sinense*) in southern United States (www.eddmaps.org/distribution/usstate).

LOCATIONS OF STUDY SITES AND AERIAL PHOTOGRAPHY OF AUBURN (2013)

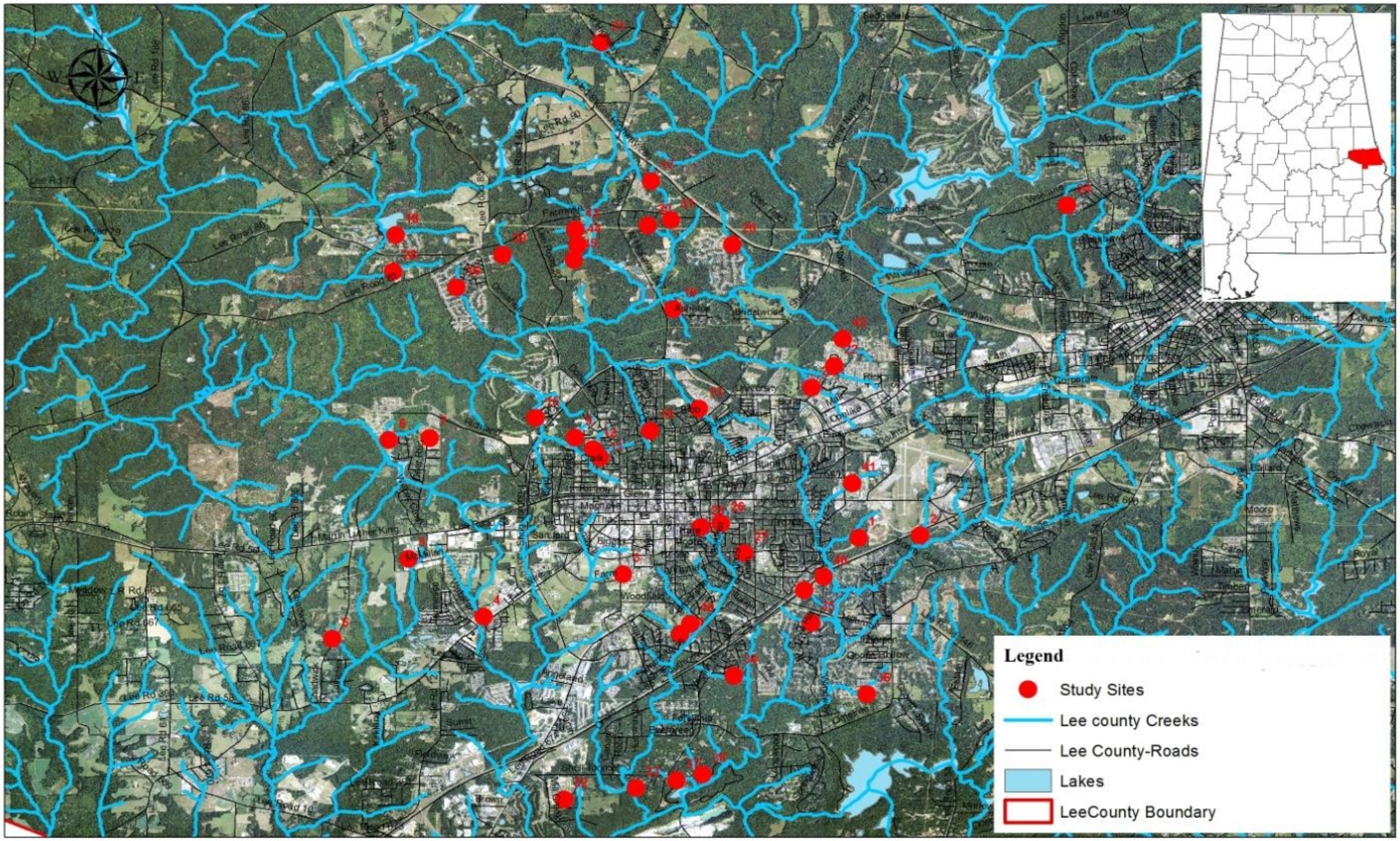

Figure 2.2: Map of 46 riparian study sites surveyed for invasive shrub species.


Figure 3.2: Infestation of Chinese privet in riparian area in Auburn, Alabama.


Figure 3.3: Flowering and invasion of Chinese privet in a study site.

LAND USE/LAND COVER MAP OF AUBURN (1976)

