A Comparison of Vegetation Composition in Urban and Rural Floodplans Following Removal of Chinese Privet (Ligustrum Sinense)

A COMPARISON OF VEGETATION COMPOSITION IN URBAN AND RURAL FLOODPLAINS FOLLOWING REMOVAL OF CHINESE PRIVET (*LIGUSTRUM SINENSE*)

by

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Under the direction of Leslie A. Edwards

ABSTRACT

An important aspect of restoration ecology is the removal of non-native invasive plants. While restorations in urban areas involve similar challenges to restorations in rural areas, urban efforts also contend with unique issues such as increased fragmentation and decreased seed sources for native species. This study examined efforts to eradicate *Ligustrum sinense* and allow native vegetation to occupy the landscape. The herbaceous layer was inventoried at study plots in riparian bottomlands of four Atlanta, Georgia, natural areas two years after start of treatment to remove *L. sinense*. Plant taxa were described and compared to a similar study conducted in a rural area of northeastern Georgia. Significant abundance of *L. sinense* was recurring at urban sites while recurrence at rural sites was low. Other non-native invasive species, frequently used in urban landscaping, were also occurring at urban sites and not at rural sites.

INDEX WORDS: Urban, Restoration, Invasive, *Ligustrum sinense*, Riparian

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Introduction

Three important trends in the study of ecosystems include the recognition of urban ecosystems as unique habitats, the realization that non-native, invasive species are serious threats to native plant communities in both rural and urban ecosystems, and advancements in the field of restoration ecology, which seeks to return degraded ecosystems to a healthier condition and improve their diversity. This research encompassed these trends, by examining vegetation composition following initial efforts to restore an urban ecosystem through removal of the non-native invasive plant species *Ligustrum sinense*.

Study Purpose

This research was undertaken with specific two goals: (1) to describe species established in the floodplains of natural areas and greenways within Atlanta, Georgia, after removal of the invasive plant species *L. sinense*; and (2) to add to the scant body of literature regarding invasive species removal and site recovery in urban areas through a comparative analysis between species occurring at sites treated for *L. sinense* removal in Atlanta, to species occurring under similar circumstances in a rural area of Georgia studied by Hanula, Horn and Taylor (2009). The intention of the first goal was to answer the following questions.

1. What was the species composition for each urban site, and all urban sites combined, two years after treatment?
2. What was the prevalence of native versus non-native taxa at urban sites?

3. Were non-native species invasive?

The second goal aimed to answer research questions regarding a comparison with similar research conducted in a rural area of Georgia (Figure 1).

1. Were more non-native invasive species colonizing urban floodplains than rural floodplains two years after removal of *L. sinense*?
2. Did native species colonizing urban floodplains exhibit less diversity compared to rural floodplains?
3. How was the diversity of the native species reflected in species richness and evenness?

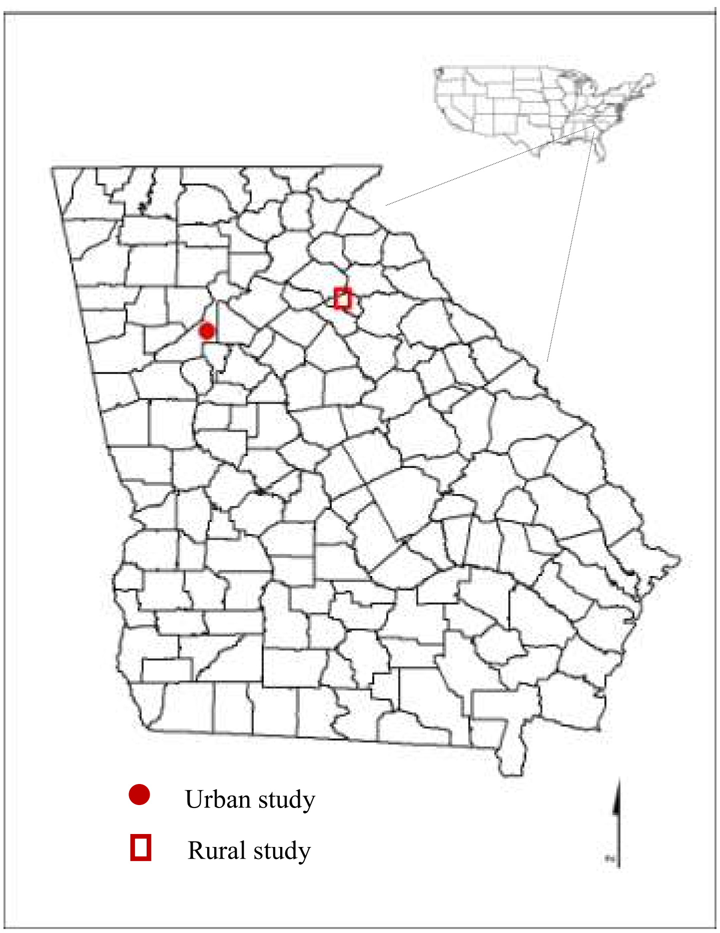

Figure 1. A map of the state of Georgia indicating location of urban and rural research sites

Research was conducted within the City of Atlanta, in partnership with the non-profit organization, Trees Atlanta. In 2008 the City of Atlanta contracted with Trees Atlanta to eradicate *Ligustrum sinense* and other invasive species such as *Nandina domestica*, *Elaeagnus pungens*, kudzu (*Pueraria Montana*), and English ivy (*Hedera helix*) from Atlanta city parks, conservation areas, and greenways. During the removal and treatment process, Trees Atlanta faced challenges typical of restorations, such as determining the former character and ecological diversity of treated areas, recurring invasive plants, and formulating a long term plan to maintain treated areas. To assist with planning, Trees Atlanta staff expressed interest in determining the species composition occurring on sites treated for *L. sinense* removal, and whether species becoming established are native or non-native.

A study of species composition after *L. sinense* removal contributes to the literature regarding restoration science particularly within urban ecosystems while providing information to assist Trees Atlanta, as well as other restoration efforts, in making longer term decisions regarding continued treatment of invasive species, and to what extent native plant species need to be seeded or planted in order to re-establish the native plant composition. Because the field of restoration ecology is relatively new, more research on removal of invasive species and recovery of treated areas is needed, particularly studies that quantify results (Moser et al. 2009; Miller and Gorchov 2004). There is also a shortage of urban restoration studies (Alberti 2008). No urban restoration studies within the City of Atlanta were identified. The comparative analysis with similar research conducted in a rural area adds to the body of knowledge regarding potential similarities and differences between urban and rural restoration efforts.

Literature Review

Restoration of urban ecosystems through removal of non-native invasive species occurs within the context of three areas of inquiry. The first is ecosystems and biodiversity, in order to understand what an ecosystem is and to ascertain the desirability of attempting restoration. The second is a description of non-native invasive species, in order to understand their negative impact upon ecosystems. In this case, the focus is *Ligustrum sinense*. The third is restoration science, to ascertain what is known about restoration techniques, particularly in terms of *L. sinense.*

Urban ecosystems are recently recognized as unique environments encompassing humans and other organisms living in urban areas, and the nonliving structures and processes with which they interact (Cutter and Renwick 2004). With recognition as ecosystems, cities are increasingly studied as sites encompassing human activities and the natural environment (Benton-Short and Short 2008; Alberti 2008). Likewise as cities have grown and encroached upon surrounding land, awareness of human impact on natural environments has increased. Human impact often results in degradation of the natural environment as human benefits and the monetary value of resources take precedence over other living organisms and their habitats. However, during the past several decades conserving the natural environment and taking action to restore degraded areas have gained priority, especially as the benefits of preserved natural areas have been realized and more highly valued (Platt 2006). These benefits include reduced storm water runoff, mitigated soil erosion, reduced urban heat island intensity, improved air quality, decreased noise levels, and increased biological diversity (Boone and Modarres 2006; Manning 2008; Clemants and Handel 2006).

Urbanization impacts the natural environment in multiple, complex ways. Development of roads, subdivisions, and commercial areas modifies land surfaces and form. Human activities within urban areas utilize large quantities of natural resources, alter energy and chemical cycles, and generate waste products. These processes disrupt nonhuman ecological systems and the organisms that depend on them (Alberti 2008). Although undeveloped sites may be set aside to preserve natural space for native plant and animal communities, as well as for human recreation and enjoyment, the impacts of urbanization are still felt within these sites. Ecosystems situated in urban areas contend with conditions altered from the environments in which they evolved, including habitat fragmentation and isolation, changed soil structure and composition, increased exposure to pollutants and toxins, higher temperatures, accumulations of nitrogen, and decreased seed sources for native species (Niemela 1999; Moll 1995).

Fragmentation, where undeveloped areas are preserved but disjunctive, is a particularly important impact. Fragmentation creates numerous forest edge environments, which are detrimental to species requiring shade and deep forest habitats, but are favorable to some species that favor higher light conditions. Fragmentation is particularly favorable to many non-native invasive plants, which often thrive in sunlight and have seeds that are spread to disjunct areas through animal droppings. In urban areas, the potential for invasion by non-native species is amplified because urban landscaping introduces new, non-native species to the environment, some of which are invasive (Alberti 2008). If climatic and geographical conditions are favorable, and without their natural insect and disease predators, non-native species escape where they were planted in the urban landscape and become established in forested areas, often outcompeting native species and causing a decline in the abundance and biodiversity of native species (Miller 2003). In the southeastern United States, *L. sinense* is one of the most troublesome non-native shrub species. Historically utilized as an ornamental plant in yards and hedges, its prolific seed production is dispersed by birds and animals. With a preference for warm, humid bottomlands, *L. sinense* has become widely established in riparian areas. Mature stands of *L. sinense* replace native plant communities and consequently reduce the insect, bird, and animal populations which depend on them.

An important aspect of restoration ecology is removing invasive species which have become established and are disrupting endemic ecosystems. Human intervention is needed to contain and remove the invaders and provide the opportunity for native plants, and the biodiversity they support, to become re-established (Hough 1995). Restoration research has focused on the most efficacious methods for removing the invader, as well as plant species composition after removal. Most of this research has been conducted in rural areas. While urban restoration efforts face similar challenges to restorations in rural areas, they also grapple with the unique conditions of urbanized areas, such as increased storm water runoff, poor air quality, and low biodiversity. Therefore, after the invading species is removed, restoration of the former habitat may be slower and require more human input and subsequent management than required in rural areas.

Ecosystems and Biodiversity

Ecosystems are systems in which materials and energy are transferred between organisms and their environment. The characteristics of, and organisms living within ecosystems, are tied to geologic, topological, edaphic, and climatic conditions of an area. An exchange of biotic and abiotic elements is constantly occurring within an ecosystem as energy passes through a series of storage and release mechanisms before returning to space as radiant energy. A major disruption or change to any of the biotic or abiotic components of an ecosystem may significantly alter the entire ecosystem (Cutter and Renwick 2004; MacDonald 2003). Human activities are dramatically changing the Earth's ecosystems through landscape changes, energy transformation, and alterations in material cycles. Such changes have accelerated during the past 50 years which has had a significant impact on biodiversity (Alberti 2008).

