

## ABSTRACT

VIDRA, REBECCA LYNN. Implications of exotic species invasion for restoration of urban riparian forests. (Under the direction of Dr. Theodore H. Shear).

Urban forests serve as remnant natural areas in otherwise degraded landscapes. Yet, these forests are commonly threatened by invasion of exotic plant species, which may compromise the structure and ecological functions of native communities. Restoration of these forests will inevitably require removal and continued control of exotic species invasions.

I focused my research on urban riparian forests within Raleigh and Cary, North Carolina, USA. My goals are to understand the impacts of exotic species invasion on these forest communities, identify factors that make these sites susceptible to invasion, and make appropriate restoration recommendations. To address these goals, three major studies structured my research.

First, I tested two major paradigms of current exotic species ecology using observational data from 23 urban riparian buffers. The competition paradigm was borne out by negative relationships between exotic and native species richness ( $r = -0.66$ ,  $p = 0.0009$ ). I also detected shifts in species composition away from native woody species in sites that have been invaded by exotic species. As suggested by the resource availability paradigm, certain environmental conditions do seem to favor exotic species in this study system. While light availability was not significantly correlated to invasion of exotic species, several measures of soil fertility were negatively correlated to native species and positively correlated to exotic species. Therefore, efforts to reduce fertilization from adjacent and upstream landscapes should be part of any restoration plan.

My second approach involved conducting an experimental removal study at the North Carolina Museum of Art. I compared the species composition and native species recovery in

three treatments: control, initially removing all vegetation and allowing regeneration, and initial removal with repeated removal of new exotic recruits. While the repeated removal treatment featured a different species composition than either of the other two treatments, this difference can be attributed to the lack of exotic species, not to the recovery of native species. These results indicate that repeated removal is necessary to prevent invasion in urban forests but that native supplemental plantings may be necessary to create a diverse community.

My third approach involved evaluating the influence of landscape structure on patterns of exotic species invasion in urban riparian buffers. I found that invasion, measured as total percent cover of exotic species, is highest in the narrowest buffers. Buffers surrounded by more forest canopy cover are also more heavily invaded than those surrounded by less canopy cover. These findings suggest that processes outside the buffer, such as bird dispersal and vegetative spreading of ornamental plantings, may contribute to the invasion inside the buffer.

Exotic species invasion does have implications for restoration of urban forests. Not only does invasion threaten native species richness, but native woody species appear to be most at risk for extirpation from these sites, with potential consequences for long-term forest succession. Because these forests are invaded by a suite of exotic plants, targeted efforts to remove those that are likely outcompeting native species is a good first step. Managing land use practices outside of the forests, while difficult, is necessary to reduce both the stress on the forest (e.g., fertilization) and invasion opportunities (e.g., bird dispersal, ornamental plantings). The results of this research will help managers identify sites at future risk for invasion and focus efforts on managing both the exotic species and site conditions to restore the health of these ecologically valuable forest communities.

**IMPLICATIONS OF EXOTIC SPECIES INVASION FOR RESTORATION OF  
URBAN RIPARIAN FORESTS**

by

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A dissertation submitted to the Graduate Faculty of North Carolina State University  
in partial fulfillment of the requirements for the Degree of Doctor of Philosophy

**FORESTRY**

**Raleigh**

**2004**

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## BIOGRAPHY

Rebecca Lynn Vidra was born on October 17, 1974 in Cleveland, Ohio to Andrew and Lynn Vidra. She grew up along Lake Erie and landed her first environmental science position in 1982 as her father's field assistant. Along with her younger sister, Jen, she collected water samples, lost test tube tops, and chased crayfish along many tributaries of Lake Erie. This field experience motivated her interest in environmental issues and foreshadowed her future field work in less than desirable field locations.

After almost 22 years in Ohio, Rebecca fled south to Florida immediately after her June 1996 graduation from the School of Natural Resources at The Ohio State University. Relieved to be in the subtropics, she worked as a naturalist for the Conservancy of Southwest Florida and then as an Americorps volunteer for The Nature Conservancy in the Florida Keys. Logging over 100 "work" dives, she was primarily responsible for training volunteers to monitor coral reefs, fish populations, and water quality. She also investigated benthic recovery on several large ship groundings in the Florida Keys and became interested in the field of restoration ecology.

Armed with an exciting research interest, Rebecca entered the University of North Carolina at Chapel Hill's Curriculum in Ecology in the Fall of 1998. There, she studied both the successes and failures of past reef restoration efforts and continued to be involved in field work in Florida. After completing a Masters degree in 2000, she decided to nourish her land grant roots with a transition to North Carolina State University.

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Rebecca has been fortunate to teach honors seminars as a *Preparing the Professoriate* Fellow and recently taught her first undergraduate course, *Environmental Conservation and Global Change*, at UNC-CH. Upon completion of her PhD, Rebecca will pursue research and teaching opportunities in restoration ecology and research ethics with a Mellon Postdoctoral Fellowship at Duke University. Her most pressing challenge will commence in May with the birth of her first child, who is sure to provide many interesting research questions.

## ACKNOWLEDGEMENTS

I have been fortunate to receive support from various people and organizations throughout my graduate career at North Carolina State University. For making my graduate lifestyle possible, I would first like to acknowledge the NCSU Department of Forestry, the NCSU Alumni Association, the NCSU Graduate School, the Garden Club of America, the National Science Foundation, and the Microsoft Corporation. Each of these organizations provided funding for various aspects of my research and training.

My intellectual growth has been encouraged by my advisory committee as well as by many faculty members in departments throughout the university. Dr. Ted Shear, who served as my major advisor, allowed me to develop my own research questions while guiding me towards the development of a project with achievable and interesting goals. His support as a mentor and as a friend was invaluable to me throughout this process. Dr. George Hess' interest in my work is greatly appreciated and I thank him for including me in the *Greenways for Wildlife* project. The positive attitude that Dr. Tom Wentworth brings to his teaching and his interest in ecology served as inspiration for me as I trudged through the sewers of Raleigh and Cary. Dr. Jon Stucky contributed many thoughtful ideas on experimental design and prevented me from spending too many hours wandering around the herbarium by identifying many of my plant specimens. Dr. Nick Haddad regularly rescued me from the morass of the research process with his enthusiasm for my work. I also thank my previous advisors, Dr. Seth Reice of UNC-CH and Dr. Rosanne Fortner of The Ohio State University, for guiding my intellectual development.

Logistic support was provided by a suite of fellow graduate students and friends, who chopped down exotics with me on a regular basis. For this help, I thank Julie Gibson, Melissa Ruiz, Sonia Weeks, Lyn Samartino, Bethany Hudnutt, Duncan Quinn, Luis Carrasco, Kim

Matthews, Sarah Luginbuhl, Chris White, Emily MacFadyen and Frank Koch. Carol Yang served as my field assistant during the summer of 2002 and her enthusiasm kept both of us mostly sane.

Dr. Chris Moorman, Jamie Mason, and Kristen Sinclair provided helpful insights for integrating my work with the *Greenways* project. Alexander Krings cheerfully assisted with my plant identification and the creation of my voucher collection. I also would like to thank Marcia Gumpertz, Cavell Brownie, David Dickey, and Joy Smith in the Statistics Department for channeling my enthusiasm for statistics into multiple multiple regressions.

Emotional support is an essential part of anyone's graduate career and I am fortunate to have been bolstered by many good friends. My UNC peers, especially Michelle Cawley and Eileen Vandenburg, have been quite supportive throughout my many graduate project transitions. My fellow graduate students at NCSU certainly made my commutes worthwhile with their diverse perspectives on conservation. In particular, I thank Luis Carrasco for his patience and good humor as he put up with my chaos while sharing an office/closet.