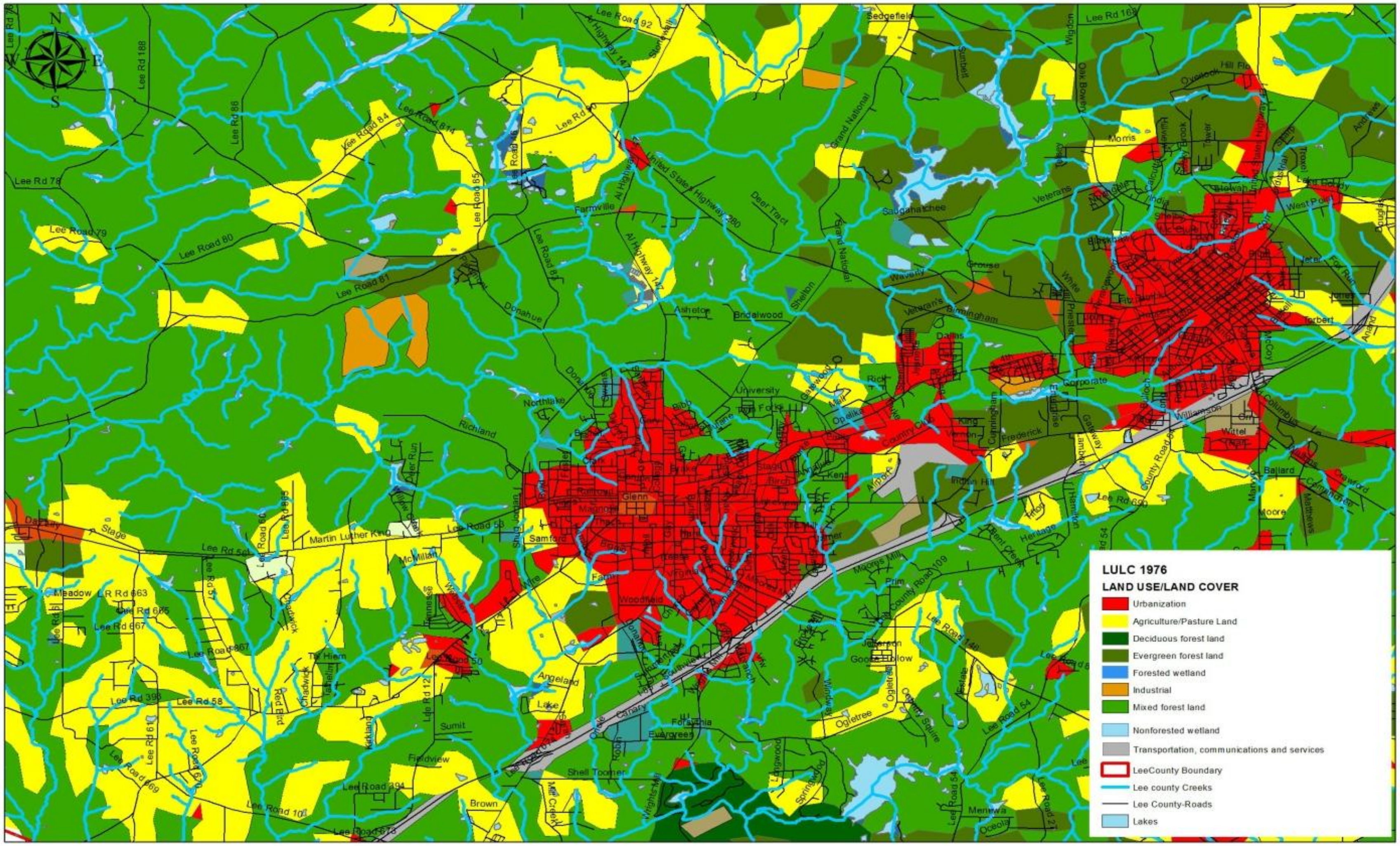

Figure 3.4: Historical LULC map of Auburn (1976).

LAND USE/LAND COVER MAP OF AUBURN (2011)

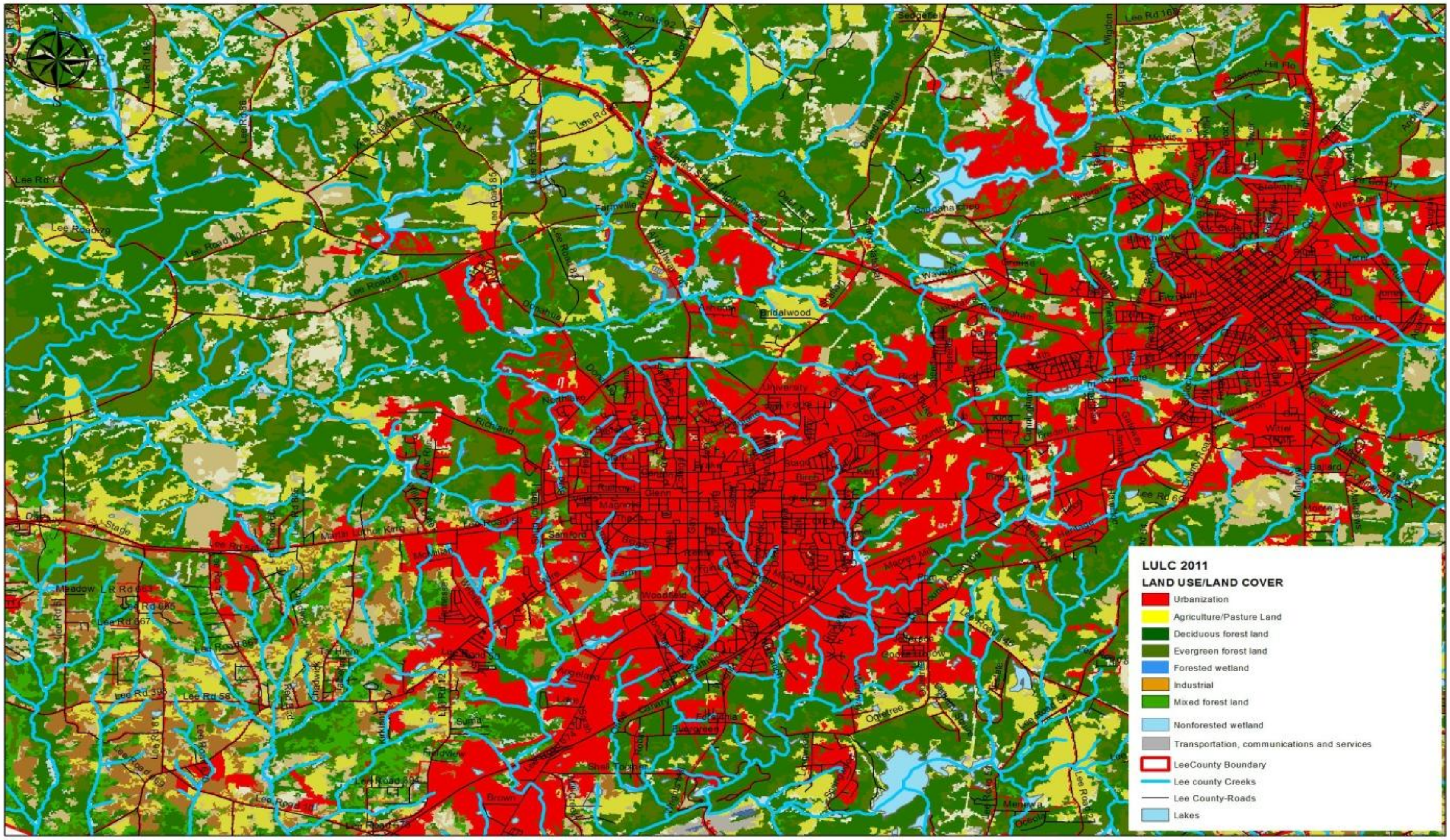

Figure 3.5: LULC map of Auburn (2011).