Biodiversity is one of the most important considerations within any ecosystem. Bacteria, plants, and animals are dependent on one another for the energy provided through the food chain. Richness in diversity is a measure of both the number of different species present in an ecosystem and genetic variability within each species. The organisms within an ecosystem have co-evolved and have symbiotic and dependent relationships upon one another. Species planted simply for aesthetic reasons and later escaped into natural areas, which did not co-evolve in the ecosystem, can change the characteristics of the entire ecosystem (Primack 1998). Hough (1995) explains changes to an ecosystem due to human activity as changes to the overall energy availability. An ecosystem relying on fewer sources of energy becomes more vulnerable (Jose et al. 2009). Urban landscapes are exemplars of human impact, with their altered and diminished energy sources on which a variety of life forms depend. The addition of invasive species to natural landscapes further exacerbates the degradation of ecosystems already impacted by urbanization, and further erodes the dependent relationships among native species (Gaston and Spicer 1998). In addition to reducing species richness, invasive species also impact the evenness of species. Evenness is a measure of species dominance, represented by the number of individuals of a given species within an area. For example, an area with low species evenness may have many individuals of chalk maple (*Acer leucoderme*) and one redbud (*Cersis canadensis*), whereas an area with higher evenness exhibits a more balanced quantity of each species. When a species becomes invasive to an area, evenness tends to decrease as the invader dominates resources, outcompeting other species and reducing the number of individuals representing a diverse variety of endemic species.

Non-native Invasive Species

In the literature on plant invasion ecology, definitions of non-native invasive species can be complex and controversial. A non-native plant species is one that is introduced to an area either intentionally or accidentally by humans. If conditions are highly favorable to an introduced species, and it lacks predators and competition, it may produce reproductive offspring which grow and proliferate in large numbers a significant distance from parent plants. A wide distribution and dense number of an introduced species eventually interferes with an endemic ecosystem and becomes invasive (Richardson et al. 2000). A native species is “a species that occurs naturally in an area, and therefore one that has not been introduced either accidentally or intentionally” (Allaby 1994). Definitions of both native and invasive species can be problematic because they reference a temporal component. This is resolved by specifying a period during which a species must have been present to be considered native, such as before global travel or before the arrival of European settlers in North America (Bergman and Swearingen 2005; Luken and Thieret 1997). For purposes of this research, a common definition of North American native species is used, those present on the continent when the colonists arrived approximately 400 years ago. Not all species fit into such a neat classification of native. Native Americans modified plant communities and planted squashes (*Cucurbita* spp.) and other foods originally from Mesoamerica, for example, giving those foods status as native plants under this definition (Schwartz 1997).

After habitat destruction, human introduction of alien invasive species is the second main cause of Earth's biodiversity loss (Cutter and Renwick 2004; Jose et al. 2009; D'Antonio 1997). Invasive species cause biodiversity loss by outcompeting native plants for resources such as light, nutrients, and water. Other competitive advantages include alterations of soil composition and formation, disruption of chemical processes and nutrient cycles, allelopathic properties, lack of insect predators, or a combination of these advantages. Theories from community ecology also suggest that site quality, disturbances such as urbanization, and phenology, or timing of leaf emergence, flowering, and seeding, may play a role in the success of invasive species in a new environment and climate (Wolkovich and Cleland 2011; Moser et al. 2009; Walker and Smith 1997; Fei et al. 2009; Woods 1997). Competitive advantages assist non-native invasive species in becoming established outside their native ranges and disrupting ecological systems in an historically short time, sometimes less than a decade (Dyer and Cowell 2009; Primack 1998). Invasive species are destructive to entire ecosystems as insects and animals can no longer depend on native species as sources of food and habitat.

Many invasive species in the United States have detrimentally impacted entire geographic regions. Some of the better known examples include Chestnut blight (*Cryphonectria parasitica*), accidentally introduced from Asia in the early 1900s which killed an estimated four billion chestnut trees (*Castanea dentata*) in the eastern United States, a quarter of the hardwoods present in the forest at the time (American Chestnut Foundation, http://www.acf.org/history.php) and kudzu (*Pueraria montana*) a woody vine that was introduced in the United States from China and Japan in the late 1800s to control erosion, provide feed for livestock, and shade southern porches (Miller 2003). Government sometimes unwittingly aids establishment of an invasive species. A program in the 1930s and 1940s under the auspices of the Soil Conservation

Service grew 84 million *P. montana* seedlings and paid farmers to plant them (Jenkins and Johnson 2009; Blaustein 2001). When successfully established, *P. montana* covers the ground growing over shrubs, trees, and built structures, aptly giving it the reputation as "the vine that ate the south" (Blaustein 2001). Although well known and highly recognizable along roadsides in the southeastern United States (Figure 2), as of March 2008 *P. montana* covered approximately 220,000 acres of U.S. Forest Service land in twelve southern states. Other species are much more problematic. The most common invasive plant on Forest Service land in the southeastern United States is Japanese honeysuckle (*Lonicera japonica*), a vine covering approximately 10.3 million acres as of March 2008, also growing over trees and shrubs and replacing many native plant species. The privets (*Ligustrum* spp.), an invasive shrub including Chinese, European, Japanese and glossy together cover approximately 3.2 million acres (Miller 2008).

Photo by SMorrell

Figure 2. Image showing an infestation of kudzu (*Pueraria montana*) along Avon Drive in Atlanta, GA

Restoration Science

Removing a non-native invasive species and restoring the natural environment present significant challenges (Niemela 1999). The field of restoration ecology is relatively new therefore the expertise to answer many important questions and provide the guidance needed to ensure successful restoration outcomes is still developing (Alberti 2008). When managing a restoration project, conditions and character of a site need to be considered as well as the desired outcomes. Part of any restoration plan that involves removal of non-native species preferably includes a return of the targeted site's ecosystem to a semblance of its structure before the invasive species became dominant, including return of both native plants and animals eliminated or degraded (Clemants and Handel 2006). However, the former ecosystem structure may be difficult to determine for a variety of reasons, especially in an urban environment. Historic records may be lacking, or the land may have undergone several land use changes over time. If an area has experienced a long period of invasion, little indication of the former ecosystem may remain and reference sites may be difficult to locate. These challenges are significant considerations particularly during the planning phase of a restoration project.

Various approaches to restorations are often based on different sub-disciplines of ecology. Hough (1995) advises using forest ecology and its processes as the basis for forest restorations, since ecosystems are not static and continue to evolve. A similarly situated uninvaded environment, when available, should be studied to determine its components and successional phase (Hough 1995). Clemants and Handel (2006) apply botany and community ecology to restoration efforts and emphasize restorations involve function rather than appearance. After the initial restoration work, changes occur over time as species become stable, disperse, and die, and other species become established, resulting in a different outcome from the original appearance (Clemants and Handel 2006). Zedler (2006) recommends adaptive management as a restoration strategy which systematically applies knowledge gained during the restoration process at a given site as well as information from reference sites. Adjustments are made to the project as new information is learned so that successful outcomes are more likely. Plans and goals must be set which determine priorities, with the understanding that for highly degraded sites, only a few objectives may be met. When funding is available and people are motivated to work, momentum is important and should be utilized whether or not a perfect outcome can be achieved (Zedler 2006). Often there is no quick fix and results can take years to achieve, particularly in reestablishing historic soil composition and function. Consideration of all factors is difficult and in some instances fundamental conditions of restoration sites change due to outside influences, or restorations may result in unintended consequences, for example, the clearing of a site with an infestation of *Hedera helix* may lead to erosion and loss of topsoil during heavy rain events. Because restoration science is relatively new, whether a project is successful or achieves less than optimal results, the experience and knowledge gained add to the body of information about restorations. Monitoring, measuring, and evaluating as the project progresses and after completion are essential (Zedler 2006; Cutter and Renwick 2004).

After eradication of an invasive species, many factors may influence the return of desirable or native species, including light levels reaching the forest floor, temperature, rainfall, and soil moisture and composition (Hartman and McCarthy 2004). Prolonged infestation may have altered soil properties such as alkalinity, requiring remediation before planting native species may occur (Hartman and McCarthy 2004; Clemants and Handel 2006). Additionally, there must be a source of plant propagules. Several methods of reestablishing native species may be employed. One is reseeding or planting of the area. Replanting may hinder reestablishment of invasive species by creating competition for space and resources and may speed successional processes (Hartman and McCarthy 2004). Another method is to depend on the existing seed bank. With exposure to light, propagules of native species that have survived the invasion may sprout and become established. Seeds from the invader species will also likely be present in the seed bank and other invasive species' propagules may be present especially in an urban area with multiple sources from urban landscaping. Clemants and Handel (2006) stress patience when restoring a site, allowing time for species to mature and begin reproducing, which can take many years. Their studies have shown the need for long term management of restoration sites to continue the original goals of the project, control establishment of new colonies of invasive species, and continually add desired seeds or replant until native species become established (Clemants and Handel 2006).

Several of the challenges documented in the literature on restoration science were experienced during *L. sinense* removal in Atlanta, Georgia, parks and greenways, including highly altered landscapes due to multiple disturbances over time, determining the former character of treated sites, lack of seed producing native plants, and the presence of non-native invasive species on private property surrounding treated sites, providing continued sources of propagules.

Ligustrum sinense

Ligustrum sinense was brought to the United States during the 1850s from temperate and subtropical China, Vietnam and Laos for use as an ornamental species in hedges and as individual specimens (Starr, Starr, and Loope 2003; Nesom 2009). It is the most common of the invasive *Ligustrum* spp. shrubs (Miller 2003). Panicles of fragrant white blooms appear from

April to June, hence its desirability in gardens for hedges and screening. The scent has a distinctive muskiness and may be considered unpleasant (Godfrey 1988). Fruits form July to March in dense ovate drupes with each fruit containing one to four seeds that ripen from pale green to purple to blackish. Seeds are eaten by many fruit-eating bird species as well as other animals and dispersed widely through droppings. *L. sinense* also sprouts from roots of mature plants and from cut stumps. It prefers moist woods, bottomlands along streams, and disturbed habitats. It readily colonizes, forming dense thickets and displacing populations of indigenous herbaceous, shrub, and sapling species (Figure 3). It also tolerates mesic uplands (Miller 2003; Godfrey 1988; Starr, Starr, and Loope 2003).