My family has supported me in numerous ways, perhaps most importantly by not incessantly asking when I am going to finish. I thank them for their encouragement and confidence in me. My husband, Dr. Aaron Moody, is primarily responsible for my success as a graduate student. He has served as my shadow committee member, helping me to clarify my thoughts, editing my manuscripts, reviewing my statistics, and continuing to challenge me. He has trudged out into the field and tended to my endless tick, chigger, and mite infestations. When I really wanted to just give up, he whisked me away to Fiji and New Zealand for some nature rejuvenation. I am grateful for all the ways that he supports me and this work and look forward to returning the favor throughout our careers.

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## **CHAPTER 1**

### **IMPLICATIONS OF EXOTIC SPECIES INVASION FOR RESTORATION OF URBAN RIPARIAN FORESTS: AN INTRODUCTION**

## RESEARCH CONTEXT

### Exotic species invasion and restoration

Restoration of disturbed ecosystems has become an important management tool for conserving species, communities, habitats, and ecosystem functions. A current focus of ecological restoration research is to understand and manipulate site dynamics so that both the structure and function of a system can recover (Jackson et al. 1995, Dobson et al. 1997). Yet, in both terrestrial and aquatic systems, invasive exotic species can interfere with this recovery process, ultimately altering community development of restoration sites (Richter 1997). Therefore, the management of exotic species and the prevention of future invasion are significant elements of many restoration plans (Palik et al. 2000, D'Antonio and Meyerson 2003).

The invasion of exotic species is leading to a homogenization of the world's flora and fauna (Vitousek et al. 1997, Lockwood and McKinney 2001) and poses a significant threat to biodiversity (Wilson 1992, Wilcove et al. 1998) and ecosystem functions (e.g., Vitousek 1986, Chapin et al. 1997). The economic impacts of exotic species invasion are also considerable, costing billions of dollars annually in the United States alone (Pimentel et al. 2000).

Invasion can degrade sites by altering the physical template (e.g., *Hydrilla sp.* in wetlands), interfering with biotic interactions (e.g., algal mats on coral reefs), or disrupting the natural disturbance regime (e.g., *Melaleuca quinquenervia* in the Florida Everglades). However, given the large number of exotic species in both terrestrial and aquatic systems, ecosystem managers need to prioritize removal and prevention efforts (Coblentz 1990, Westman 1990). Further research is necessary to determine which species have significant impacts on native diversity and ecosystem function and to evaluate the potential recovery of the system once these species are removed (Byers et al. 2002).

While many studies have examined life history characteristics to predict the invasion potential of certain exotic species (e.g., Bazzaz 1986), there is also great interest in understanding site dynamics that either encourage or resist invasion by exotics (Drake et al. 1989, Hobbs and Humphries 1995, Byers et al. 2002, D'Antonio and Kark 2002). It is critical to identify the scale of these processes to better focus prediction and management of future exotic species invasion (DeFerrari and Naiman 1994, With 2002). The results of such studies will benefit restoration projects with recommendations for preventing future exotic species invasion (Coblentz 1990, Byers et al. 2002).

Underscoring the ecological and economic threats of exotic species, former U.S. President Clinton issued Executive Order 13112 in 1999. This order directs federal agencies to “provide for restoration of native species and habitat conditions in ecosystems that have been invaded” and “to conduct research on invasive species...to provide for environmental sound control of exotics.” It is now time to direct research efforts towards developing solutions for preventing exotic species invasion and restoring currently invaded sites (Byers et al. 2002).

### **Invasion of urban forest ecosystems**

While the majority of exotic species research has focused on human-disturbed ecosystems, little attention has been paid to the invasion of urban forests by exotic plants. These relatively small natural areas may serve as refuges of biodiversity in otherwise degraded landscapes (Miller and Hobbs 2002). In particular, riparian forests are increasingly representative of the majority of open space in urban areas (Wenger and Fowler 2000). Often unsuitable for development due to steep slopes and frequent flooding, these forests provide many important ecological functions, from providing wildlife habitat (Keller et al. 1993, Kilgo et

al. 1998) to filtering sediment and pollution (Welsch 1991). However, these urban forests are often subject to higher invasion rates by exotic species due to frequent internal and external disturbances, close proximity to exotic seed sources, and high edge to area ratios. Restoration efforts are needed to sustain the species diversity and functions of these forests. These efforts will inevitably require removal or control of invasive exotic species (Coblentz 1990, Miller and Hobbs 2002).

## **RESEARCH QUESTIONS**

The goal of my research is to provide restoration recommendations for prioritizing removal and preventing future invasions of exotic species in urban riparian forests. I use the term “exotic species” to designate invasive species of concern in my study area. All of these species are non-indigenous to the United States, with the majority of them native to eastern Asia. My research is structured by the following three ecological questions:

### **What are the consequences of exotic species invasion for urban forest communities?**

Because of the great conservation interest in exotic species invasion, significant research efforts have been directed towards understanding the impacts of exotics on native ecosystems (see review in Levine et al. 2003). Several paradigms are emerging from this body of work, including the competition and resource availability paradigms. The competition paradigm suggests that exotic species may outcompete native species and alter species composition of ecological communities. To determine how these exotic species have shifted native species composition, ecologists have taken two general approaches. Correlative approaches have been widely used to observe the post-invasion community structure and to understand the

relationship between exotic and native species richness. These studies use a natural experiment approach (*sensu* Diamond 1986): by observing existing patterns, we can develop ideas about the relationship between pattern and process. The most widely used study design is to collect plot-level data on species richness and diversity in areas that have been invaded and compare these data to data collected in uninvaded areas (e.g., Woods 1993, Hutchinson and Vankat 1997). However, in some areas there may be no truly “uninvaded” areas and studies must then focus on the relationship between exotic species abundance and native species diversity. I address these challenges in Chapter 2 by describing the results of a descriptive, correlative study and placing them in the context of the competition paradigm.

A major limitation of correlative studies is that they do not provide evidence of competition, although we assume that exotic species are able to outcompete natives because of their particular life history characteristics (Bazzaz 1986). However, to determine if the exotics are actually suppressing native species instead of occupying space where no natives previously existed, removal studies are necessary (e.g., Meekins and McCarthy 1999, Gould and Gorchov 2000).

If an exotic species is experimentally removed from a site, subsequent changes in vegetation may indicate the potential for recovery of native species (Byers et al. 2002). Initial results from exotic species removal provide useful evidence of the potential suppression effects of the exotics and can be used to suggest longer-term effects. However, long-term studies are needed to fully understand the trajectory of recovery in heavily invaded systems and restoration sites could provide excellent systems for these studies (Bell et al. 1997).

Exotic removal itself can be a major disturbance to the ecosystem and may actually result in subsequent invasion by other invasive species (Westman 1990, Hobbs and Humphries 1995,

D'Antonio and Meyerson 2003). Exotic species may also contribute to species diversity and ecosystem function at some sites (Westman 1990, Myers et al. 2000). We must therefore examine both the impacts of exotics on native communities and the potential for recovery once the exotics are removed. Removal treatments must be tested before committing the resources for exotic species removal as part of a restoration plan. In Chapter 3, I investigate the potential for native forests to recover once exotic species are removed, highlighting results for an experimental removal study conducted at the North Carolina Museum of Art property in Raleigh, NC.

### **Do environmental conditions promote exotic species invasion of urban forests?**

Predicting the susceptibility of sites to future invasion is a powerful management tool for both preventing and controlling invasion (Baker 1986, Hobbs and Humphries 1995, Byers et al. 2002, D'Antonio and Meyerson 2003). A second paradigm that has emerged from exotic species research, the resource availability paradigm (Davis et al. 2000), can potentially be used to identify susceptible sites.