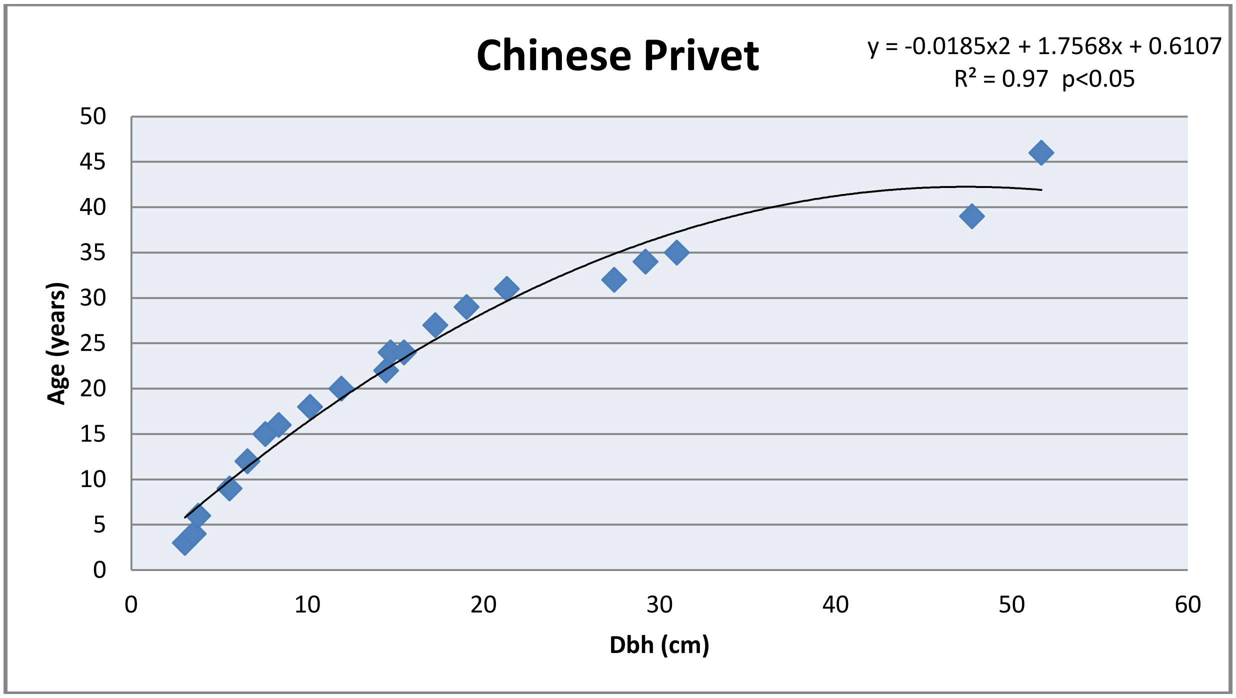

Figure 3.6: DBH-Age relationship of 20 Chinese privet samples derived from Town Creek Park, Auburn, AL.

DISTRIBUTION OF OLDEST CHINESE PRIVET AND LULC MAP OF AUBURN (1976)

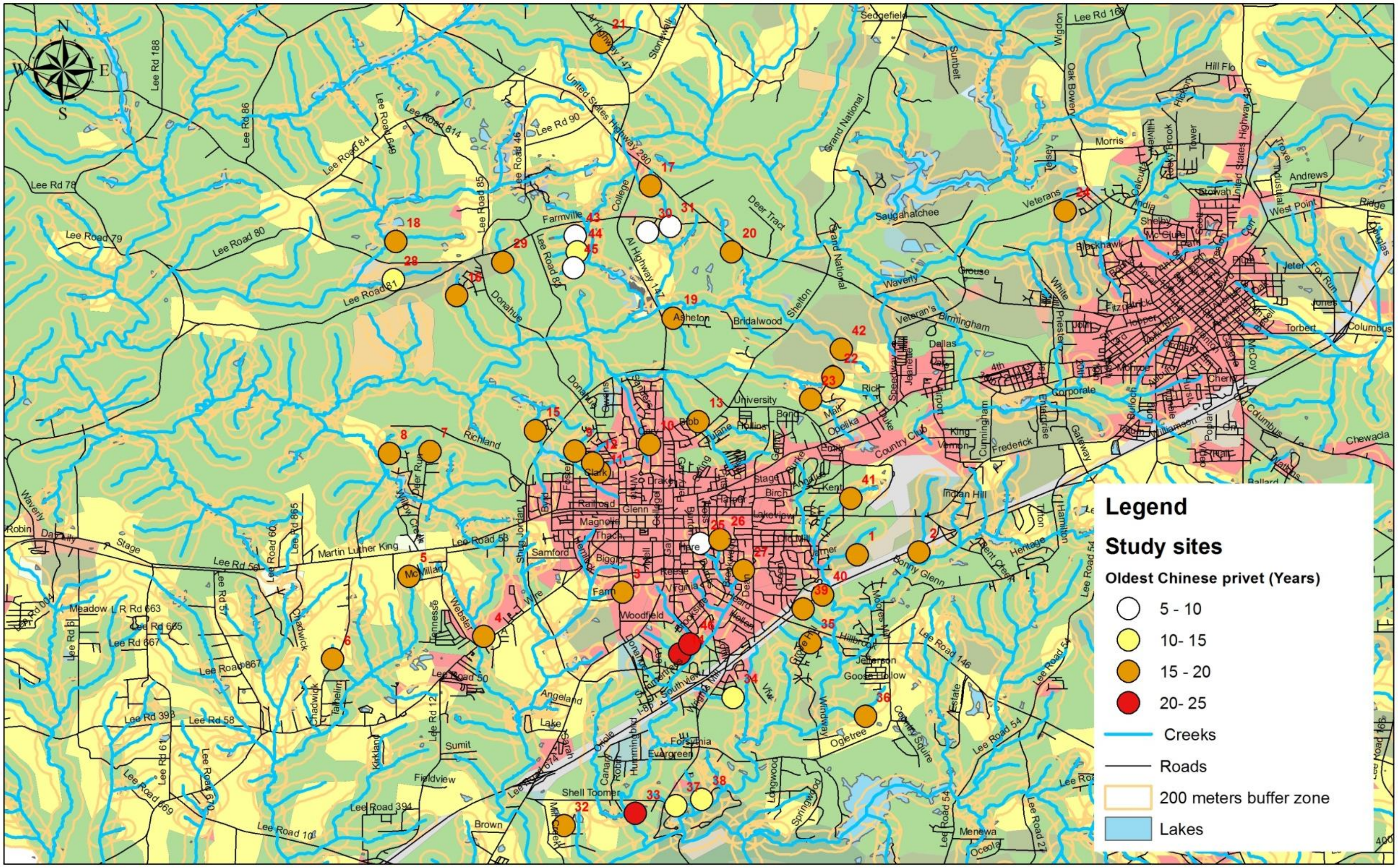

Figure 3.7: Distribution of oldest Chinese privet calculated for total study sites on 1976 LULC map of Auburn, AL. (Pink= Urban, Light green= Forest, and Yellow= Agriculture).

DISTRIBUTION OF OLDEST CHINESE PRIVET AND LULC MAP OF AUBURN (2011)