Photo by MMorrell

Figure 3. Image of a restoration site at Hampton Tract Greenway, Atlanta, GA. Foreground has been treated for *Ligustrum sinense* removal while background has not

Competitive advantages of *L. sinense* include reaching a mature height up to thirty feet within a decade and treelike form with a vertical rather than horizontal branch and leaf arrangement which may capture more light compared to native shrubs such as *Calycanthus floridus* and *Forestiera ligustrina* (Morris, Walck, and Hidayati 2002). Flowers are monoclinous, with both stamens and pistils, increasing pollination success compared to native shrubs such as *C. floridus*, the flowers of which are dioecious, having male stamens and female pistils on separate individual blooms. While *L. sinense* is deciduous in its native range, humid areas of eastern and southern Asia, in southeastern United States forests *L. sinense* is semi-evergreen to evergreen unless conditions are unusually cold or dry. This gives it photosynthesis and resource usage advantages compared to native deciduous species (Morris, Walck, and Hidayati 2002). Its prolific production of seeds is also advantageous. In Georgia, *L. sinense* is classified as a Category I invasive species, the most problematic group, by the Georgia Exotic Pest Plant Council (http://www.invasive.org/browse/subinfo.cfm?sub=3035). It is estimated to cover approximately 2.69 million acres in 12 southeastern states (Miller 2008). However this is an estimate of interior forest plots and does not include urban parks, private property, or forest edges and therefore underestimates the extent of invasion (Hanula and Horn 2011).

Wangen and Webster (2009) recommend formulating invasive control management plans based on an invasive species' life-history traits. These traits encompass how an invasive species reproduces and disperses, whether it is an annual, perennial or biennial, whether it has natural enemies, and when it is most vulnerable to treatment compared to native species (Wangen and Webster 2006). *L. sinense* exhibits many life-history traits typical of invasive species and has few documented insect enemies (Morris, Walck, and Hidayati 2002). As a proficient space invader, with a fast colonization rate in open, disturbed areas, *L. sinense* is difficult to eradicate (Miller 2003; Luken 1997). When cut or pulled to reduce or eliminate its biomass, numerous sprouts grow from cut trunks and stems, and roots left in contact with soil (Godfrey 1988; Miller 2003).

The most common treatment method for eliminating *L. sinense* is cutting either by Gyrotrac® mulching machine, chainsaw, or hand, followed by chemical spraying of stumps, and additional chemical foliar application during subsequent years to kill sprouts and seedlings (Hanula, Horn, and Taylor 2009). Hanula, Horn and Taylor's (2009) research compared the efficacy of two *L. sinense* removal methods in the Oconee River watershed of northeastern Georgia. One treatment method involved mechanically cutting with a Gyrotrac® mulching machine and the other utilized hand-felling followed by chemical treatment. Results from the two methods were compared to untreated control plots and similarly situated plots uninvaded by *L. sinense*. Overall biomass was reduced by both treatment methods however neither method resulted in a reduction of quantity of *L. sinense* plants. Only subsequent treatment with chemical foliar spray a year after the initial cutting reduced plant numbers. Hanula, Horn and Taylor (2009) also inventoried herbaceous, shrub, and tree species after *L. sinense* removal to analyze recurring species diversity and abundance. Diversity and abundance for the shrub layer was not significant between treatment methods. Although treatment methods did not affect tree cover, they found that when compared to the uninvaded plot, tree abundance had a negative correlation with *L. sinense* cover. As older trees died and created open gaps in the forest, no immature trees were present to fill the gaps. This was due to *L. sinense's* dense coverage which outcompetes tree seedlings and saplings, giving *L. sinense* further opportunity to extend its coverage (Hanula, Horn, and Taylor 2009). Hanula, Horn and Taylor (2009) also found *L. sinense* removal had a high impact on the herbaceous layer two years later, with species richness similar to study plots not invaded by *L. sinense* although the plant communities differed significantly from the uninvaded plots. Their research showed treated sites were not reverting back to a desired former condition within the first two years after removal (Hanula, Horn, and Taylor 2009).

A study by Merriam and Feil (2002) measured the effects of *L. sinense* in a mixed hardwood forest in western North Carolina and found a negative correlation between *L. sinense* cover and native plants under the *L. sinense* for tree seedlings and herbaceous plants. When *L. sinense* was removed, both tree seedlings and herbaceous plants increased in abundance during the subsequent growing season (Merriam and Feil 2002). Other studies conducted on the effects of the invasive shrub, Amur honeysuckle (*Lonicera maackii*), have yielded similar results with shrub density negatively impacting growth and reproduction of herbaceous native species (Miller and Gorchov 2004).

Research in restoration science has surveyed and tested the challenges faced when attempting to restore ecosystems devastated by an aggressive invasive species. To add to the literature, the premise of the research for this thesis was that those challenges are magnified when restoration sites are situated within an urban environment due to the added stresses imposed by urbanization. The information provided by this research will assist in determining if composition of species occurring after removal of an invasive species is affected by location within an urban ecosystem.

CHAPTER 2

STUDY SITES AND METHODOLOGY

Site Selection

Because one of the major purposes of this study was to compare urban to rural areas, the nature preserves and greenways chosen for research were situated within the city of Atlanta, Georgia. Sites were selected in collaboration with Trees Atlanta. Trees Atlanta's goal was to protect and improve Atlanta's urban forests and trees. This goal was achieved primarily with tree plantings and community education. The organization also included a Forest Restoration program that focused on restoring Atlanta's protected forests, greenspaces, and urban native plant communities. Much of this focus was on removing invasive plant species through neighborhood volunteer programs, contractor spraying and removal, community education, and replanting native species (Trees Atlanta, http://www.treesatlanta.org).

Site selection was based on four factors. The first was potential for biodiversity of recurring species after *Ligustrum sinense* removal. This potential was determined through conversations with Trees Atlanta staff and based on their observations of plant species diversity in the immediate areas where *L. sinense* was removed and subsequently treated. Because of the forested characteristic and lack of development within certain natural areas where invasive removal efforts were concentrated, Trees Atlanta staff members stated they had observed more biodiversity in several city parks than was observed in many typical urban parks maintained for open grassy areas, with a sparse tree layer and little to no shrub layer. Trees Atlanta staff therefore expressed interest in the vegetation becoming established after *L. sinense* removal in the city parks chosen for this research (Figure 4).

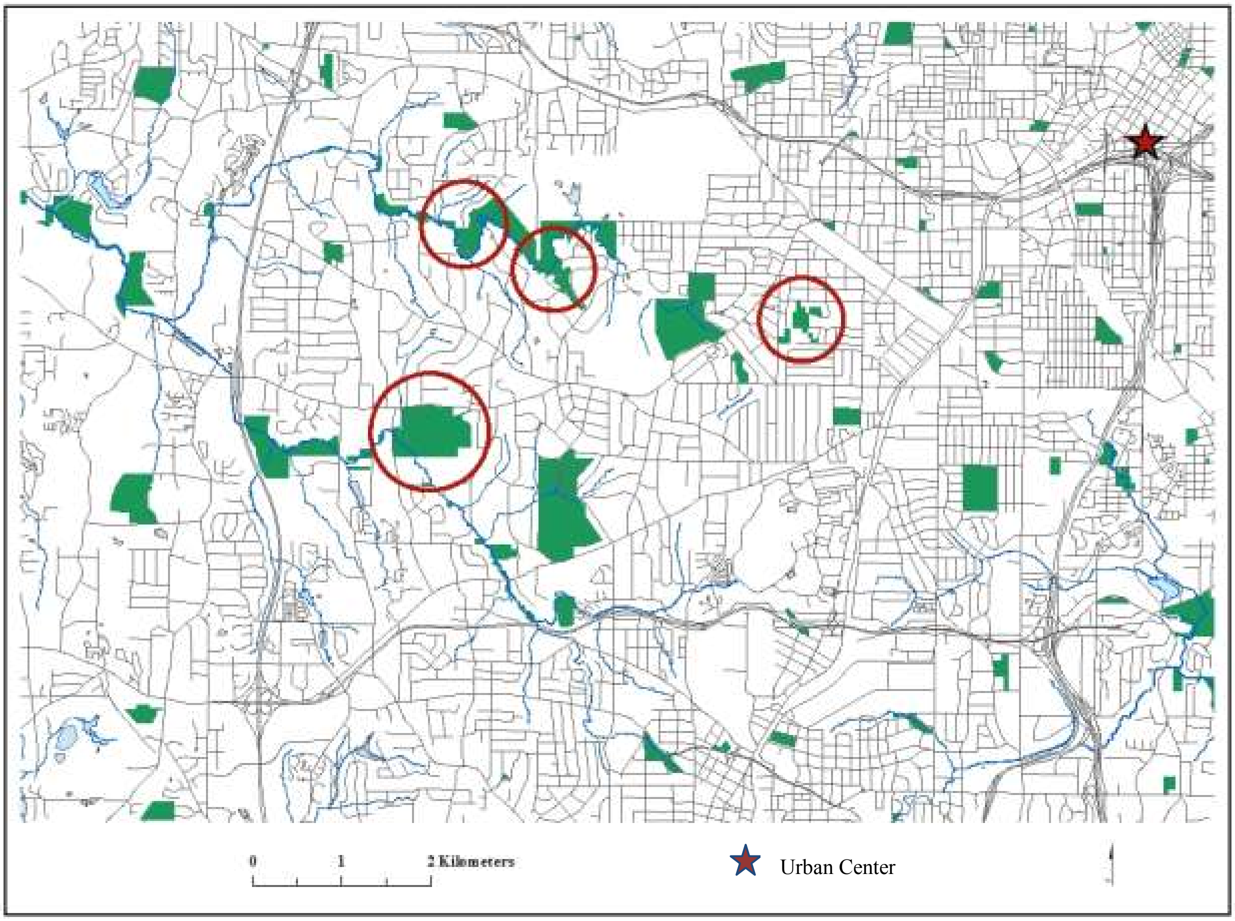

Figure 4. A map of the southwest quadrant of downtown Atlanta with locations of study sites demarcated by red circles. Green areas represent city parks and greenways

The second factor considered in site selection was timing and method of *L. sinense* treatment. A Trees Atlanta subcontractor began *L. sinense* treatment in Atlanta city parks in November 2008 by cutting plants and treating stumps with 13.6% triclopyr herbicide mixed with basal oil. Cuttings were stacked in piles for on-site composting. In subsequent years, seedlings and resproutings were treated with a foliar application of 3.7% Accord herbicide. *L. sinense* was typically treated on all study sites during winter and early spring months to minimize impact to other flora that remained in winter dormancy. To confirm consistency with *L. sinense* removal timing and treatment methods at rural sites, subcontractor invoices and site surveys were reviewed and summarized for treatment start dates, subsequent treatment dates, and treatment methods.

At the rural sites used for comparison with this research, researchers utilized several treatment methods. One of those methods was hand cutting and painting stumps with 30% triclopyr herbicide. Subsequent treatment of regrowth occurred approximately one year later with a broadcast herbicidal spray of 2% glyphosate and surfactant (Hanula, Horn, and Taylor 2009). Although the herbicide concentrations used for initial and regrowth treatments varied between the urban and rural areas, method and timing were similar. Treatment began approximately two years prior to inventorying species at both the urban and rural sites.