Because of their particular life history characteristics, invasive exotic species are able to dominate sites with available resources. These available resources can result from a decline in native vegetation, disturbance, or climate change and provide the invading species opportunity for successful establishment and a chance to outcompete the native vegetation in resource use. Identifying these life history characteristics along with environmental conditions that promote establishment and growth of exotic species should help us to understand and predict the invasibility of a particular community or ecosystem (Luken et al. 1997, Levine et al. 2003).

Much of the exotic species research effort has focused on investigating criteria that may allow certain species to become invasive outside of their natural range (Bazzaz 1986, Orians 1986, Rejmanek and Richardson 1996, Williamson and Fitter 1996) and the use of those criteria to predict which species will be invasive (Mack 1996, Reichard and Hamilton 1997, Zalba et al. 2000). Fortunately, studies that identify life history traits of exotic plant species also have value in understanding what makes communities invasible by highlighting the environmental characteristics necessary for an exotic species to colonize a site (Baker 1986). First, the environmental factors, such as climate and soil characteristics, must be amenable to successful germination, growth, and reproduction (Lonsdale 1999). Second, pollination opportunities must be available and densities of the species should be high enough to allow for successful reproduction (Crawley 1987). Third, the life forms, or structure, of the vegetation should be similar to the species' native habitat (Maron and Connors 1996). Fourth, the species must be generalists, able to survive and adapt to wide variation in environmental conditions and ecotypes (Lonsdale 1999). Fifth, resources need to be available (Davis et al. 2000). Finally, vegetative reproduction encourages the invasion of a species when native species do not reproduce frequently by seed.

Another focus of exotic species research has been on the interaction between disturbance and exotic species invasion. Some types of anthropogenic disturbance, such as fragmentation, nutrient additions, fire, flooding, and forest thinning, have resulted in significant invasion by exotic species (e.g., Pysek and Prach 1993, Milberg and Lamont 1995, Lonsdale 1999, Parendes and Jones 2000). The effects of a disturbance may be to open space, reduce competition, add nutrients, and increase light availability, thereby promoting invasion by exotics (Orians 1986, Crawley 1987, Hobbs and Huenneke 1992, Milberg et al. 1999).

To begin to understand the site factors of urban forests that promote or resist exotic species invasion, we must first detect general patterns and relationships. Studies that identify potential interactions between abiotic resources and exotic species invasion will lead to more sophisticated experimental tests to further understand these relationships. If we know which environmental conditions favor exotic species invasion, we can develop specific plans for restoring sites and preventing further invasion. In Chapter 2, I test the resource availability paradigm by evaluating the relationships between exotic species invasion and light availability and soil fertility. I then recommend restoration practices and plans to address these relationships to prevent future invasion of these urban riparian forests.

### **How do landscape-scale processes influence exotic species invasion?**

There have been very few studies of the influence of landscape structure influences invasion or invasibility of specific patches (With 2002). Current rates of habitat fragmentation, particularly in urban areas, require that we understand the spatial scale of the ecological processes affecting patterns of invasion (Higgins 1996, Zalba 2000). Because restoration sites are open systems, often within a complex landscape matrix, the influences of propagules, climate, disturbance, and other large-scale external processes on community development and maintenance need to be addressed (Parker 1997). In order to develop restoration plans for highly fragmented urban forests, we must identify the influence of these landscape-scale factors on the invasion process.

Perhaps the best example of the effects of scale on invasion is the current debate on the relationship between species diversity and invasibility. Although ecologists have accepted Elton's classic theory (Elton 1958) that less diverse systems are more heavily invaded, the

question of whether diverse communities are more resistant to invasion has been debated in the literature for many years, with evidence for both perspectives (e.g., Stohlgren et al. 1999, Levine 2000). Naeem et al. (2000) recently pointed out that observational studies of invasions often provide evidence that more diverse systems are more readily invaded but that experimental studies suggest the opposite. They suggest that greater species diversity also leads to crowding and decreased availability of resources. Thus, “covarying extrinsic” factors, such as disturbance, climate, or soil fertility, may override the effects of diversity on invasion success. This assertion was modeled by Case (1990) who showed that communities composed of strongly interacting species limit the invasion possibilities for new species.

Recently, attempts have been made to reconcile these results with experiments at several scales. For instance, Levine (2000) manipulated local diversity with a seed addition and exotic removal study in a riparian system in California. His results indicate that at smaller scales, more diverse systems are more invasion resistant while at larger scales, these diverse systems are more likely to be invaded. Therefore, studies of invasion may best be done at a variety of scales across a landscape.

The structure of the landscape, including the number, size, distribution, and connectivity of patches, influences the spread of exotic species. If we understand the relationship between landscape structure and exotic species invasion, we may better predict and control future invasions. With an experimental manipulation, Bergelson et al. (1993) demonstrated that gap size and distribution are important for predicting the rate of spread of exotics. Connectivity has also been shown to influence the spread of exotic species (Hutchinson and Vankat 1998) but studies to elucidate the mechanisms of this spread are difficult to implement (e.g., Herlin and Fry 2000).

Fragmented landscapes result in higher edge to area ratios in habitat patches, accelerating the invasion of exotic species into patches (Brothers and Spingarn 1992, Harrison 1997, Cadenasso and Pickett 2001). The landscape context of a site is also thought to influence rates of invasion (Pysek and Prach 1993, DeFerrari and Naiman 1994, Hutchinson and Vankat 1998, Gould and Gorchoy 2000). Thus, a forest patch surrounded by high-density development may be more invulnerable than a patch surrounded by low-density development. Few studies have addressed the effect of landscape context on exotic species distributions, particularly because the context is difficult to characterize and the effects of local site factors confound results (DeFerrari and Naiman 1994).

Surveys of exotic species distributions have been used to correlate invasion to specific landscape context factors. Hutchinson and Vankat (1998) conducted an aerial survey of *Lonicera maackii* populations to determine that forest patches surrounded by an agricultural matrix have lower exotic species abundance, presumably because the fields serve as a barrier to dispersal. However, a study of the exotic species distributions across forest stands in Oregon indicated that site-level factors were more important than landscape context (DeFerrari and Naiman 1994). In order to identify factors that enhance site susceptibility to invasion, we must simultaneously investigate the influence of these site factors, or environmental conditions, and landscape-scale factors.

In Chapter 4, I describe a landscape-scale study of exotic species distributions along riparian buffers in Raleigh and Cary, NC. I discuss how two measures of these buffers, their width and surrounding landscape context, can be used to predict the extent of their invasion by a suite of exotic species.

## **CONCLUSIONS**

The invasion of urban forests by exotic plant species certainly has many implications for restoration, in terms of prioritizing species removal efforts and identifying factors that promote invasion. I have taken several different approaches to addressing research questions that are currently driving the fields of exotic species ecology and restoration ecology. However, there are still many unanswered questions about the role of exotic species in these highly disturbed forest fragments and the future recovery of these communities from the onslaught of invasive exotic plant species. I hope this research encourages future work in order to restore and protect these valuable remnant urban forests.