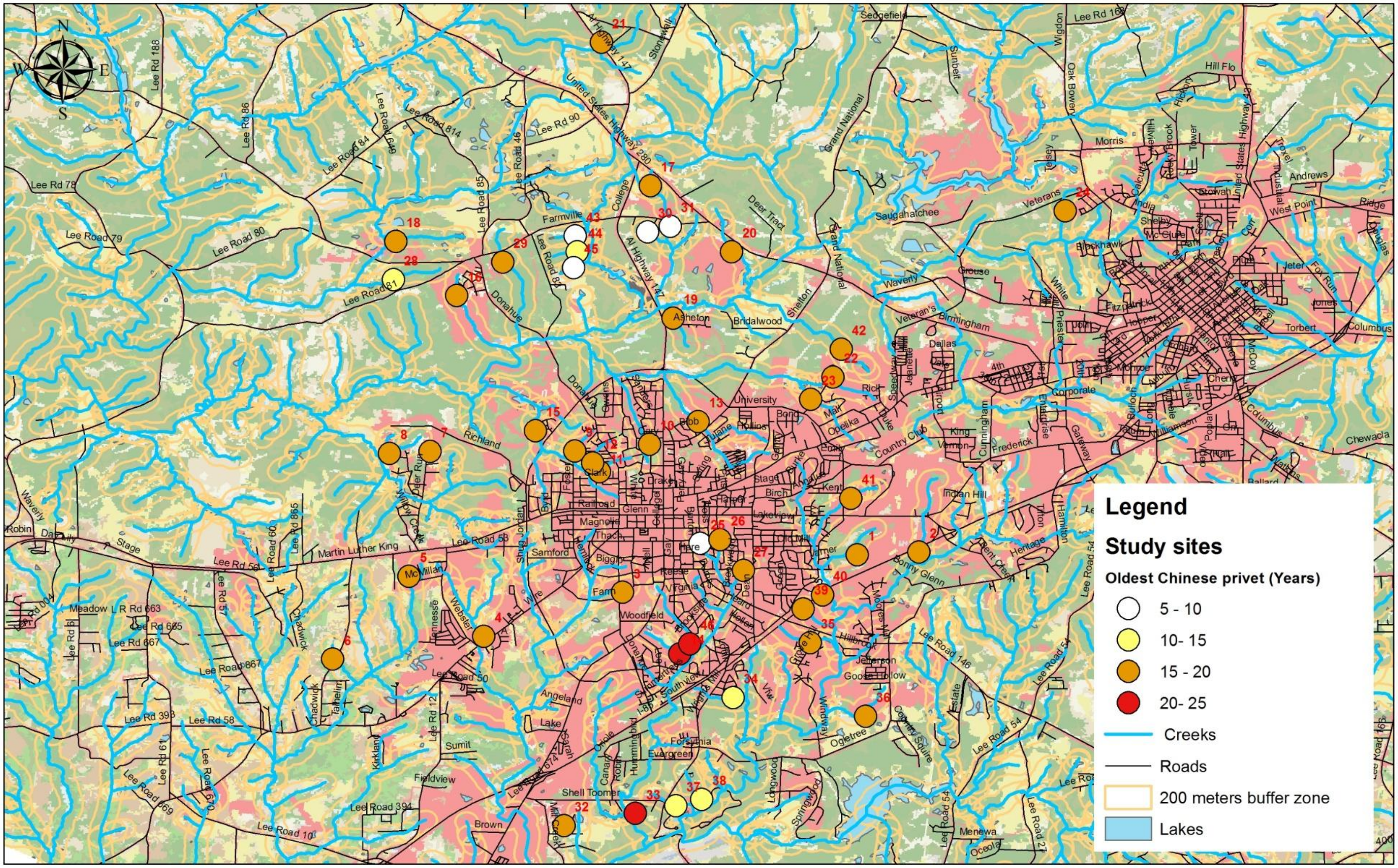

Figure 3.8: Distribution of oldest Chinese privet calculated for total study sites on 2011 LULC map of Auburn, AL. (Pink= Urban, Light green= Forest, and Yellow= Agriculture).

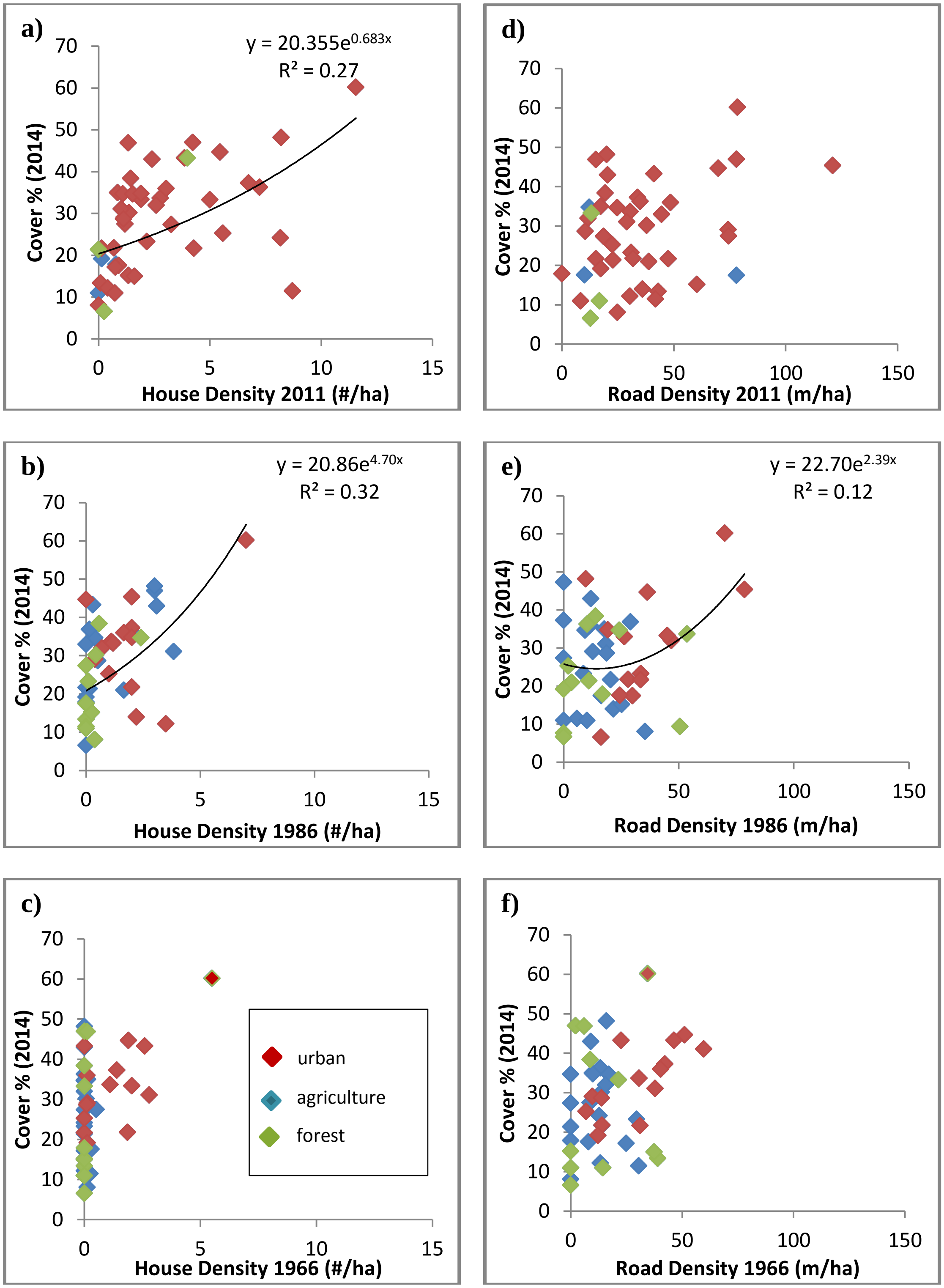

Figure 3.9 : The relationship between a) cover % and 2011 house density ($p<0.05$), b) cover % and 1986 house density ($p<0.10$), c) cover % and 1966 house density, d) cover % and 2011 road density, e) cover % and 1986 road density ($p<0.10$), f) cover % and 1966 road density for Chinese privet.