To ensure research in both the urban and rural areas was conducted in similar natural communities, the third consideration for study site selection was situation in riparian bottomlands. Three of the four urban natural areas included active stream channels with floodplain zones. The fourth included an ephemeral creek bed. Additionally, research for both studies was conducted within the same ecoregion of Georgia.

The fourth consideration in site selection was *L. sinense* infestation level prior to treatment. Trees Atlanta conducted pretreatment surveys with invasive species consultants and the treatment subcontractor, which was documented in the subcontractor's records. This was compared to the pretreatment *L. sinense* infestation level at sites in the rural study. Study sites for both the rural and urban studies included a medium to heavy infestation of *L. sinense* before treatment began. Generally, these levels of infestation were characterized as mature stands, growing unimpeded for at least a decade. Heavy infestation was found in bottomland areas where *L. sinense* stands were 15 to 30 feet tall with density that was impassable. Medium level infestation occurred on portions of the bottomlands farthest from stream channels, with similar aged stands exhibiting less height due to less light through the closed tree canopy. This level of infestation was dense although passable.

Site Descriptions

Selected nature preserves and greenways were within the City of Atlanta boundary in urban neighborhoods proximal to residential and business land uses. Based on the selection criteria, the four study sites identified for this research were Cascade Springs Nature Preserve, Outdoor Activity Center, Beecher Hills Greenway Tract, and Lionel Hampton Greenway Tract (Table 1). Unlike many urban parks with open expanses of mown grass and built facilities, these nature preserves and greenways were forested with little to no open areas and few built facilities. Generally, the areas were conserved natural areas utilized for appreciation of the natural environment and for exercising such as walking and running. The forest was fragmented due to its situation in the urban built environment. Adjacent urban residential and private properties included many typical urban landscaping plant species utilized in the southeastern United States such as *Euonymus* spp., *Impatiens* spp., *Begonia* spp., *Hedera helix*, and *Nandina domestica*. Landscapes also included *L. sinense* hedgerows, thickets, and mature individual specimens providing a continued source of propagules to areas treated for *L. sinense* removal.

Table 1. Summary of urban study sites

Park/Greenway	Address	Type	Total Acres	Treatment Began	Approx. *L. sinense* Acres Treated
Cascade Springs Nature Preserve	2852 Cascade Rd SW	Nature Preserve	120	November, 2008	20
Outdoor Activity Center	1442 Richland Rd SW	Nature Preserve	22	February, 2009	3
Beecher Hills Tract	Bolling Brook Dr SW	Greenway Acquisition Project	60	February, 2009	17
Lionel Hampton Tract	Flamingo Rd	Greenway Acquisition Project	54	January, 2009	19

Source: Clean Water Atlanta, Greenway Acquisition http://www.cleanwateratlanta.org/greenway/Properties/default.htm and Trees Atlanta records

Cascade Springs Nature Preserve had no built facilities except a dilapidated wooden structure and a tiled spring house once used as a resort for people to visit the springs and bathe in the water for healing purposes. It was acquired by the city in the early 1970s and included several hiking paths and a small waterfall. The preserve included a rich population of native plants, with an exceptional stand of Bigleaf Magnolia (*Magnolia macrophylla*), on a northeastern slope. It also appeared to have been farmed, indicated by rock terraces and fencing material on a flatter upland area, and several older trees that perhaps provided shade to buildings no longer present.

Lionel Hampton and Beecher Hills Greenway Tracts were adjacent properties acquired by the city in 2002 as part of a settlement with the United States Environmental Protection Agency, Georgia Environmental Protection Division, Upper Chattahoochee River Fund, Inc., and other parties for city violations of the Federal Water Pollution Control Act and Georgia Water Control Act. Under the settlement, the city acquires and protects in perpetuity, properties adjacent to rivers and streams in a natural, undisturbed state within the metro Atlanta area (City of Atlanta, http://atlantawatershed.org/greenway). The Lionel Hampton Greenway Tract was adjacent to the PATH Foundation's Lionel Hampton Trail, a paved multiuse path which eventually will connect to the BeltLine Trail (Figure 5).

Photo by S Morrell

Figure 5. Image of Lionel Hampton Trail adjacent to Lionel Hampton and Beecher Hills Greenways

The only site with functioning built facilities, the Outdoor Activity Center was established as a nature preserve and to educate the public about the natural environment. The land was acquired by the city in the 1970s and the building, which included an educational display, meeting area, and offices, was dedicated in 1992. In front of the building were a World Wildlife Fund Certified Wildlife Habitat and a bio-garden project that demonstrated recycling water between a sustainable, organic vegetable garden and adjacent fish tank. The Center housed offices of the West Atlanta Watershed Alliance, a community based non-profit organization, originally established during a struggle for environmental justice over discriminatory waste water treatment practices in southwest Atlanta. The Alliance's mission was to protect, conserve, and restore the area's natural resources. It sponsored outdoor, natural environment, and clean water educational and service programs particularly aimed at youth in southwestern Atlanta's minority and low income communities. Education is an important component of eradicating invasive species as infestations occur on private property adjacent to many local city parks and removal efforts take place site by site at smaller spatial scales (Hartman and McCarthy 2004). Since *L. sinense* is widely spread by birds eating its numerous propagules each year, assistance from volunteers and adjacent private property owners is imperative for eradication from natural areas.

At each of the four selected City of Atlanta natural areas, study sites were identified in bottomland areas near streams. South Utoy Creek flowed through Cascade Springs Nature Preserve; North Utoy Creek flowed through the Hampton and Beecher Hills Greenway Tracts. The two creeks merged approximately two and one-half miles west of the study parks forming Utoy Creek which flowed into the Chattahoochee River approximately four miles to the west. At Outdoor Activity Center an ephemeral creek bed ran through the bottomland. Streams ran relatively low during normal to dry periods and became heavily swollen during moderate to heavy rains. Runoff into the streams was exacerbated by surrounding urban structures and pavement. Heavy runoff and flooding exposed the area to frequent disturbance. The streams were means of seed dispersal and sources of nutrients, providing favorable growing conditions for *L. sinense* (Schiffman 2009). Dense stands of mature *L. sinense* were left on stream banks due to concerns that removal would lead to soil erosion during rain events. Sites included canopy trees characteristic of many southeastern United States floodplains, including tulip poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), red maple (*Acer rubrum*), and box elder (*Acer negundo*), as well as species of mesic portions of bottomlands such as chalk bark maple (*Acer barbatum*) and Eastern hophornbeam (*Ostrya virginiana*).

Data Collection

On each of the four selected study sites, three quadrats were randomly identified in bottomland areas previously infested with *L. sinense*, for a total of 12 study plots (Figure 6). All inventoried quadrats were located at least ten meters from stream channels to avoid including the untreated buffer along stream edges.

Figure 6. Maps of study areas (green), streams (blue) and study plots (brown). Clockwise from top left, Hampton Greenway, Beecher Hills Greenway, Outdoor Activity Center, and Cascade Springs Nature Preserve

Each quadrat was laid out as a 20 meter by 20 meter square with two points randomly selected along the northern edge to run transect lines north to south within the plot (Figure 7).

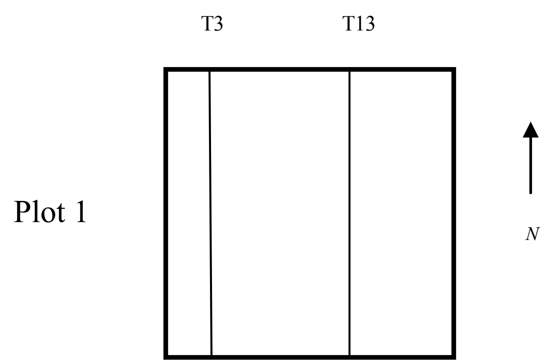

Figure 7. Diagram of 20 x 20 meter study plot with two transects at randomly selected whole meter points

A modified point-line method was used to inventory the herbaceous and shrub/sapling plant layers by laying a meter stick perpendicular to transects at each whole meter point along the transect (Godínez-Alvarez et al. 2009). Plants intercepting the meter stick were inventoried, including overhanging shrubs and saplings resulting in a total of 20 meters inventoried per transect (Figure 8). To facilitate comparison with the rural study, plants intercepting each whole point along transects were also recorded. Trees less than four meters in height (approximately 13 feet, 1.5 inches) were recorded as saplings. Each plant was identified to species where possible. Voucher specimens were collected in cases of difficult identification. In a few instances, identification to only genus or family taxonomic level was possible. Plants that could not be identified because of their small size or due to insect or treatment damage were labeled as unknown.

Photo by MMorrell

Figure 8. Image showing meter line method of inventorying plant species

Species and counts inventoried at each meter line and meter point were recorded per quadrat. The three quadrats per site were summarized at site level by species and counts, and assigned a native or nonnative indicator using Flora of the Southern and Mid-Atlantic States (Weakley 2011). Species considered invasive to the region by the United States Department of Agriculture and Southeast Exotic Pest Plant Council were also indicated.

Data Analysis and Interpretation

Quantitative data collected were compiled and mathematically compared for similarities and dissimilarities between data sets to describe and compare community structure. Relative density of each species, represented by the formula

$$\text{Relative density} = \frac{\text{individuals of species x}}{\text{total individuals of all species}} \times 100$$

was calculated per site and for all sites combined. Relative density was also calculated separately for native and non-native species.

To fulfill the first goal of this research which was to describe urban species composition, line data were utilized because they included more species richness and abundance than the point data, providing a more robust representation of plants becoming established. Point data were used for the second, comparative goal, of this research. Point data facilitated comparison for two reasons. First, data gathered at transect points were used in the rural study. Second, these data avoided counting clusters of plants found within a small area due to limited seed dispersal or locally favorable germination conditions for a specific species. However, using point data presented several limitations due to disparity in site and quadrat size between the urban and rural research. Smaller overall size of the urban natural areas and greenways compared to the rural watershed dictated smaller study sites within the urban natural areas and greenways. Hence, space for plots was limited within the bottomlands of urban study sites. Rural plots were two hectares each. One rural plot was considerably larger than the treated area at the Outdoor Activity Center, and approximately one quarter the size of the treated areas at the other three urban study sites.

Another limitation of the urban study sites was the discontinuous pattern of invasive species present. For example, the section of the bottomlands in Beecher Hills Greenway where study plots were located was treated for *L. sinense*, while an adjoining section was treated primarily for an invasion of *Wisteria* spp. Another adjacent section was treated primarily for an invasion of *Hedera helix*. Differing treatment methods were used for the different invasive species. Timing of treatments was also different. Therefore, long transects within an area treated consistently for *L. sinense* were difficult to select at most sites. The site where a strategy of long transects may have been achievable was Hampton Greenway. Species richness was very poor at this site, therefore additional transect length would most likely not have changed outcomes. Urban study quadrats were 20 square meters each, a reasonable size for inventorying the urban sites within the limited total area. The ratio of individual plants inventoried on transect points was 79 urban to 497 rural. The low ratio was partly due to a significant number of bare points on urban study plots, limiting the abundance of individual plants inventoried within the urban plot sizes.