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## CHAPTER 2

### TESTING THE PARADIGMS OF EXOTIC SPECIES INVASION IN URBAN RIPARIAN FORESTS

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Submitted to *Natural Areas Journal*

## ABSTRACT

Exotic species research has generated several paradigms about the effects of invasion on native ecosystems and site characteristics that promote invasibility. We are interested in translating these theoretical paradigms into management recommendations. Using vegetation surveys of riparian forests in central North Carolina, we tested two of these paradigms, the competition and resource availability paradigms. We assessed the impact of exotic species, prioritized removal efforts, and identified site susceptibility to future invasion. Exotic and native species richness are negatively correlated ( $r=-0.66$ ,  $p=0.0009$ ), conforming to the predictions of the competition paradigm. In particular, native woody species are negatively correlated to several exotic growth forms. Two of the most common exotic species, *Hedera helix* (English ivy) and *Microstegium vimineum* (Japanese stilt grass), do not co-occur with several native woody plants, suggesting that they may displace native woody plants from these communities. Our results have less direct implications for the resource availability paradigm. There were no correlations between light availability and exotic species. However, exotic species were generally positively correlated to soil fertility. These results suggest that the competition and resource availability paradigms are useful for understanding the dynamics of urban riparian forests that are invaded by a suite of exotic species. We recommend that removal efforts focus on two of the most common invasive plants, *H. helix* and *M. vimineum*, and that native woody plants be re-established. While soil fertility is difficult to manage at a site level, we urge managers to lobby for strict regulations on nutrient inputs from upstream and adjacent development.

## INTRODUCTION

Invasions by exotic species threaten to homogenize the world's flora and fauna (Vitousek et al. 1997, Lockwood and McKinney 2001) and pose a significant threat to biodiversity (Wilcove et al. 1998). Exotic species can displace native species (e.g., Strayer 1999), modify habitats (e.g., Schmidt and Whelan 1999), and alter ecological functions of ecosystems (e.g., Vitousek 1986). Some of these species, such as the gypsy moth (*Lymantria dispar*), zebra mussel (*Dreissena polymorpha*), and cheatgrass (*Bromus tectorum*), have attracted attention for their impacts on agriculture, forestry, fishing, and biodiversity. Their invasion dynamics are well-studied and eradication attempts well-funded (Myers et al. 2000). Yet, thousands of other invasive exotic species are threatening terrestrial and aquatic ecosystems globally, with long-term effects on native biodiversity and ecological functioning only surmised at this point.

Since Elton's seminal work, *The Ecology of Plant and Animal Invasions* (1958), ecologists have investigated the consequences of exotic species invasion on native ecosystems. Several theoretical paradigms have since been established and continue to drive current research (Levine et al. 2003). However, there has been little translation of these paradigms to management and restoration recommendations (Coblentz 1990, Byers et al. 2002, D'Antonio and Meyerson 2003).

The competition paradigm states that exotic species can outcompete native species for limited resources because of certain life history characteristics, such as prolific reproduction or rapid growth (Bazzaz 1986, Rejmanek and Richardson 1996, Williamson and Fitter 1996). Exotic species also have a competitive advantage, having escaped natural predators and other population controls (Keane and Crawley 2002, Mitchell and Power 2003). Several studies have extended this paradigm to examine the influence of spatial scale on the relationship between exotic and native species richness. At small scales, exotic species outcompete native species for

limiting resources, resulting in a negative relationship between the two (e.g., Brown and Peet 2003). At larger scales, however, exotic and native species richness have been found to be positively correlated, suggesting that both groups of species are positively responding to resource availability or increased niche opportunities (e.g., Levine 2000). These findings have caused some concern that “hotspots” of biodiversity may be threatened by exotic species invasion (Stohlgren et al. 1999, Kennedy et al. 2002, McKinney 2002).

A corollary to the competition paradigm suggests that particular exotic species may adversely affect particular native species or native growth forms. The negative effect of exotic woody shrubs on native tree seedlings and native shrubs has been investigated with observational (Woods 1993) and experimental studies both in the field (Gould and Gorchoff 2000, Gorchoff and Trisler 2003) and in the greenhouse (Luken et al. 1997). Herbaceous exotic species have also been shown to reduce native annual plants in forest understories (Meekins and McCarthy 1999) while exotic grasses suppress native shrubs in grasslands (D’Antonio et al. 1998). While the mechanisms whereby an exotic plant outcompetes the native plants may differ (Levine et al. 2003), there is some evidence that the effects of exotic species invasion extend beyond just reducing native diversity to shifting the native community composition. It is therefore important to prioritize removal efforts by identifying the exotic species or exotic growth form with the greatest effects on native communities (Byers et al. 2002, D’Antonio and Meyerson 2003).

The resource availability paradigm indicates that exotic species invasions are successful in systems with available resources (Davis et al. 2000), such as soil nutrients (Stohlgren et al. 1999, Milberg et al. 1999) and light (Brothers and Spingarn 1992, Hutchinson and Vankat 1997, Luken et al. 1997, Parendes and Jones 2000). While some systems simply may have an abundance of

resources naturally, these resources are also often freed up by disturbances. In fact, much of the observational and experimental work has shown a positive relationship between disturbance and invasion (Hobbs and Huenneke 1992). Exotic species, because of their particular life history characteristics, are better equipped to respond to increased resource availability and have been shown to dominate sites after fire (Milberg and Lamont 1995), flooding (Pysek and Prach 1993, Pyle 1995), hurricanes (White 1999), and anthropogenic disturbances (Parendes and Jones 2000, Gelbard and Belnap 2003).

Taken together, these two paradigms suggest that exotic species will outcompete native species at small scales and will be most successful at sites with available resources. We tested these paradigms in urban riparian forests, highly disturbed forest fragments that have been invaded by a suite of exotic species. Riparian forests often represent the majority of protected natural areas in urban regions and their ability to provide pollution filtration, recreational opportunities, and wildlife habitat has increased the value of their protection (Wenger and Fowler 2000). However, they are often degraded by natural and anthropogenic disturbances and invaded by a suite of exotic species. It is conceivable that the exotic species are actually increasing the species richness of these forest communities (Westman 1990, Ehrenfeld 1997, Myers et al. 2000).

The objectives of this study were to: (1) describe shifts in species composition associated with presence of a suite of invasive species of different growth forms; (2) evaluate the relationship between exotic and native species and growth form richness; (3) identify specific growth forms at risk from exotic species invasion; (4) investigate the role of environmental variables in promoting local exotic species invasion; and (5) translate our results into specific management and restoration recommendations.

## **METHODS**

### **Study Site Selection**

We focused on riparian forests in central North Carolina, a rapidly urbanizing metropolitan area. These forests are part of the Raleigh and Cary greenways systems, which protect over 65 miles of recreational trails through mostly riparian forests. As part of a larger research study, we chose 23 300m long greenway segments distributed across these two municipalities (Hull 2003). Every greenway segment has a similar management regime, including maintenance of recreational trails and sewer easements.

Mixed native hardwoods, including sycamore (*Platanus occidentalis*), red maple (*Acer rubrum*), tulip poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), and a variety of oak species (*Quercus spp.*), dominate the canopies of these riparian forests. Several exotic plants have invaded the understories of these forests, particularly in floodplain or other disturbed areas. While the soils are severely eroded in many sites, soil cations and organic matter are sufficient for plant growth (unpublished data).

### **Sampling Design**

We surveyed six 2m x 2m vegetation plots placed immediately adjacent to the top of the streambank at each site. This small plot size allowed us to detect the effects of the suite of exotic plants, which ranged from small grasses (*Microstegium vimineum*) to large shrubs (*Elaeagnus spp.*). We established the first plot at a randomly chosen distance within the first 50m of the 300m segment. Five additional plots were subsequently placed 50m apart. To ensure that we were capturing the understory vegetation, we adjusted the plots to avoid trees greater than 10cm

dbh by moving the plots slightly downstream. Each plot was marked in one corner with a steel stake for subsequent relocation.

We conducted vegetation surveys of all plots between July and September 2002, during the height of the growing season. Using the nomenclature provided by Radford et al. (1968), we identified all individuals rooted within the plot boundaries and visually estimated percentage cover by species. Because these plots included shrubs and saplings less than 10cm in diameter, many plots had multiple layers of vegetation and total percentage cover exceeded 100%.