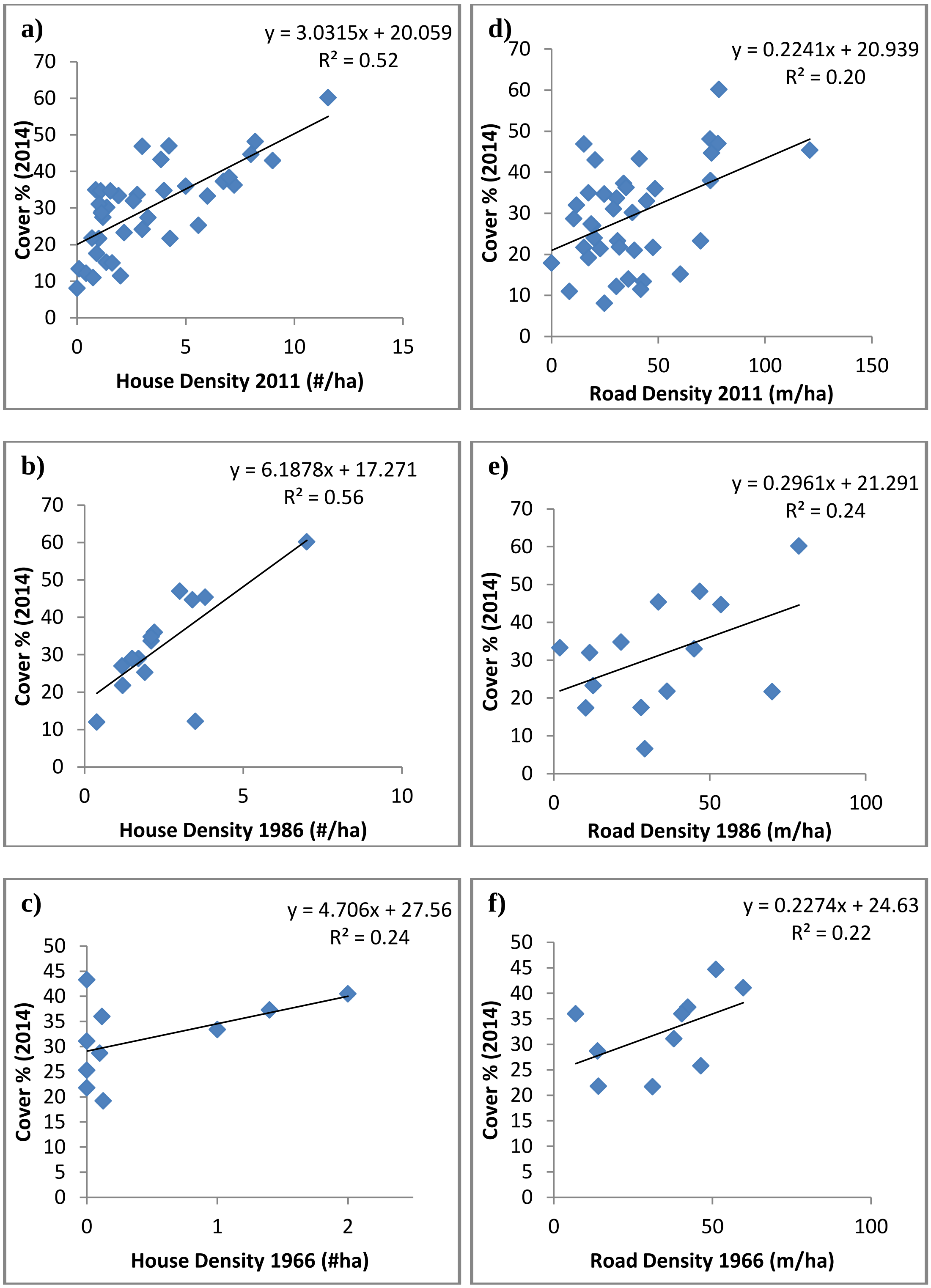

Figure 3.10 : For all sites designated urban land use in 1966, 1986 and 2011, the relationship between a) privet cover % and 2011 house density ($p<0.05$), b) privet cover % and 1986 house density ($p<0.05$), c) privet cover % and 1966 house density ($p<0.10$), d) privet cover % and 2011 road density ($p<0.05$), e) privet cover % and 1986 road density ($p<0.10$), and f) privet cover % and 1966 road density ($p<0.10$).

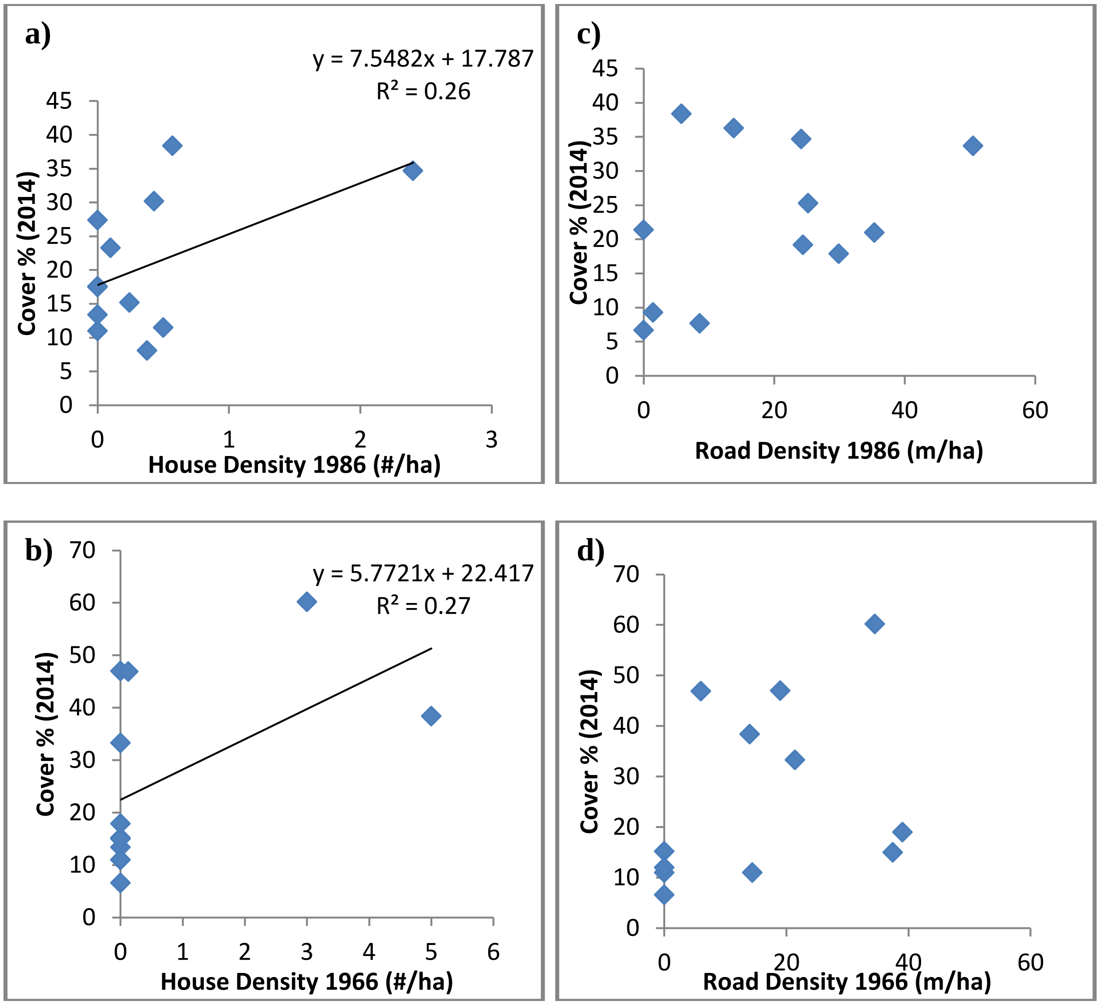

Figure 3.11 : For all sites designated forested in 1966 and 1986, the relationship between a) privet cover % and 1986 house density ($p<0.10$), b) privet cover % and 1966 house density ($p<0.10$), c) privet cover % and 1986 road density, and d) privet cover % and 1966 road density.