A visual representation of the urban plant communities was plotted with a detrended correspondence analysis ordination using the PAST program. Ordination is a basic ecological technique that provides a two-dimensional representation of similarity and difference in plant communities. Similar communities plot closely together and different communities plot farther apart (Hammer, Harper, and Ryan 2001).

To compare native species composition between urban and rural studies, an index of similarity was calculated which measures degree of difference. The index is a method of determining diversity based on the presence or absence of species. The index of similarity equation is:

$$\text{Index of similarity} = \frac{2 \text{ x number of species occurring in both samples}}{\text{total species in sample 1} + \text{total species in sample 2}}$$

A limitation of the index of similarity is that it does not consider abundance, or the number of individuals present for each species. Therefore a single, or few species may dominate a sample (MacDonald 2003). To determine whether native species colonizing the urban floodplains exhibited less species richness than native species colonizing rural floodplains, the Shannon index (H') was calculated which takes both species richness and evenness into consideration with number of taxa and number of individuals. The index is represented by the equation

$$H' = -\sum pi \, In \, pi$$

where p_i is the proportion of the *i*th species and In is the natural logarithm. An estimate of evenness (E) alone was measured with the equation

$$E = H'/\text{In } S$$

where H' equals the Shannon Index, In is the natural logarithm, and S is species richness (MacDonald 2003). To visually represent the urban and rural data, a diversity profile was plotted using the PAST program. The diversity profile compared diversity and abundance for the two studies (Hammer, Harper, and Ryan 2001).

Non-metric multidimensional scaling is an ordination technique that preserves the ranked differences between species. Non-metric multidimensional scaling ordination was performed for urban and rural quadrats, and included data from desired future condition sites surveyed in the rural study. The resulting plot visually represented differences between the urban and rural studies, as well as differences with plots not impacted by an invasion of *L. sinense*. The PAST program was used to perform this ordination with the Raup-Crick randomization method which is a probably measure of whether samples have dissimilar species composition (Hammer, Harper, and Ryan 2001).

CHAPTER 3

RESULTS AND DISCUSSION

Urban species composition

The first goal of this research was to describe species occurring in the floodplains of natural areas and greenways within Atlanta, Georgia, after removal of the invasive plant species *Ligustrum sinense*. The intention of this goal was to answer the following questions.

1. What was the species composition for each urban site, and all urban sites combined, two years after treatment?
2. What was the prevalence of native versus non-native taxa at urban sites?
3. Were non-native species invasive?

At all study sites combined, the most abundantly occurring species was *L. sinense* at 27.3% of species inventoried (see Appendix A for a table of all species inventoried). Six of the ten most abundant species were non-native (Table 2), of which five including *L. sinense,* were classified as invasive in the state of Georgia by the Southeast Exotic Pest Plant Council (http://www.se-eppc.org/). The four other non-native species were *Hedera helix*, honeysuckle vine (*Lonicera japonica*), winter creeper (*Euonymus fortunei*), and *Wisteria* spp. Of the native taxa occurring, grape vines (*Vitis* spp.) were the most common. The second most common native species occurring was *Liriodendron tulipifera*.

Table 2. Ten most common species inventoried on urban study plots (meter line data)

Species	Plantspp	Type	Native	Invasive	% of Total
Ligustrum sinense	privet	seedling	n	y	27.28
Hedera helix	English ivy	vine	n	y	19.78
Vitis spp.	grape	vine	y	na	8.37
Euonymus fortunei	winter creeper	vine	n	y	7.61
Liriodendron tulipifera	tulip poplar	seedling	y	na	7.50
Lonicera japonica	honeysuckle vine	vine	n	y	6.85
Youngia japonica	false hawksbeard	herbaceous	n	n	4.02
Parthenocissus quinquefolia	Virginia creeper	vine	y	na	2.28
Phytolacca americana	pokeweed	herbaceous	y	na	1.96
Wisteria spp.	wisteria	vine	n	y	1.74

The detrended correspondence analysis diagram showed species composition varied considerably among the urban study sites. Outdoor Activity Center and Beecher Hills Greenway exhibited the most similarity and were distinctly different from Hampton Greenway and Cascade Springs Nature Preserve (Figure 9).

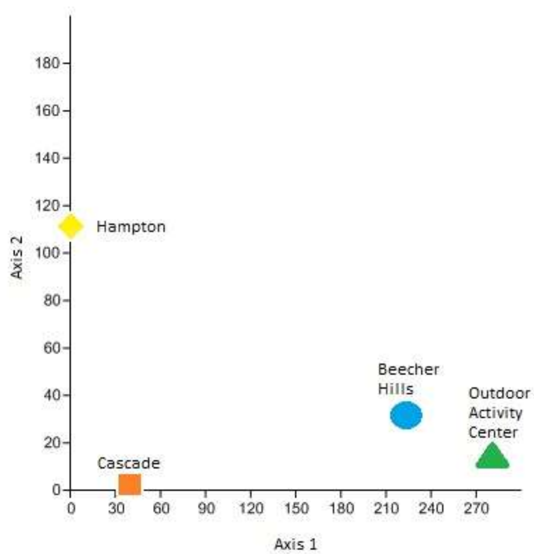

Figure 9. Diagram of detrended correspondence analysis ordination graph for urban plots

Neither native nor non-native invasive species were evenly distributed across the study sites. Each site had a different non-native invasive species most commonly occurring, and a different native species most commonly occurring. At Outdoor Activity Center and Beecher Hills Greenway, the two sites with the most similarity, *Vitis* spp. and *Liriodendron tulipifera* were the most commonly occurring native species, although at Beecher Hills, *Liriodendron tulipifera* was the most commonly occurring native species and at Outdoor Activity Center it was the second most commonly occurring species. The two sites differed in the number of overall species and the percent native versus non-native of those species. At Outdoor Activity Center, 21 species were identified on the study plots (Table 3). Of these, five (23.8%) were non-native, all classified as invasive in Georgia. The other 16 species inventoried (76.2%) were native. The number of native and non-native individuals inventoried was almost evenly split at 50.2% and 49.8%, respectively.

Table 3. Outdoor Activity Center inventory

Species	Plantspp	Type	Total	%	Native	Invasive	Nonnative Cnt	% Nonnative	Native Cnt	% Native
Euonymus fortunei	winter creeper	vine	68	25.56	n	y	68	51.52		
Vitis spp.	grape	vine	56	21.05	y	na			56	42.11
Hedera helix	English ivy	vine	53	19.92	n	y	53	40.15		
Liriodendron tulipifera	tulip poplar	seedling	33	12.41	y	na			33	24.81
Parthenocissus quinquefolia	Virginia creeper	vine	15	5.64	y	na			15	11.28
Ligustrum sinense	Chinese privet	shrub	8	3.01	n	y	8	6.06		
Sanicula canadensis	black snakeroot	herbaceous	5	1.88	y	na			5	3.76
Phytolacca americana	pokeweed	herbaceous	4	1.50	y	na			4	3.01
Liquidambar styraciflua	sweetgum	seedling	4	1.50	y	na			4	3.01
Smilax bona-nox	greenbrier	vine	3	1.13	y	na			3	2.26
Toxicodendron radicans	poison ivy	vine	3	1.13	y	na			3	2.26
Eleagnus pungens	eleagnus	seedling	2	0.75	n	y	2	1.52		
Smilax laurifolia	laurel leaf greenbrier	vine	2	0.75	y	na			2	1.50
Passiflora lutea	yellow passionflower	vine	2	0.75	y	na			2	1.50
Uvularia grandiflora	bellwort	herbaceous	1	0.38	y	na			1	0.75
Mikania scandans	climbing hempweed	vine	1	0.38	y	na			1	0.75
Tiarella cordifolia	foam flower	herbaceous	1	0.38	y	na			1	0.75
Lonicera japonica	honeysuckle vine	vine	1	0.38	n	y	1	0.76		
Pinus sp.	pine	seedling	1	0.38	y	na			1	0.75
Cersis canadensis	redbud	seedling	1	0.38	y	na			1	0.75
unknown	unknown	na	1	0.38	na	na				
Viola sp.	violet	herbaceous	1	0.38	y	na			1	0.75
Total			266	100.00			132	100.00	133	100.00
% Total							49.81%		50.19%	

At Beecher Hills Greenway, 30 total species were identified of which eight (26.7%) were non-native (Table 4). Of these, seven were classified as invasive in Georgia. The number of non-native individuals accounted for 76% of the total plants inventoried. Although 22 native species (73.3%) were inventoried, native individual plants accounted for only 24% of the total plants inventoried.

Table 4. Beecher Hills Greenway inventory

Species	Plantspp	Type	Total	%	Native	Invasive	Nonnative Cnt	% Nonnative	Native Cnt	% Native
Hedera helix	English ivy	vine	129	38.86	n	y	129	51.60		
Lonicera japonica	honeysuckle vine	vine	58	17.47	n	y	58	23.20		
Ligustrum sinense	Chinese privet	seedling	39	11.75	n	y	39	15.60		
Liriodendron tulipifera	tulip poplar	seedling	30	9.04	y	na			30	37.97
Wisteria spp.	wisteria	vine	16	4.82	n	y	16	6.40		
Vitis spp.	grape	vine	16	4.82	y	na			16	20.25
Smilax bona-nox	greenbrier	vine	5	1.51	y	na			5	6.33
Acer negundo	box elder	seedling	4	1.20	y	na			4	5.06
unknown	unknown	seedling	3	0.90	na	na				
Albizia julibrissin	Chinese silktree	seedling	3	0.90	n	y	3	1.20		
Parthenocissus quinquefolia	Virginia creeper	vine	3	0.90	y	na			3	3.80
Dioscorea alata	Chinese yam	vine	2	0.60	n	y	2	0.80		
Euonymus fortunei	winter creeper	vine	2	0.60	n	y	2	0.80		
Acer barbatum	southern sugar maple	seedling	2	0.60	y	na			2	2.53
Athyrium asplenioides	southern lady fern	herbaceous	2	0.60	y	na			2	2.53
Persicaria pensylvanica	smartweed	herbaceous	2	0.60	y	na			2	2.53
Phytolacca americana	pokeweed	herbaceous	2	0.60	y	na			2	2.53
Prunus caroliniana	cherry laurel	seedling	1	0.30	n	n	1	0.40		
Acer sp.	maple	seedling	1	0.30	y	na			1	1.27
Acer rubrum	red maple	seedling	1	0.30	y	na			1	1.27
Bignonia capreolata	cross vine	vine	1	0.30	y	na			1	1.27
Carex sp.	sedge	sedge	1	0.30	y	na			1	1.27
Carya sp.	hickory	seedling	1	0.30	y	na			1	1.27
Clematis sp.	clematis	vine	1	0.30	y	na			1	1.27
Erechtites hieraciifolius	fireweed	herbaceous	1	0.30	y	na			1	1.27
Fagus grandifolia	beech	seedling	1	0.30	y	na			1	1.27
Liquidambar styraciflua	sweetgum	seedling	1	0.30	y	na			1	1.27
Pinus taeda	loblolly pine	seedling	1	0.30	y	na			1	1.27
Polystichum acrostichoides	christmas fern	herbaceous	1	0.30	y	na			1	1.27
Quercus sp.	oak	seedling	1	0.30	y	na			1	1.27
Toxicodendron radicans	poison ivy	vine	1	0.30	y	na			1	1.27
Total			332	100.00			250	100.00	79	100.00
% Total							75.99%		24.01%	

At Cascade Springs Nature Preserve 19 plant species were identified of which five (26.3%) were non-native (Table 5). Of these, three were classified as invasive in Georgia. The number of non-native individuals accounted for 54.3% of the total plants inventoried. The 14 native species identified were 73.7% of the total species inventoried and 45.7% of the individual plants inventoried.