Several environmental variables were also recorded for each plot. We used a spherical densiometer with a concave mirror to estimate the amount of canopy cover directly over the midpoint of each plot in the four cardinal directions and then averaged these measurements (Lemmon 1956). A soil sample (volume approximately 50 cm<sup>3</sup>) was collected from the top 10 cm of the soil in the center of the plot. Soil samples were analyzed by the North Carolina Soil Testing Lab (North Carolina Agronomic Division, Raleigh, NC) for percent humic matter, weight to volume ratio, cation exchange capacity, percent base saturation, exchangeable acidity, pH, and concentrations of phosphorous, potassium, calcium, magnesium, manganese, zinc, copper, and sodium.

## **Data Analysis**

Using the USDA PLANTS database (2004), we classified each species according to its status as a native or as an exotic, with the exotic species being non-indigenous to the United States. We also classified species by their growth form: fern, herb, grass, vine, or woody plant (Table 2.1). We condensed our vegetation surveys at the site level (N=23) for most analyses to avoid problems with pseudoreplication. Combining the six plots for each site, we averaged

mean percentage cover by species and totaled species richness for native species and exotic species, and for native and exotic growth forms.

#### *Patterns of species composition*

We used non-metric multidimensional scaling (NMS) in PC-ORD (McCune and Mefford 1999) to examine patterns in species composition across sites. We first ran the ordination in the autopilot mode to obtain a two-axis solution. We then reran the ordination manually with 20 runs and plotted the solution using varimax rotation. Vectors, representing native and exotic species and native and exotic growth forms, with the highest  $R^2$  values were overlaid on this ordination.

#### *Species richness relationships*

To better elucidate which groups of exotic species may be displacing native groups, we used a series of correlation analyses. We tested for significant ( $p < 0.05$ ) relationships between total exotic and native species richness, total exotic and native species cover, and exotic and native growth form richness and cover.

We used indicator species analysis (Dufrene and Legendre 1997) to detect the influence of the three most common exotic species, *Hedera helix*, *Lonicera japonica*, and *Microstegium vimineum* on native species. We conducted this analysis using percentage cover of species at the plot scale so that we could detect potential local interactions between the exotic and native species. We first divided plots into those that included a particular exotic species and those that did not include that species. Indicator species analysis was then used to identify species that either did

or did not co-occur with these exotic species. A Monte Carlo test for significance was used to identify indicator species with  $p$ -values  $< 0.05$ .

#### *Role of environmental variables*

We tested the hypothesis that increased resource availability would be positively related to exotic species invasion using a series of correlation analyses. We tested for significant ( $p < 0.05$ ) relationships between the response variables (native species richness, native species cover, exotic species richness and exotic species cover) and canopy cover and several measures of soil fertility.

## **RESULTS**

### **Patterns of species composition**

Two axes seemed appropriate for efficient representation of intersite differences in the NMS ordination (Fig. 2.1). The relationship between intersite distances and ordination distances is well-explained by this ordination ( $r^2=0.70$ ). The second axis is associated with exotic species cover and exotic species richness, with the most invaded sites clustered at the bottom of this axis. While the majority of the sites are clustered near the top half of the second axis, there is some separation of these sites based on native woody species cover along the first axis. This upper cluster of sites is also characterized by higher native species richness and cover.

## Species richness relationships

We examined species richness relationships at the site level, although we repeated all analyses at the plot level and found similar patterns. We present the site level data here to avoid the challenges of pseudoreplication.

There are negative relationships between native and exotic species cover ( $r = -0.54$ ,  $p = 0.008$ ) (Fig. 2.2a) and between native and exotic species richness ( $r = -0.66$ ,  $p = 0.0009$ ) (Fig. 2.2b). Several exotic and native growth forms are also negatively correlated (Table 2.2).

Because the native woody species seemed to be absent from the understories of the invaded plots, we examined the relationship between woody species richness and several exotic growth forms. Exotic herb richness, exotic vine richness, exotic woody richness, and total exotic species richness are all negatively correlated with native woody species richness.

The three most common exotic species, *Lonicera japonica*, *Microstegium vimineum* and *Hedera helix*, were uncorrelated to one another both in terms of percentage cover. Based on the indicator species analysis, we did find that the woody vine *Lonicera japonica* was associated with some native woody species, including *Carya* sp. and *Cornus florida*, but not associated with others (*Fagus grandifolia* and *Oxydendrum arboreum*) (Table 2.3). *L. japonica* was also associated with *Clitoria mariana* and *Toxicodendron radicans*, two common native vines in these forest communities.

*Microstegium vimineum*, a C4 grass, was associated with *Duchesnea indica*, *Liquidambar styraciflua*, and *Viola* sp but not with *Carya* spp., *Cornus florida*, *Quercus alba*, and *Viburnum acerifolium* (Table 2.3).

*Hedera helix* was associated with several other exotic species: *Bromus japonicus*, *Elaeagnus umbellata*, *Elaeagnus pungens*, *Morus alba*, *Phyllostachys* sp., *Trifolium* sp., and *Wisteria* spp. *H. helix* did

not co-occur with two woody species, *Acer rubrum* and *Viburnum rafinesquianum*, and the native vine species *Smilax* spp. and *Vitis rotundifolia* (Table 2.3).

### **Role of environmental variables**

Our tests of the resource availability paradigm yielded mixed results. There were no relationships between canopy cover and exotic species cover ( $r = 0.10$ ,  $p = 0.66$ ) or exotic species richness ( $r = 0.19$ ,  $p = 0.39$ ). However, exotic species were positively correlated to several soil variables while native species were negatively correlated to the same variables (Table 2.4).

Based on these patterns, we chose to investigate two soil variables, phosphorous and soil pH, more closely. These properties can be easily and inexpensively measured and both represent measures of soil fertility. Phosphorous is positively correlated to exotic species richness (Fig. 2.3a) and negatively correlated to native species richness (Fig. 2.3c). Exotic herb richness and exotic vine richness respond positively to phosphorous while native herb richness, native grass richness, and native woody richness respond negatively (Table 2.5).

Soil pH is also positively correlated to exotic species richness (Fig. 2.3b) and negatively correlated to native species richness (Fig. 2.3d). While exotic herb richness, exotic vine richness, and exotic woody richness are all positively correlated to soil pH, native woody richness is the only native growth form with a significant negative relationship to soil pH (Table 2.5)

## **DISCUSSION**

### **Testing the competition paradigm**

The predictions of the competition paradigm were borne out in this observational study. We chose to evaluate the relationship among exotic and native species in urban riparian forests in several ways, focusing on the cumulative impact of the suite of exotic species on both native species richness and richness of several native growth forms. Using this approach, we are able to prioritize exotic species removal efforts and to identify the native growth forms that may be most at risk for future extirpation by exotic species.

Using non-metric multidimensional scaling allowed us to further examine this shift in terms of specific species and growth forms. Sites with higher total exotic species richness and cover are separated from the rest of the sites in this ordination. In addition, this ordination allowed us to detect a shift towards native woody species when exotic species invasion was low or absent. Based on these initial findings, we identified native woody species as a potential growth form of concern in these forest communities.

As predicted by the competition paradigm, both exotic and native species richness and cover were negatively correlated. Evaluating the relationships between richness of exotic and native growth forms allowed us to identify potential interactions among groups of species. This is an important tool when particular growth forms are of concern or are at risk. We found that native woody plants are not only negatively correlated to exotic woody plants but also to several other growth forms. Because we are evaluating the composition of the forest understory, the potential suppression or extirpation of these native woody species could have long-term consequences for canopy regeneration.

Finally, we examined the effects of three common exotic species on species composition using indicator species analysis. This analysis allowed us to detect species that did not co-occur with the exotic species either because the exotic species are outcompeting native species or the environmental conditions favor the establishment of exotic species. We also can identify particular species that may be at risk of being extirpated from sites invaded by a particular species.