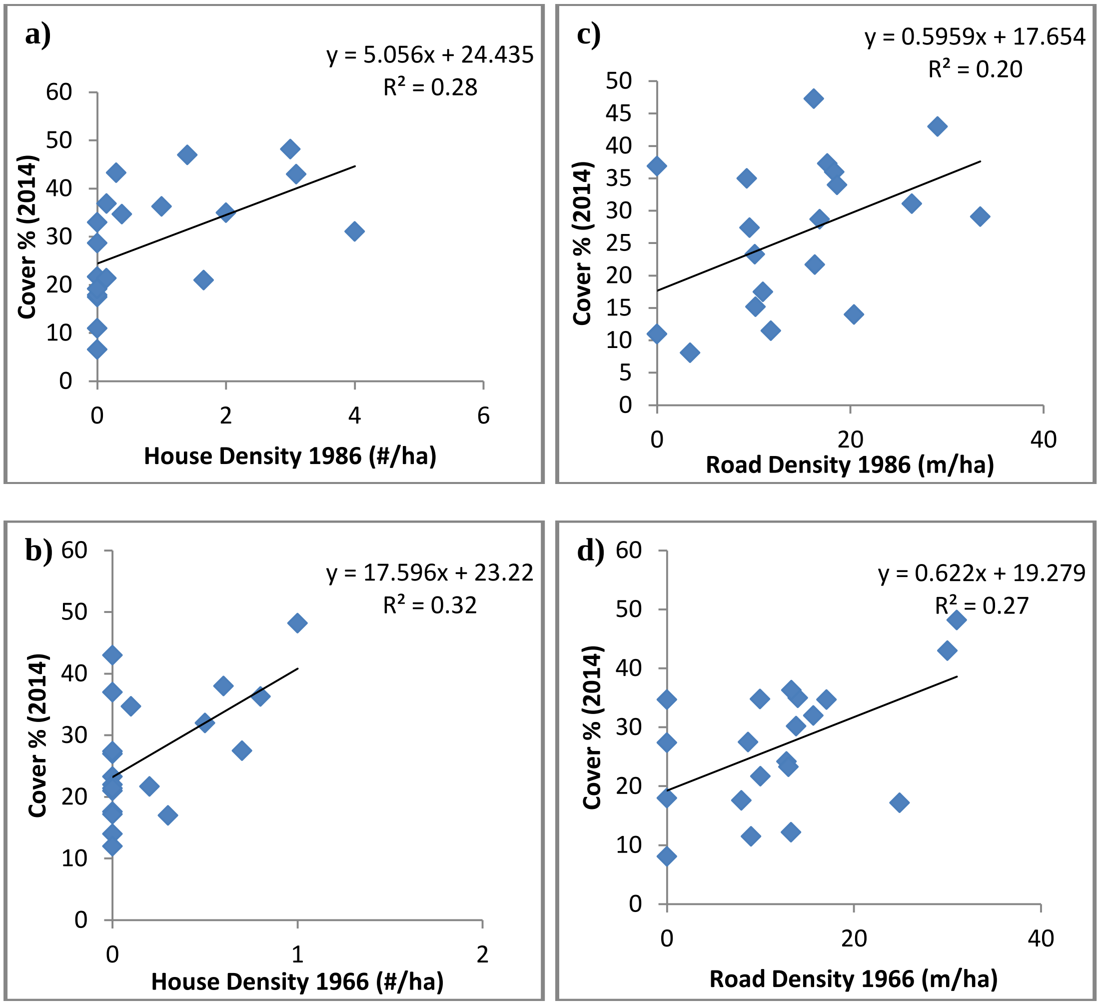

Figure 3.12 : For all sites designated agriculture land use in 1966, 1986 and 2011, the relationship between a) privet cover % and 1986 house density ($p<0.05$), b) privet cover % and 1966 house density ($p<0.05$), c) privet cover % and 1986 road density ($p<0.05$), d) privet cover % and 1966 road density ($p<0.05$).