Table 5. Cascade Springs Nature Preserve inventory

Species	Plantspp	Type	Total	%	Native	Invasive	Nonnative Cnt	% Nonnative	Native Cnt	% Native
Youngia japonica	false hawksbeard	herbaceous	37	31.62	n	n	37	58.73		
Ligustrum sinense	Chinese privet	seedling	21	17.95	n	y	21	33.33		
Calystegia sepium	wild morning glory	vine	12	10.26	y	na			12	22.64
Impatiens capensis	jewelweed	herbaceous	10	8.55	y	na			10	18.87
Acer spp.	maple	seedling	5	4.27	y	na			5	9.43
Liriodendron tulipifera	tulip poplar	seedling	5	4.27	y	na			5	9.43
Phylotacca americana	pokeweed	herbaceous	5	4.27	y	na			5	9.43
Vitis spp.	grape	vine	4	3.42	y	na			4	7.55
Pinus taeda	loblolly pine	seedling	3	2.56	y	na			3	5.66
Prunus caroliniana	cherry laurel	seedling	2	1.71	n	n	2	3.17		
Lonicera japonica	honeysuckle vine	vine	2	1.71	n	y	2	3.17		
Parthenocissus quinquefolia	Virginia creeper	vine	2	1.71	y	na			2	3.77
Smilax bona-nox	greenbrier	vine	2	1.71	y	na			2	3.77
Dioscorea alata	Chinese yam	vine	1	0.85	n	y	1	1.59		
Acer negundo	box elder	seedling	1	0.85	y	na			1	1.89
Acer rubrum	red maple	seedling	1	0.85	y	na			1	1.89
Carya tomentosa	mockernut hickory	seedling	1	0.85	y	na			1	1.89
Liquidambar styraciflua	sweetgum	seedling	1	0.85	y	na			1	1.89
Toxicodendron radicans	poison ivy	vine	1	0.85	y	na			1	1.89
unknown	unknown	seedling	1	0.85	na	na				
Total			117	100.00			63	100.00	53	100.00
% Total							54.31%		45.69%	

Of the four study sites, Hampton Greenway had the lowest plant richness (Table 6). Only ten species were identified at Hampton Greenway. Individuals of *L. sinense* were the most commonly occurring at 91% of total individuals inventoried. Two other non-native species were present, both invasive to Georgia. Therefore, non-native species were 30% of the total species present and 93.03% of the individual plants inventoried. Seven native species (70%) were identified of the total inventoried, representing just 6.97% of individual plants inventoried.

Table 6. Hampton Greenway inventory

Species	Common name	Type	Total	%	Native	Invasive	Nonnative Cnt	% Nonnative	Native Cnt	% Native
Ligustrum sinense	Chinese privet	seedling	183	89.27	n	y	183	97.86		
Phytolacca americana	pokeweed	herbaceous	7	3.41	y	na			7	50.00
unknown	unknown	seedling	4	1.95	na	na				
Acer negundo	box elder	seedling	2	0.98	y	na			2	14.29
Euonymus alatus	burning bush	seedling	2	0.98	n	y	2	1.07		
Lonicera japonica	honeysuckle vine	vine	2	0.98	n	y	2	1.07		
Vitis sp.	grape	vine	1	0.49	y	na			1	7.14
Acer sp.	maple sp.	seedling	1	0.49	y	na			1	7.14
Acer rubrum	red maple	seedling	1	0.49	y	na			1	7.14
Liriodendron tulipifera	tulip poplar	seedling	1	0.49	y	na			1	7.14
Parthenocissus quinquefolia	Virginia creeper	vine	1	0.49	y	na			1	7.14
Total			205	100.00			187	100.00	14	100.00
% Total							93.03%		6.97%	

Low evenness between the non-native and native species existed on all study sites due to a high number of non-native invasive individual plants in samples inventoried and a low number of native individual plants. At Outdoor Activity Center, 16 of the 21 species occurring were native and 13 of the 16 (81.3%) had five or fewer individuals inventoried. Of the 30 species on the quadrats at Beecher Hills Greenway, 22 species were native and 20 of those had an individual plant count of five or less. Abundance of native species was particularly low at Beecher Hills. Only one individual plant was found for 13 of the 22 native species. At Cascade Springs Nature Preserve, 19 species were identified of which 14 were native. Twelve of the native species had five or fewer individuals represented in the samples. At Hampton Greenway, the site with the lowest diversity, ten species were inventoried of which seven were native. Of the native species, six had two or fewer individuals represented in the inventory. Due to the reoccurring dominance of *L. sinense* present at Hampton Greenway, evenness among both non-native invasive and native species was extremely low.

Species becoming established at the urban sites were a mix of non-native and native species, with non-native invasive species more abundant than native species. Species richness was highest at Beecher Hills Greenway and lowest at Hampton Greenway. A few non-native and native species dominated each site with the majority of species represented by a low number of individuals. Dominant taxa were most likely indicative of seed sources in the immediately surrounding areas, including the non-native invasive species pervasive in residential and commercial landscapes. Non-native invasive species impacted richness of native species in two possible ways. First, during a long term invasion, survival rate of native seeds in the seed bank tends to be low. Second, there were few mature surviving native species producing seeds because they were outcompeted by the invasive species (Biggerstaff and Beck 2007; Miller and Gorchov 2004; Schiffman 2009).

Of the four sites inventoried two years after treatment for removal of *L. sinense* in urban Atlanta, three had significant abundance of *L. sinense* recurring. This abundance was indicative of *L. sinense*'s ability to outcompete native species, its lack of predators, and its prodigious propagation, along with the favorable climate and geography in the southeastern United States. It was also the result of mature fruiting *L. sinense* on and near study sites, providing constant new sources of seeds. In addition to the abundance of *L. sinense,* the most commonly occurring species at each site studied were non-native invasive. The dominant non-native invasive was also different at each site. At Outdoor Activity Center, *Euonymus fortunei,* an evergreen ornamental woody vine from Asia that can overcome and topple trees, was the most commonly occurring species at 25.6% of total individual plants. *Hedera helix*, an evergreen vine from Europe which can also topple trees, was the most common species at Beecher Hills Greenway and 38.9% of individual plants inventoried there. At Cascade Springs Nature Preserve *Youngia japonica*, an herbaceous species from Japan (Miller 2003) was the most commonly occurring species representing 31.6% of individuals inventoried.

Urban versus rural differences in vegetative composition

The second goal of this research was to add to the body of literature regarding invasive species removal and site recovery in urbanized areas through a comparative analysis with similar research conducted by Hanula, Horn and Taylor (2009) in a rural area of Georgia. The following research questions supported this goal.

1. Were more non-native invasive species colonizing urban floodplains than rural floodplains two years after removal of *L. sinense*?
2. Did native species colonizing urban floodplains exhibit less diversity compared to native species colonizing rural floodplains?
3. How was the diversity of the native species reflected in species richness and evenness?

Comparative analysis was performed using urban research data collected along transect points (see Appendix B for a table of native species inventoried). As with data collected along meter lines, non-native invasive species dominated data collected along urban transect points although point data differed somewhat from data collected at meter lines. Nineteen taxa were found on transect points inventoried at urban sites. Six (31.6%) of the 19 species were non-native. Five of the six non-native species were classified as invasive. The most commonly inventoried species on all urban sites combined was *Hedera helix* at 20.3% of individual plants. *L. sinense* was the second most common species inventoried at 16.5% (Table 7). Both are non-native invasive species.

Table 7. Ten most common species inventoried on urban study plots (point data)

Species	Plantspp	Type	Native	Invasive	% of Total
Hedera helix	English ivy	vine	n	y	20.25
Ligustrum sinense	Chinese privet	seedling	n	y	16.46
Vitex spp.	Grape spp	vine	y	na	11.39
Liriodendron tulipifera	Tulip poplar	seedling	y	na	11.39
Parthenocissus quinquefolia	Virginia creeper	vine	y	na	7.59
Lonicera japonica	Honeysuckle vine	vine	n	y	6.33
Euonymus fortunei	Winter creeper	vine	n	y	5.06
Youngia japonica	False hawksbeard	herb	n	n	3.80
Phytolacca americana	Pokeweed	herbaceous	y	na	3.80
Liquidambar styraciflua	Sweetgum	seedling	y	na	2.53

For the rural study, 42 species of plants were inventoried on all plots combined. The ten most abundant species occurring were native to the ecoregion (Table 8). Of the 42 total species inventoried, nine species (21.4%) were non-native of which three were classified as invasive. Three taxa inventoried at rural sites could not be categorized as native or non-native due to lack of species specific information.

Table 8. Ten most common species inventoried on rural study plots

Species	Plantspp	Type	Native	Invasive	% of Total
Acer negundo	Box elder	seedling	y	na	16.50
Erechtites hieracifolia	Fireweed	herbaceous	y	na	16.50
Phytolacca americana	Pokeweed	herbaceous	y	na	12.88
Urtica dioica	Nettle	herbaceous	y	na	11.07
Vitis rotundifolia	Grape spp	vine	y	na	10.26
Viola spp.	Violet spp	herbaceous	y	na	4.63
Acer leucoderme	Chalk bark maple	seedling	y	na	3.62
Parthenocissus quinquefolia	Virginia creeper	vine	y	na	2.41
Fraxinus pennsylvanica	Green ash	seedling	y	na	2.01
Toxicodendron radicans	Poison ivy	vine	y	na	2.01

Source: Data provided by J. Hanula, USDA Forest Service

Urban sites exhibited a higher percentage of non-native taxa to total taxa, at 31.6% versus rural sites with 21.4%. While the difference in non-native taxa was ten percentage points between urban and rural sites, the abundance of non-native taxa on urban sites was significantly higher than on rural sites. Non-native taxa comprised 53.2% of individual plants inventoried at urban sites, and only 5.4 % of individuals at rural sites. Abundance of non-native invasive species on urban sites was 49.4%, a significant portion of overall native and non-native species abundance. Abundance of non-native invasive species on rural study sites was only 2.4%. While all ten of the most commonly occurring species on rural sites were native to the ecoregion, five of the ten most commonly occurring species on urban sites were non-native.