The presence of *Hedera helix* indicates that the site will also be invaded by several other exotic species in these urban forest communities. Because this suite of species occurs together, we offer two potential explanations. First, *H. helix* may promote the invasion of other exotic species as it quickly dominates the forest floor, outcompeting the majority of species for space and light. Only the most aggressive species can compete with *H. helix* and perhaps some of these other exotic plants have the ability to do so. The second explanation could be that *H. helix* is serving as a general indicator of disturbance and is present in only the most degraded sites where other exotics are also likely to invade. In this study, this species only occurred in the narrowest greenways, with highly incised stream channels and intense recreational pressure.

While we cannot determine if species interactions or site conditions are responsible for the co-occurrence of these exotic species, we do know that *H. helix* is negatively correlated with native species richness ( $r=-0.42$ ,  $p<0.0001$ ). We therefore recommend that restoration projects target removal of this species.

*Microstegium vimineum* was a much more common exotic plant in this study, occurring at all 23 sites. *M. vimineum* does not co-occur with several native woody plants, including the overstory *Carya sp.* and *Quercus alba*. Again, this lack of co-occurrence could be due either to competition or environmental conditions. During the summer of 2002, when this sampling took

place, the region was suffering from a drought which may have limited the spread of *M. vimineum* in many areas (personal communication, Carrie Judge, Department of Horticulture, North Carolina State University). Because this species is known to cover floodplains and other habitats in thick mats (Barden 1987, Miller 2003), future studies should address the potential for *M. vimineum* to suppress germination and survival of these woody plants as well as other native growth forms.

*Lonicera japonica*, another common exotic plant of these communities, co-occurs with a more diverse set of native woody plants than either *H. belix* or *M. vimineum*. In this study, we found no relationship between *L. japonica* and native species richness, suggesting that other species should have higher priority for removal.

These correlation analyses allowed us to identify species and growth forms most at risk from exotic species invasion in this forest community. We are also able to prioritize removal efforts by evaluating the specific correlations between common exotic species and native species. While these correlation analyses do not provide evidence of competition, they do suggest that there are negative interactions among native and exotic species which could be investigated experimentally.

### **Testing the resource availability paradigm**

The resource availability paradigm suggests that ecosystems with higher resource availability are more susceptible to invasion. Increased light, due to disturbance-facilitated canopy gaps or at forest edges, has been linked to exotic species invasion (Brothers and Spingarn 1992, Hutchinson and Vankat 1997). In this study, we found no relationship between canopy cover (a surrogate for light availability) and percentage cover or richness of exotic species.

However, we did not sample any sites with less than 40% canopy cover and suggest that further sampling of sites with low canopy cover is necessary to further test this hypothesis.

Soil fertility has also been linked to invasion in systems that have been recently disturbed (e.g., Milberg and Lamont 1995). The exotic growth forms responded positively to several measures of soil fertility. In contrast, native growth forms responded negatively to the same set of soil fertility variables. While the correlations are moderate, these opposing relationships suggest that more fertile sites are favored by exotic species.

### **MANAGEMENT IMPLICATIONS**

Urban riparian forests are highly disturbed environments but they can serve as refuges of biodiversity in otherwise degraded landscapes (Miller and Hobbs 2002). Future management of these forests will inevitably require removal or control of several invasive exotic plants. This study represents one approach for translating theoretical paradigms into management recommendations and could be reasonably replicated in other areas with the collection of observational data.

Because these forests are invaded by a suite of exotic species, we need more detailed information about the negative relationship between exotic and native species richness. The analyses presented here suggest that exotic species invasion is not only associated with lower native species richness but also is reflected in fundamental changes in species composition. The negative correlations between exotic and native growth forms suggest that exotic species are differentially affecting native species in this community. Native woody plants appear to be the most sensitive growth form, negatively correlated to most of the exotic growth forms. As invasion of these forests continues, it is these native woody plants that are potentially at the

highest risk for extirpation. The lack of germination and survival of these plants has potentially long-term consequences for canopy regeneration of these forests.

While *Lonicera japonica* is a very common plant in these ecosystems, it is actually positively correlated to native species richness. The co-occurrence of this vine with the hardwood trees suggests that, at this time, we do not need to concentrate on its removal. *Hedera helix*, on the other hand, does not co-occur with many native growth forms as well as to total native species richness and should be a target for control and removal. *Microstegium vimineum*, perhaps the fastest spreading invader in this system, is negatively correlated with total native species cover and richness but tends to co-occur with species with similar environmental and life history characteristics.

We urge managers to target *H. helix* and *M. vimineum* in future removal and control efforts. Because the suite of exotic species tends to be positively correlated with soil fertility measures, we also recommend that managers take this into account. While it may not be feasible to actively reduce soil fertility, efforts to control nutrient inputs from upstream and adjacent development should be pursued.

Exotic species research is generating paradigms that have yet to be translated into restoration recommendations in many local areas. A simple inventory approach, like the one used here, is useful for identifying native species at risk, prioritizing removal efforts, and assessing the role of environmental conditions on future invasion probability. Because this study focuses primarily on detecting patterns in disturbed forest communities, we encourage further research to examine the specific impacts that these invaders have in this system through more sophisticated experimental manipulations.

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**Table 2.1.** Exotic species encountered in vegetation surveys of 23 urban riparian forests.

<b>SPECIES</b>	<b>LIFE FORM</b>	<b>FREQUENCY (by site)</b>
<i>Bromus japonicus</i>	Grass	1
<i>Microstegium vimineum</i>	Grass	23
<i>Ophiopogon</i> sp.	Grass	2
<i>Phyllostachys</i> sp.	Grass	4
<i>Commelina communis</i>	Herb	1
<i>Duchesnea indica</i>	Herb	5
<i>Glechoma hederacea</i>	Herb	5
<i>Hemerocallis</i> sp.	Herb	1
<i>Lespedeza cuneata</i>	Herb	8
<i>Trifolium</i> sp.	Herb	3
<i>Celastrus orbiculata</i>	Vine	1
<i>Hedera helix</i>	Vine	4
<i>Humulus japonicus</i>	Vine	1
<i>Lonicera japonica</i>	Vine	22
<i>Vinca</i> spp.	Vine	1
<i>Wisteria</i> spp.	Vine	2
<i>Albizia julibrissin</i>	Woody	2
<i>Elaeagnus pungens</i>	Woody	4
<i>Elaeagnus umbellata</i>	Woody	6
<i>Ligustrum japonicum</i>	Woody	1
<i>Ligustrum sinense</i>	Woody	8
<i>Mahonia bealii</i>	Woody	1
<i>Morus alba</i>	Woody	3
<i>Nandina domestica</i>	Woody	1
<i>Pyrus communis</i>	Woody	1
<i>Rosa multiflora</i>	Woody	4

**Table 2.2.** Correlations between richness of exotic species and native species by growth form. Significant relationships ( $p < 0.05$ ) shown here.

	<b>Exotic Grasses</b>	<b>Exotic Herbs</b>	<b>Exotic Vines</b>	<b>Exotic Woody</b>	<b>Total Exotic</b>
<b>Native Ferns</b>					
<b>Native Grasses</b>			$r = -0.55, p = 0.006$		$r = -0.46, p = 0.03$
<b>Native Herbs</b>		$r = -0.46, p = 0.03$	$r = -0.66, p = 0.0006$		$r = -0.48, p = 0.02$
<b>Native Vines</b>	$r = -0.42, p = 0.05$				
<b>Native Woody</b>		$r = -0.57, p = 0.004$	$r = -0.46, p = 0.03$	$r = -0.46, p = 0.03$	$r = -0.58, p = 0.004$
<b>Total Exotic</b>		$r = -0.61, p = 0.002$	$r = -0.60, p = 0.002$	$r = -0.46, p = 0.03$	$r = -0.65, p = 0.0009$

**Table 2.3.** Indicator species of plots with and without common exotic species (Monte Carlo test,  $p < 0.05$ ). \* indicates exotic species, # indicates native woody species.