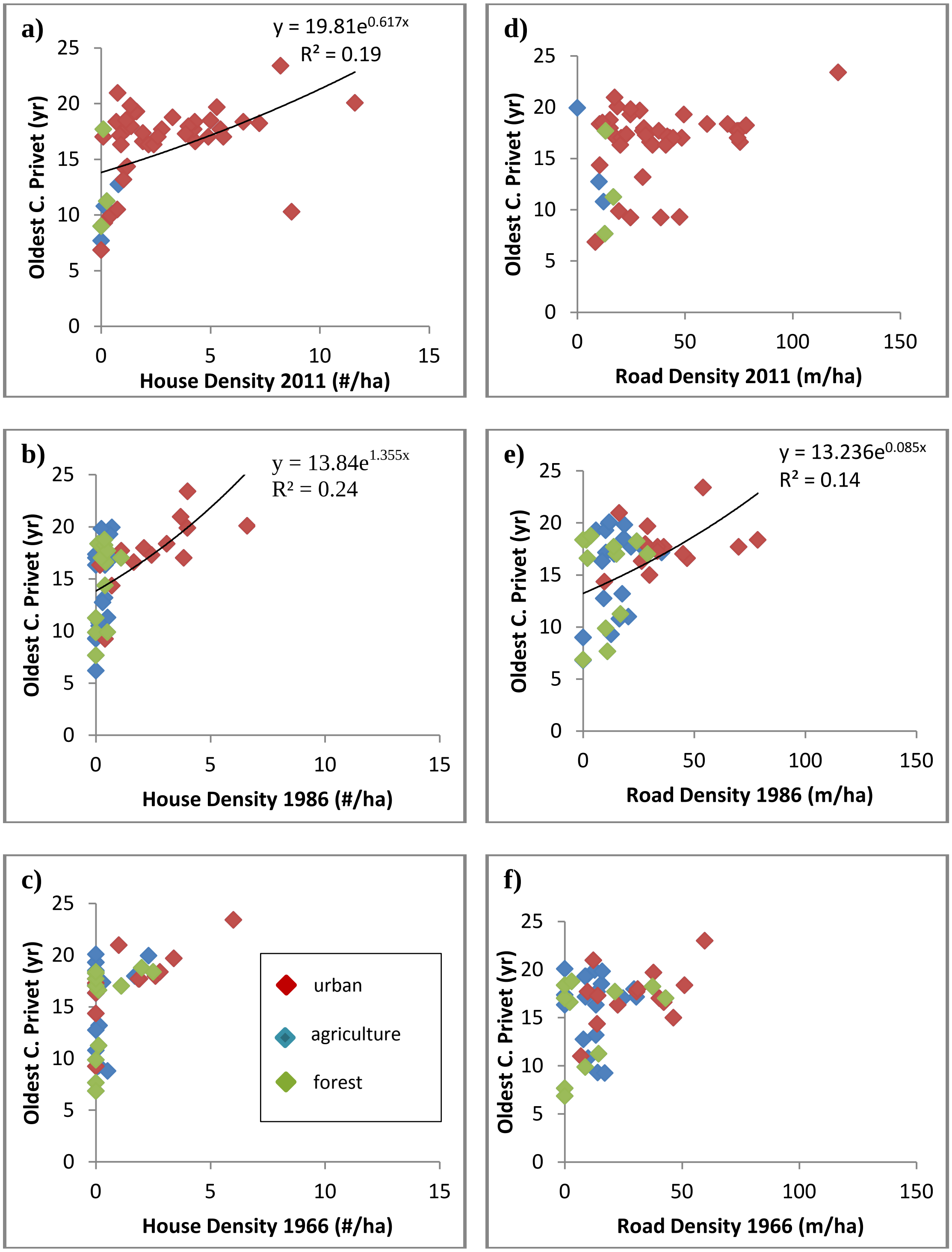

Figure 3.13 : The relationship between a) privet stand age and 2011 house density ($p<0.05$), b) privet stand age and 1986 house density ($p<0.05$), c) privet stand age and 1966 house density, d) privet stand age and 2011 road density, e) privet stand age and 1986 road density ($p<0.10$), f) privet stand age and 1966 road density for Chinese privet (oldest privet were calculated based on 2014 DBH).

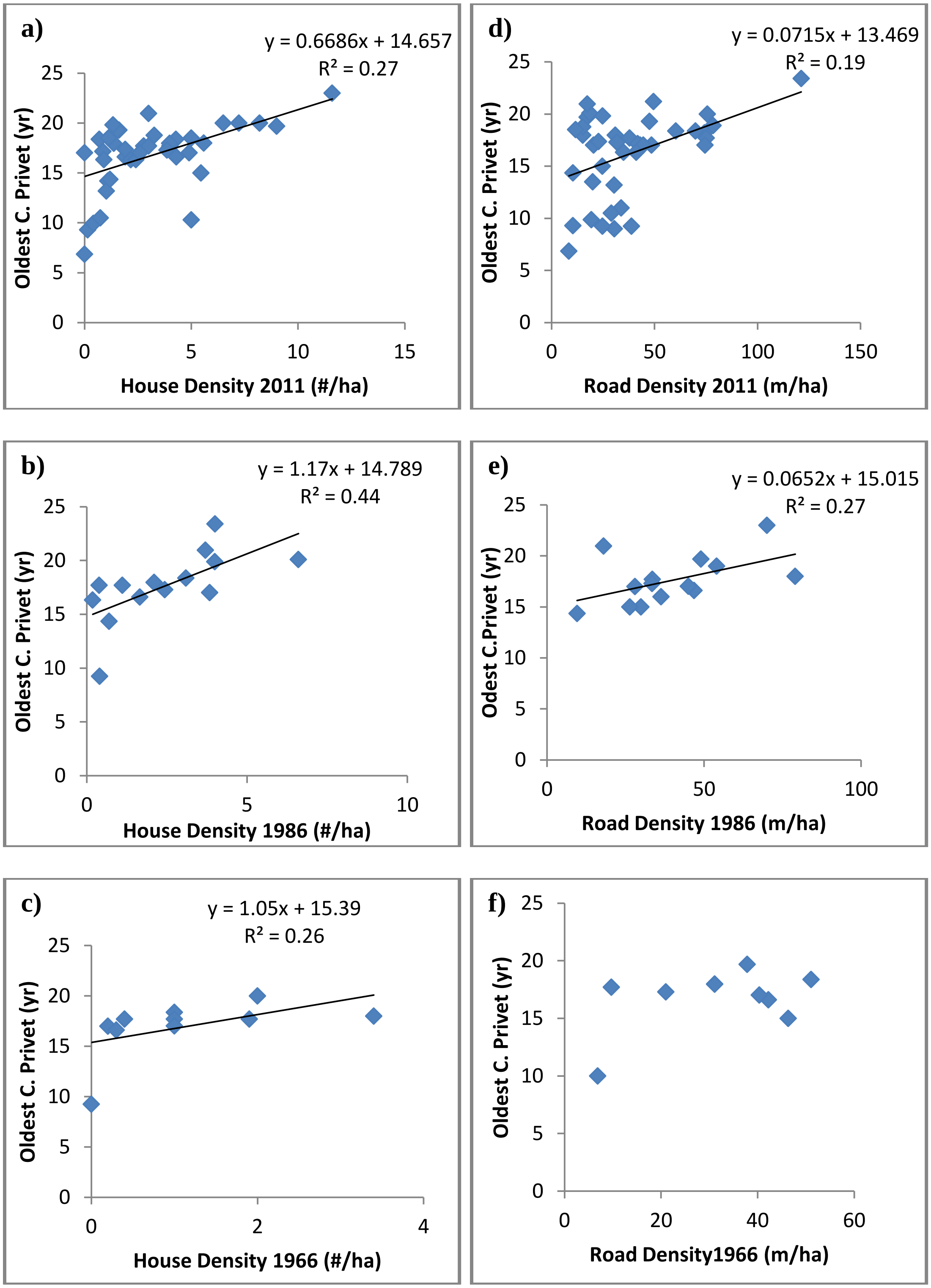

Figure 3.14 : For all sites designated urban land use in 1966, 1986 and 2011, the relationship between a) privet stand age and 2011 house density ($p<0.05$), b) privet stand age and 1986 house density ($p<0.05$), c) privet stand age and 1966 house density ($p<0.10$), d) privet stand age and 2011 road density ($p<0.05$), e) privet stand age and 1986 road density ($p<0.10$) and f) privet

stand age and 1966 road density for Chinese privet in urban lands. Stand ages were calculated based on 2014 DBH data.