An interesting characteristic of the data was the occurrence of vines. For urban sites, five of the ten most commonly occurring taxa were vines, of which three, *Hedera helix*, honeysuckle vine (*Lonicera japonica*), and winter creeper (*Euonymus fortunei*) were non-native invasive species frequently used in urban landscaping. The other two vine taxa, *Vitis* spp. and Virginia creeper (*Parthenocissus quinquefolia*) were native early colonizers in disturbed southeastern forests and woodlands. Both native vines were also found in the ten most commonly occurring taxa on the rural sites along with an additional native vine, poison ivy (*Taxicodendron radicans*).

Another characteristic of the inventories taken at both urban and rural sites was the dominance of bare points, or points with no plants occurring. Bare points were more significant on urban sites, comprising 83.5% of the total points inventoried (Table 9). They were 58.8% of the points in the rural study (Table 10). This was another indication of lack of seed sources in the urban seed bank and the vulnerability of treated areas to colonization by the dominant mature seed producing taxa in the immediate areas.

Table 9. Bare points at urban study sites

Urban Sites	Bare Points	Total Points	%
Cascade	110	120	91.667
Beecher	93	120	77.500
Hampton	106	120	88.333
OAC	92	120	76.667
Totals	401	480	83.542

Table 10. Bare points at rural study sites

Rural Sites	Bare Points	Total Points	%
BG	149	281	53.025
SC	176	290	60.690
SS	194	300	64.667
WS	163	308	52.922
Totals	682	1179	58.846

Source: Data provided by J. Hanula, USDA Forest Service

The most commonly occurring species on rural sites was *Acer negundo* at 16.5% of all individuals inventoried compared to the most commonly occurring urban species, *Hedera helix*, at 20.3% of individuals inventoried. The most abundant non-native species occurring on rural sites was beefsteak plant (*Perilla frutescens*), an invasive herbaceous species native to India (Weakley 2011). It was the twelfth most abundant species inventoried on rural sites, and just 1.6% of total plants inventoried.

A graphic representation of non-metric multidimensional scaling ordination supported the difference in species composition between the study sites. Rural sites were grouped together, showing more similarity between them. The urban sites were grouped together as well. Urban and rural sites were grouped separately from one another although one urban site grouped closely to the rural sites (Figure 10). The grouping for both rural and urban sites differed from the grouping of sites not invaded by *L. sinense*, indicating the newly established plant communities are dissimilar to the desired future condition species composition inventoried in the rural study.

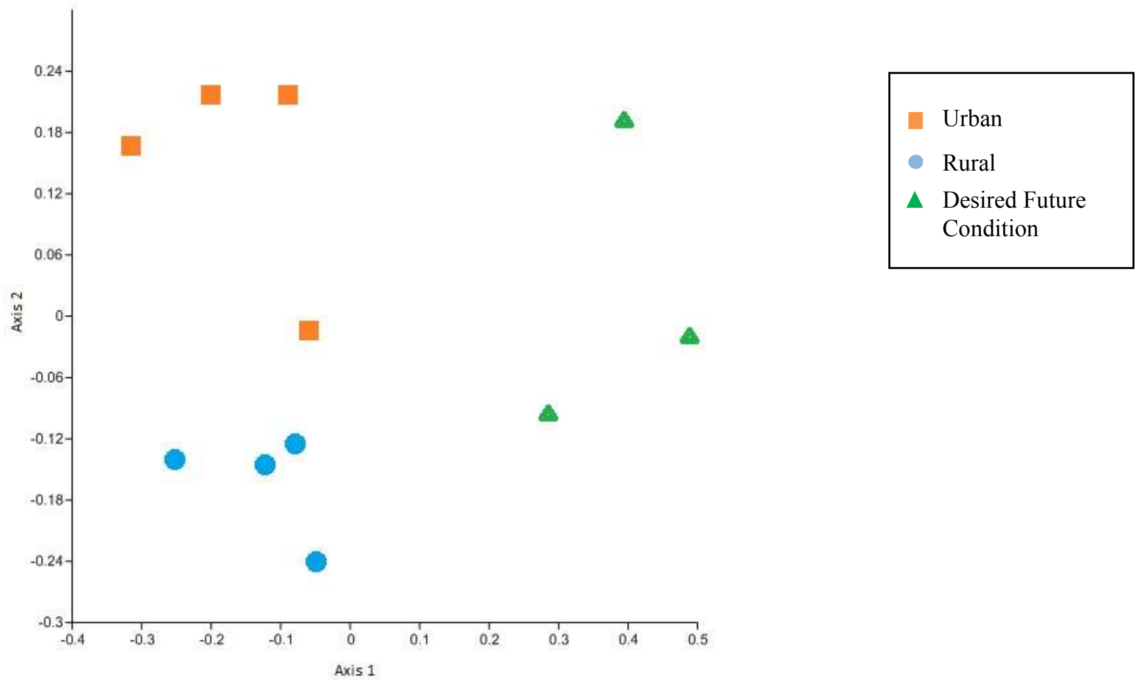

Figure 10. Diagram of nonmetric multidimensional scaling ordination graph for urban and rural study sites

For all taxa inventoried, rural data exhibited greater species richness with 42 different species, when compared to urban sites which had a total of 19 species. The diversity profile is a visual representation of the native taxa in rural and urban areas and their species abundance, and exhibits distinct curves for each (Figure 11).

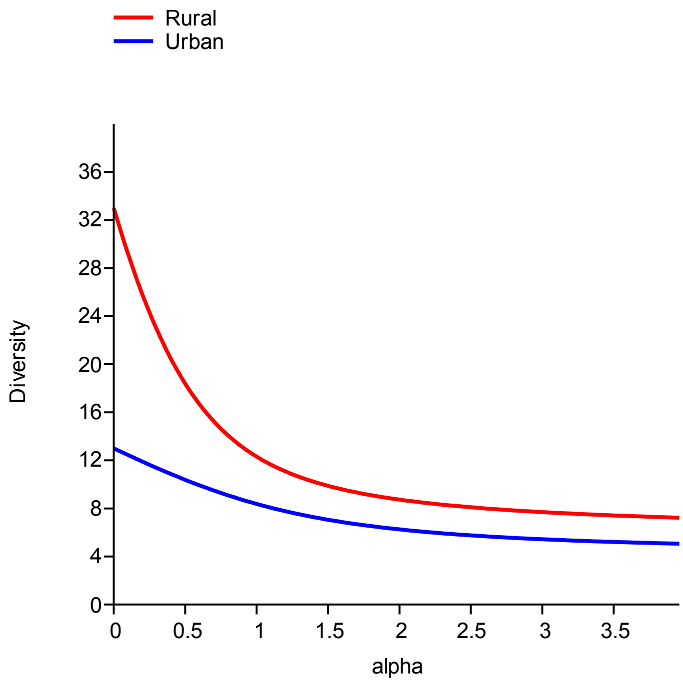

Figure 11. Diagram of diversity profile for native taxa inventoried at rural and urban sites

For comparison of species diversity, richness, and evenness, non-native taxa were removed from the urban and rural plant inventories and native taxa examined (Table 11). Several native taxa dominated the urban and rural inventories although the dominant species differed for each study. *Vitis* spp., *Liriondendron tulipifera,* and *Parthenocisus quinquefolia* were the most dominantly occurring urban species, while *Acer nugundo*, fireweed (*Erechtites hieraciifolius*), pokeweed (*Phytolacca Americana*), stinging nettle (*Urtica dioica*), and *Vitis* spp. were the most dominant species in the more rural areas studied by Hanula et al (2009). Six native species were common to both studies. The most abundant species in common were early colonizing vines of disturbed areas, *Vitis* spp. and *Parthenocisus quinquefolia.*

Table 11. Most abundant native species in urban and rural studies (point data)

URBAN		
Plant species	**Count**	**% of Total**
Vitis spp.	9	24.32
Liriodendron tulipifera	9	24.32
Parthenocissus quinquefolia	6	16.22
Phytolacca americana	3	8.11
Liquidambar styraciflua	2	5.41
Uvularia grandifolia	1	2.70
Sanicula canadensis	1	2.70
Smilax bona-nox	1	2.70
Impatiens capensis	1	2.70
Quercus sp.	1	2.70
Pinus sp.	1	2.70
Persicaria pensylvanica	1	2.70
Calystegia sepium	1	2.70

RURAL		
Plant species	**Count**	**% of Total**
Acer negundo	82	16.50
Erechtites hieraciifolius	82	16.50
Phytolacca americana	64	12.88
Urtica dioica	55	11.07
Vitis spp.	51	10.26
Viola spp.	23	4.63
Acer leucoderme	18	3.62
Parthenocissus quinquefolia	12	2.41
Fraxinus pennsylvanica	10	2.01
Toxicodendron radicans	10	2.01
Campsis radicans	10	2.16
Carex spp.	8	1.72
Persicaria pensylvanica	7	1.51

Source: Rural data provided by J. Hanula, USDA Forest Service

To determine whether native species colonizing the urban floodplains exhibited less species richness than native species colonizing rural floodplains, an index of similarity was calculated between the two samples of native species. An index of zero indicates no similarity between two samples and an index of one indicates the samples are exactly the same (MacDonald 2003). There were 33 unique native species in the rural study and 13 unique native species in the urban study. Six species were common to both samples. The index of similarity was 0.26 for urban and rural native species becoming established which indicated low similarity between rural and urban sites.

The Shannon index (*H'*), which considers species richness and evenness, was 2.5 for the rural study and 2.1 for the urban study. This indicated moderate richness and evenness for the rural sites, while the urban sites exhibited lower richness and evenness. However, the statistical estimate of evenness alone indicated higher evenness for the urban study at 0.64 compared to

0.37 for the rural study. This result could be an indication that urban sites do not have as many seed sources for native species which impacts their colonization and presents a continual challenge for urban natural areas. The most abundant native taxa on urban sites were *Vitis* spp. at nine individuals. Over half of the native taxa included just one individual documented. Although species richness was greater for rural sites than for urban sites, the range of individuals counted for each taxon was much wider on rural sites, influencing the measure of evenness for the data.

Variation in species richness and evenness between rural and urban data was further underscored when all taxa inventoried were ranked. As with the native taxa, percentages of individuals for both native and non-native species were high for the top several species then decreased dramatically especially on urban sites. The second most common species, *Hedera helix*, comprised 19.8% of the total urban inventory while the third most common species, *Vitis* spp., was 8.4% of the inventory, a drop of over 11 percentage points. The five most common species on rural sites were more evenly distributed, within 6.25 percentage points.