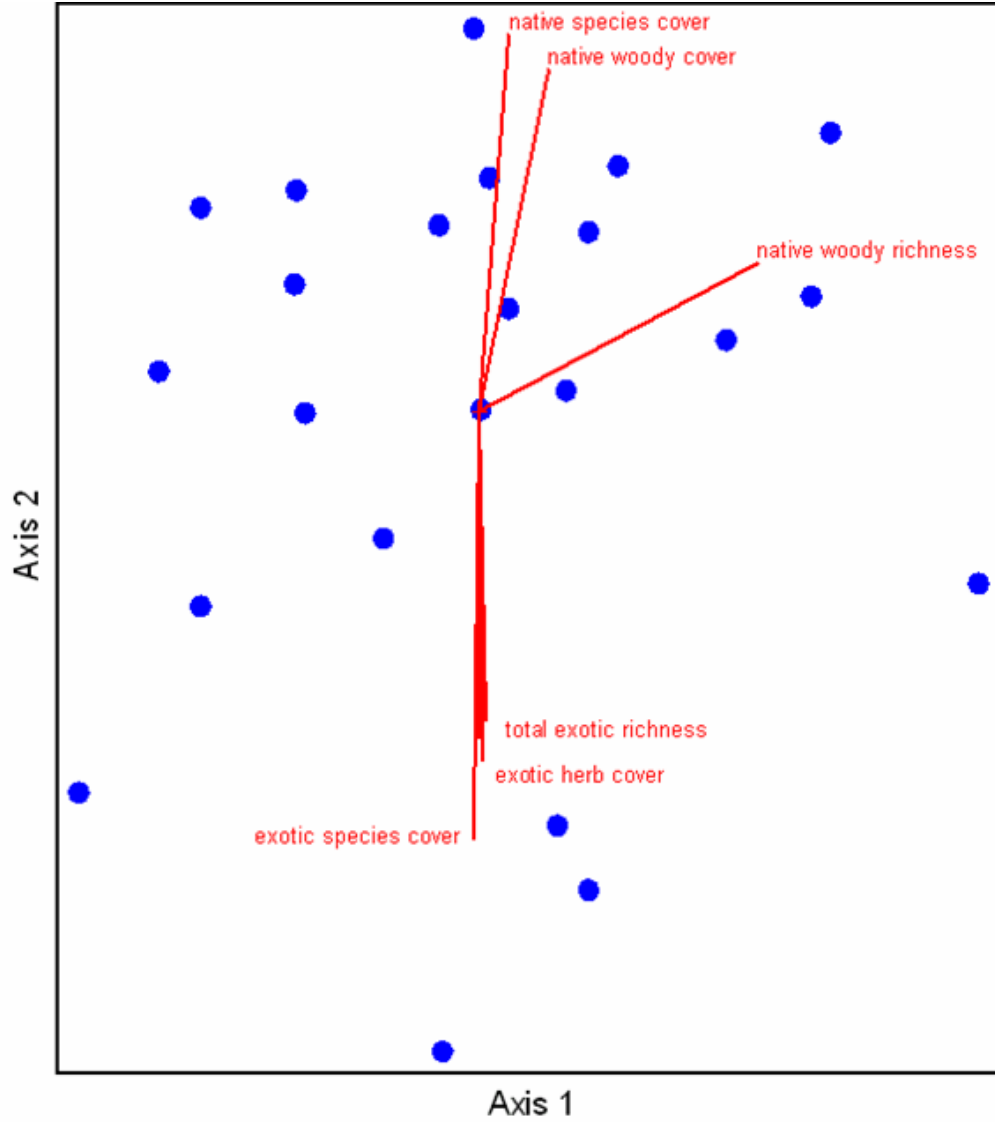
<i>Hedera helix</i>		<i>Lonicera japonica</i>		<i>Microstegium vimineum</i>	
Presence	Absence	Presence	Absence	Presence	Absence
<i>Bromus japonica</i> *	<i>Acer rubrum</i> #	<i>Carya</i> spp.#	<i>Fagus grandifolia</i> #	<i>Liquidambar styraciflua</i> #	<i>Cornus florida</i> #
<i>Duchesnea indica</i> *	<i>Lonicera japonica</i> *	<i>Clitoria mariana</i>	<i>Hedera helix</i> *	<i>Duchesnea indica</i> *	<i>Carya</i> sp.#
<i>Elaeagnus pungens</i> *	<i>Smilax</i> spp.	<i>Cornus florida</i> #	<i>Microstegium vimineum</i> *	<i>Viola</i> sp.	<i>Quercus alba</i> #
<i>Elaeagnus umbellata</i> *	<i>Viburnum rafinesquianum</i> #	<i>Toxicodendron radicans</i>	<i>Morus alba</i> *		<i>Viburnum acerifolium</i> #
<i>Glechoma hederacea</i> *	<i>Vitis rotundifolia</i>		<i>Oxydendrum arboreum</i> #		
<i>Ligustrum sinense</i> *			<i>Viola</i> sp.		
<i>Morus alba</i> *					
<i>Phyllostachys</i> sp. *					
<i>Trifolium</i> sp.*					
<i>Wisteria</i> spp.*					

**Table 2.4.** Relationships among growth forms and soil variables, with positive relationships indicated with a (+) and negative relationships indicated by a (-). Significant relationships ( $p < 0.05$ ) marked with \*.

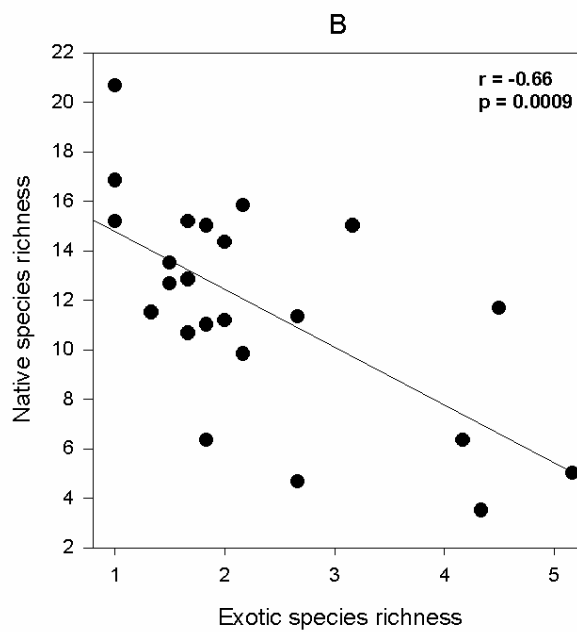
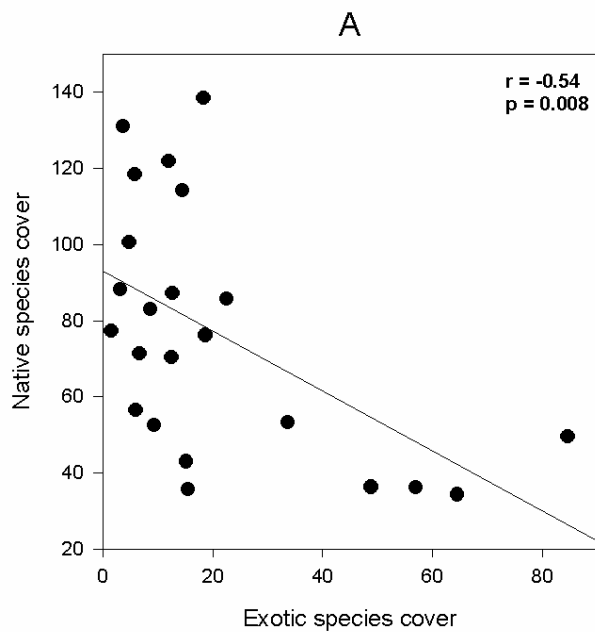
<b>Soil variable</b>	<b>Exotic cover</b>	<b>Native cover</b>	<b>Exotic richness</b>	<b>Native richness</b>
<b>P</b>	+*	-*	+*	-
<b>K</b>	+	-	+	+
<b>Ca</b>	+*	-*	+*	-
<b>Mg</b>	-	-	+	-
<b>Na</b>	+*	-	+*	-
<b>pH</b>	+*	-*	+*	-
<b>CEC</b>	+*	-*	+*	-*
<b>Base Saturation</b>	+*	-*	+*	-
<b>Mn</b>	-	+	+	+
<b>Zn</b>	+*	-*	+*	-*
<b>Cu</b>	+*	-*	+	-*
<b>Humic Matter</b>	+	-	+	-