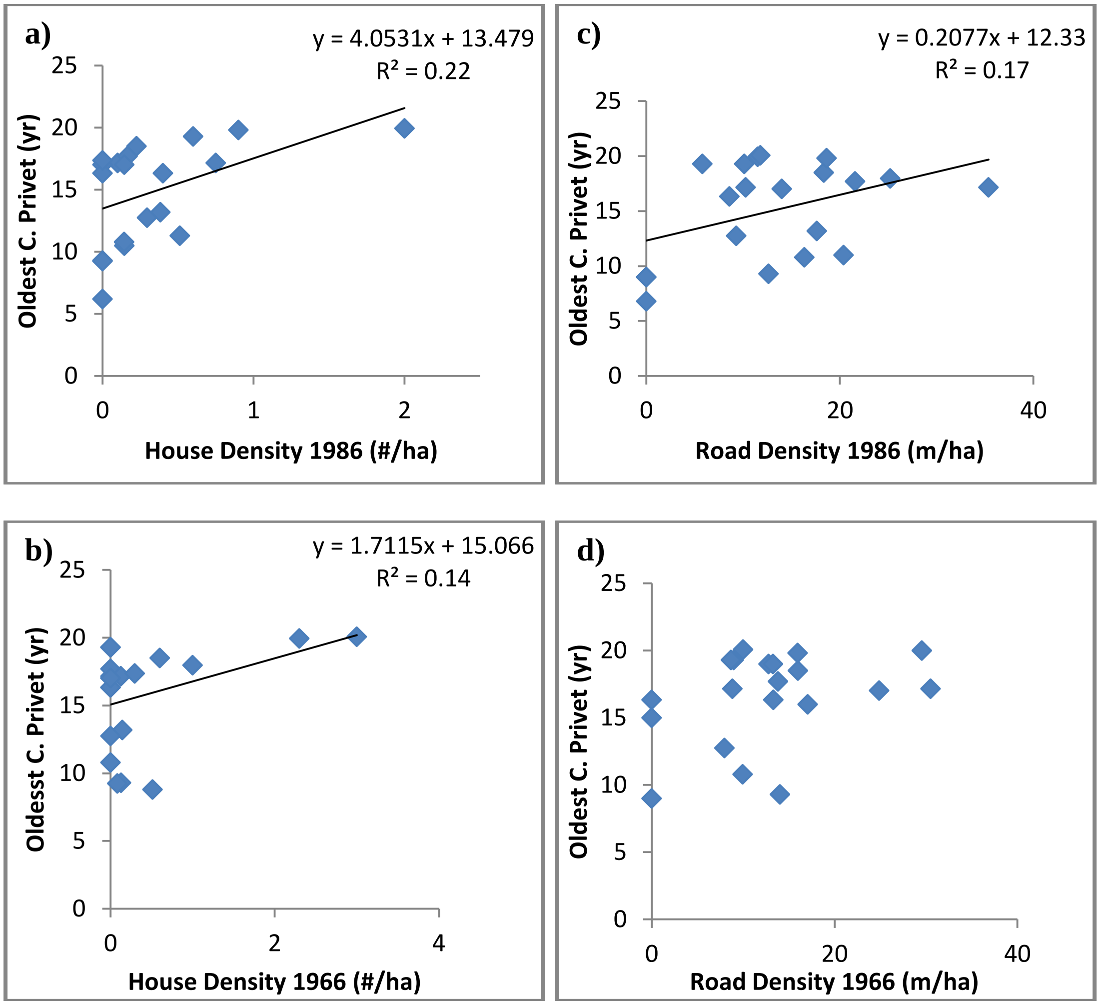

Figure 3.15 : For all sites designated agriculture land use for 1966, 1986 and 2011, the relationship between a) privet stand age and 1986 house density ($p<0.10$), b) privet stand age and 1966 house density ($p<0.10$), c) privet stand age and 1986 road density ($p<0.10$), and d) privet stand age and 1966 road density. Stand ages were calculated based on 2014 DBH data.

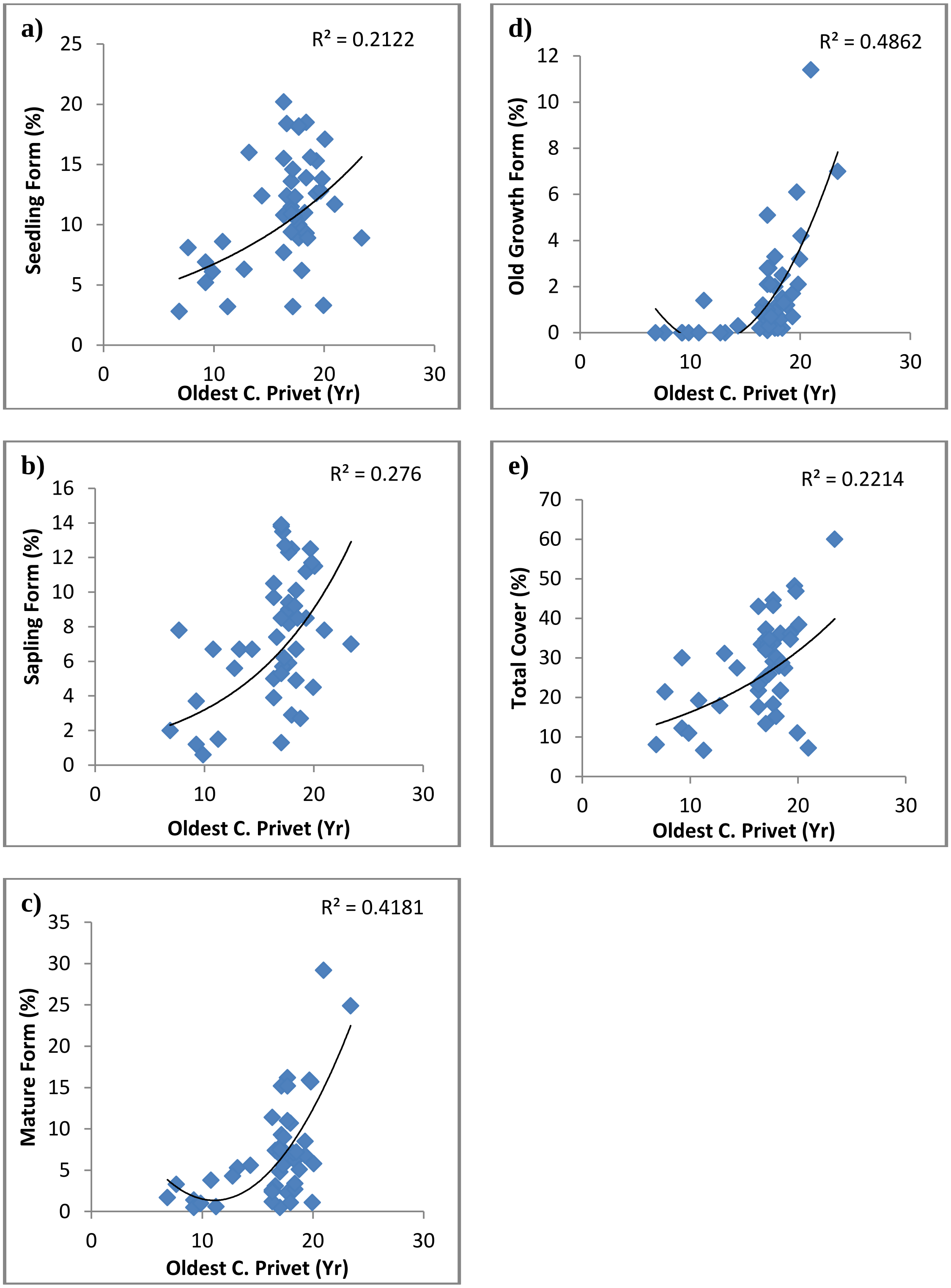

Figure 3.16 :The relationship between a) seedling form cover and stand age, b) mean sapling form and stand age, c) mean mature form and stand age, d) mean old-growth form and stand age, and e) mean total cover % and stand age for Chinese privet ($p<0.05$). Stand ages were calculated based on 2014 DBH data.