The species composition in both urban and rural areas two years after treatment to remove *L. sinense* was mainly represented by early colonizers of disturbed areas, typical of the southeastern Piedmont region (Hanula, Horn, and Taylor 2009). Significantly more non-native invasive species were colonizing urban floodplains than rural floodplains two years after removal. Additionally, non-native invasive species at urban sites exhibited high abundance compared to the abundance of native species. Native species at urban sites exhibited less diversity, richness, and evenness compared to native species on rural sites.

CHAPTER 4

CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE STUDIES

The importance of studying urban ecosystems as unique habitats substantially impacted by human activities was supported by this research. Although similar methods were used to remove *Ligustrum sinense* occurring in urban and rural floodplains within the same ecoregion of Georgia, the returning species composition after removal was distinctly different at urban and rural sites. The significant challenge of removing an invasive species within the urban environment was especially apparent in the high percentages of non-native invasive species becoming established. Not only was *L. sinense* recolonizing the urban sites at a significant rate, but it appeared that other non-native invasive species were colonizing some of the areas left bare by its removal. Species commonly used in urban landscaping, particularly *Hedera helix*, *Euonymus fortunei*, and *Lonicera japonica,* were occurring at similar rates to the rate at which *L. sinense* was reoccurring. Many non-native invasive species were thriving in the landscape surrounding the urban parks and appeared to be a continued source of propagules to the treated sites within the urban parks and greenways.

In addition to the prolific sources of non-native invasive species propagules in the immediate study area, the absence of sources for native species propagules was striking. While some mature native individuals were observed in the area, they were less abundant than mature non-native invasive plants. Most observed mature native species were annuals that produce seeds and quickly become established, or woody vines that produce fruits eaten and dispersed by birds and other animals. Native perennial forbs and shrubs were notably absent. Therefore efforts by Trees Atlanta, as well as other restoration activities, need to consider protection of recurring native species when planning maintenance and further treatment for removal of the non-native invasive species. Another observation from the research sites was the high number of bare areas observed. This was due to the considerable extent of *L. sinense* removed. Long term dominance of *L. sinense* in the landscape likely caused mortality of native seeds in the seed bank, resulting in slow colonization of sites by native species. In some areas, application of herbicide may also have slowed colonization of native species. Bare areas should be monitored closely at varying times of the year for the establishment of native species or further dominance of non-native species. Occurrence of increased abundance of native species, or more non-native invasive individuals, in bare areas should influence ongoing maintenance and site recovery plans.

The dominance of non-native invasive species in this urban study points to an essential component of urban restoration efforts: public education about the importance of healthy urban ecosystems and the detrimental effects of non-native invasive species on those ecosystems. It is recommended that Trees Atlanta enhance the educational aspect of their programs to include active management of *L. sinense* and other invasive species by proximate private property owners in order to alter the significant distribution of invasive species surrounding public lands and achieve the goal of eradication. Outreach to residents and businesses, for example through Neighborhood Planning Unit monthly meetings and educational web sites, is essential. Landowners are important stakeholders in the efforts to stop non-native invasive species from propagating and negatively impacting urban ecosystems. Commitment from private property owners to eradicate non-native species is especially important because it may be easier for them to monitor their property on an ongoing basis and follow up when the invading species reoccurs, than it is for governmental and non-profit organizations that are monitoring large areas and which are subject to changes in staff and funding.

Opportunities for future research include better understanding the effects of the herbicidal chemicals and surfactants utilized on *L. sinense*, especially during the broadcast spraying of new growth during follow up treatment, as well as chemicals used to treat other invasive species. The extensive presence of invasive species requires use of chemicals to manage them but the consequences of doing so must be better understood so that restoration science can determine the appropriate balance between chemical utilization and allowing the return of biodiversity to an area. During this research, desirable native species were observed on study plots and in the immediate area. Any detrimental effects of broadcast spraying on native species should be fully understood if this treatment method continues to be utilized to control invasive species regrowth.

Attitudes and perceptions of property owners and stakeholders is another area for future research. In order to maximize the benefits of educating people about the negative effects of non-native invasive species, the current level of general knowledge about and attitudes towards the environment need to be understood. Understanding public attitudes and perceptions could also assist in formulating public policies and implementing regulations to control the introduction of new non-native species which may become invasive in the future. Stricter regulation and enforcement may be necessary to mitigate the negative consequences of bringing new non-native species into the country or intentionally distributing non-native invasive species into an area. Economics of invasive species is another opportunity for future research. Often people become more aware of an issue if an economic impact is clear. Most of the non-native invasive species found during this research were widely available in the nursery trade. An interesting study would be to measure the income generated from wholesale and retail sales of non-native invasive species compared to the millions of tax dollars spent annually trying to control them.

The complexities involved with restoring urban ecosystems to a more healthy state cannot be overemphasized. Plant communities in this study were significantly altered by long term invasions of *Ligustrum sinense*. Study sites exhibited an influx of new invading species, high levels of bare areas, and they differed significantly from desired future condition sites. Proximity to continual sources of non-native invasive species propagules indicated that sites will require ongoing invasive species removal efforts for many years. Time is a significant consideration for invasive removal efforts and for further research on the results of those efforts. Two years is insufficient to expect site recovery. Successional and abiotic factors will continually affect the species composition in treated areas; the current composition of early colonizers will give way to other species which will enhance biodiversity over time. Determining whether native species will become more abundant, and biodiversity increase, is a long term project worthy of continued research. Additionally, emphasis for this research was placed on plant taxa, which are only one component of biodiversity. Healthy urban ecosystems provide habitat to a variety of other organisms, including insects, birds, and other animals. In the long run, the richness of biodiversity present will be a determining characteristic of whether non-native species eradication and urban ecosystem restorations are successful.

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APPENDICES

Appendix A: Urban species inventoried (using line method)

Species	Plantspp	Type	Total	% Total	Native	Invasive
Ligustrum sinense	Chinese privet	seedling	251	27.28	n	y
Hedera helix	English ivy	vine	182	19.78	n	y
Vitis spp.	grape	vine	77	8.37	y	na
Euonymus fortunei	winter creeper	vine	70	7.61	n	y
Liriodendron tulipifera	tulip poplar	seedling	69	7.50	y	na
Lonicera japonica	honeysuckle vine	vine	63	6.85	n	y
Youngia japonica	false hawksbeard	herbaceous	37	4.02	n	n
Parthenocissus quinquefolia	Virginia creeper	vine	21	2.28	y	na
Phytocca americana	pokeweed	herbaceous	18	1.96	y	na
Wisteria spp.	wisteria	vine	16	1.74	n	y
Calystegia sepium	wild morning glory	vine	12	1.30	y	na
Impatiens capensis	jewelweed	herbaceous	10	1.09	y	na
Smilax bona-nox	greenbrier	vine	10	1.09	y	na
unknown	unknown	seedling	9	0.98	na	na
Acer spp.	maple spp	seedling	7	0.76	y	na
Acer negundo	box elder	seedling	7	0.76	y	na
Liquidambar styraciflua	sweetgum	seedling	6	0.65	y	na
Pinus spp.	pine	seedling	5	0.54	y	na
Sanicula canadensis	black snakeroot	herbaceous	5	0.54	y	na
Toxicodendron radicans	poison ivy	vine	5	0.54	y	na
Acer rubrum	red maple	seedling	3	0.33	y	na
Albizia julibrissin	Chinese silktree	seedling	3	0.33	n	y
Dioscorea alata	Chinese yam	vine	3	0.33	n	y
Prunus caroliana	cherry laurel	seedling	3	0.33	n	n
Acer barbatum	southern sugar maple	seedling	2	0.22	y	na
Athyrium asplenioides	southern lady fern	herbaceous	2	0.22	y	na
Eleagnus pungens	eleagnus	seedling	2	0.22	n	y
Euonymus alata	burning bush	seedling	2	0.22	n	y
Passiflora lutea	yellow passionflower	vine	2	0.22	y	na
Persicaria pensylvanica	smartweed	herbaceous	2	0.22	y	na
Smilax laurifolia	laurel leaf greenbrier	vine	2	0.22	y	na
Bignonia capreolata	cross vine	vine	1	0.11	y	na
Carex sp.	sedge sp.	herbaceous	1	0.11	y	na
Carya tomentosa	mockernut hickory	seedling	1	0.11	y	na
Carya sp.	hickory sp.	seedling	1	0.11	y	na
Cersis canadensis	redbud	seedling	1	0.11	y	na
Clematis sp.	clematis	vine	1	0.11	y	na
Erechtites hieraciifolius	fireweed	herbaceous	1	0.11	y	na
Fagus grandifolia	beech	seedling	1	0.11	y	na
Mikania scandans	climbing hempweed	vine	1	0.11	y	na
Polystichum acrostichoides	christmas fern	herbaceous	1	0.11	y	na
Quercus sp.	oak sp.	seedling	1	0.11	y	na
Tiarella cordifolia	foam flower	herbaceous	1	0.11	y	na
Uvularia grandiflora	bellwort	herbaceous	1	0.11	y	na
Viola sp.	violet	herbaceous	1	0.11	y	na
Total			920	100.00		

Appendix B: Urban species inventoried (using point method)

Species	Plantspp	Type	Total	% Total	Native	Invasive
Hedera helix	English ivy	vine	16	20.25	n	y
Ligustrum sinense	Chinese privet	seedling	13	16.46	n	y
Vitis spp.	grape	vine	9	11.39	y	na
Liriodendron tulipifera	tulip poplar	seedling	9	11.39	y	na
Parthenocissus quinquefolia	Virginia creeper	vine	6	7.59	y	na
Lonicera japonica	honeysuckle	vine	5	6.33	n	y
Euonymus fortunei	winter creeper	vine	4	5.06	n	y
Youngia japonica	false hawksbeard	herbaceous	3	3.80	n	n
Phytolacca americana	pokeweed	herbaceous	3	3.80	y	na
Liquidambar styraciflua	sweetgum	seedling	2	2.53	y	na
Wisteria sp.	wisteria	vine	1	1.27	n	y
Uvularia grandifolia	bellwort	herbaceous	1	1.27	y	na
Sanicula canadensis	black snakeroot	herbaceous	1	1.27	y	na
Smilax bona-nox	greenbrier	vine	1	1.27	y	na
Impatiens capensis	jewelweed	herbaceous	1	1.27	y	na
Quercus sp.	oak sp.	seedling	1	1.27	y	na
Pinus sp.	pine sp.	seedling	1	1.27	y	na
Persicaria pensylvanica	smartweed	herbaceous	1	1.27	y	na
Calystegia sepium	wild morning glory	vine	1	1.27	y	na
Total			79	100.00		