**Table 2.5.** Correlations between phosphorous (P) and soil pH and growth forms. Significant relationships ( $p < 0.05$ ) shown here.

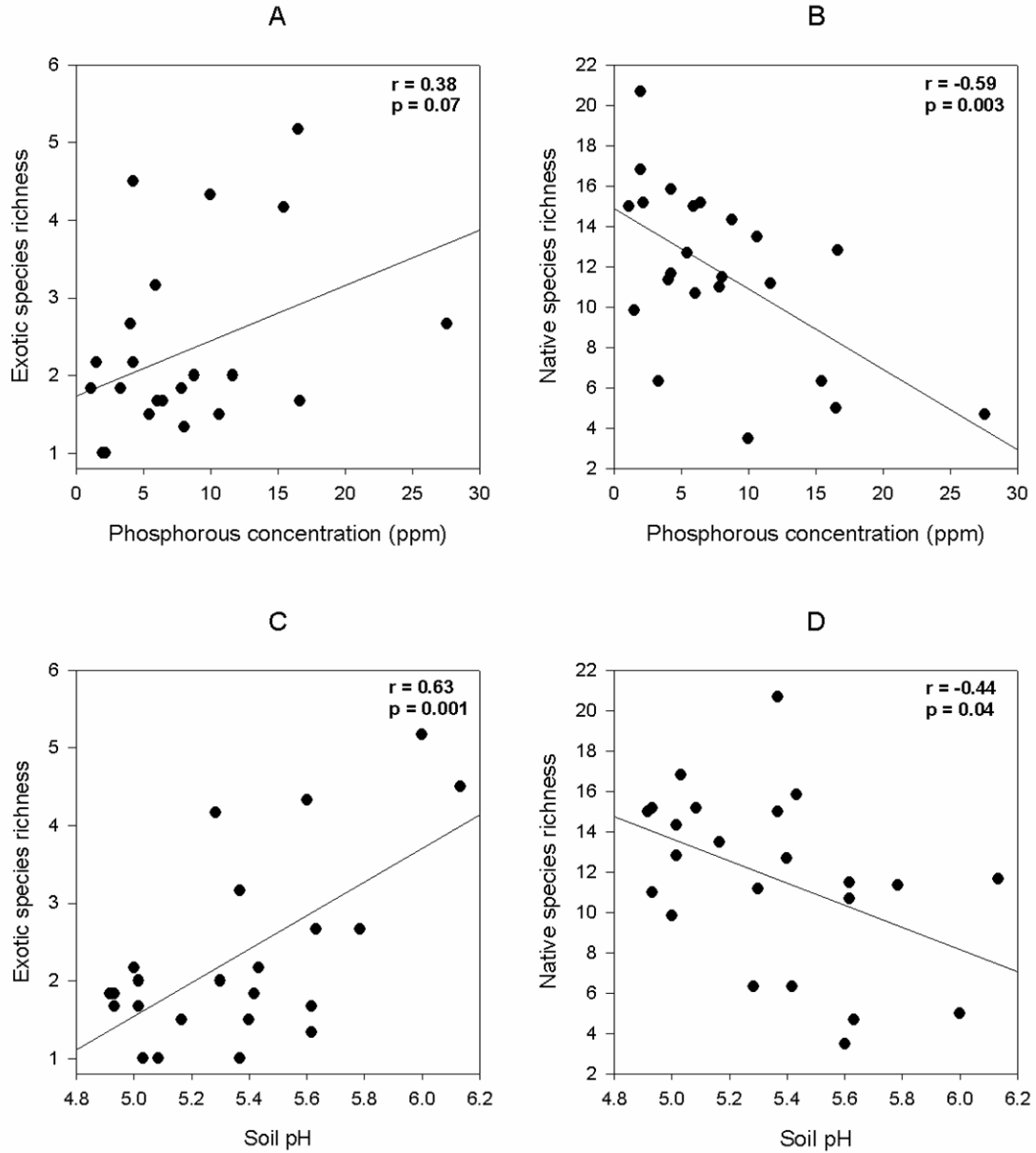
	<b>Phosphorous</b>	<b>Soil pH</b>
<b>Native Grasses</b>	$r = -0.48, p = 0.02$	
<b>Exotic Grasses</b>		
<b>Native Herbs</b>	$r = -0.49, p = 0.02$	
<b>Exotic Herbs</b>	$r = 0.61, p = 0.002$	$r = 0.61, p = 0.002$
<b>Native Vines</b>		
<b>Exotic Vines</b>	$r = 0.42, p = 0.05$	$r = -0.45, p = 0.03$
<b>Native Woody</b>	$r = -0.49, p = 0.02$	$r = -0.59, p = 0.003$
<b>Exotic Woody</b>		$r = 0.73, p < 0.0001$
<b>Total Native</b>	$r = -0.59, p = 0.003$	$r = -0.44, p = 0.04$
<b>Total Exotic</b>		$r = 0.63, p = 0.001$



**Figure 2.1.** NMS ordination depicting sites sorted along two axes ( $R^2 = 0.70$ ). Variables with  $r^2$  greater than 0.60 are overlaid on the ordination.



**Figure 2.2.** Relationships between exotic and native percentage cover (A) and between exotic and native species richness (B) at the site level.



**Figure 2.3.** Exotic species richness is positively correlated to phosphorous (A) and soil pH (B) while native species richness is negatively correlated to both phosphorous (C) and soil pH (D).

## CHAPTER 3

### EFFECTS OF EXOTIC SPECIES REMOVAL ON NATIVE SPECIES RECOVERY IN AN URBAN FOREST

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Submitted to *Restoration Ecology*

## ABSTRACT

Exotic species invasion of urban forests threatens native biodiversity of these isolated natural areas. We investigated the response of native species to removal of four common exotic species: *Elaeagnus umbellata*, *Lonicera japonica*, *Ligustrum sinense*, and *Microstegium vimineum* in a forest within Raleigh, NC, USA. In the summer of 2001, we initiated a removal experiment with three treatments. In the “repeated removal” treatment, all vegetation was removed by clipping and exotic species were removed every 2 weeks throughout the growing seasons. The “initial removal” treatment involved only removing all vegetation initially but no further weeding of new exotic recruits. Control plots had no intervention throughout the study period. We conducted vegetation surveys of the plots prior to treatment initiation and then in April and August of 2002 and 2003. With a non-metric multidimensional scaling (NMS) ordination, we were able to discern differences in species composition between the repeated removal treatment and the other two treatments. However, using repeated measures ANOVA, we found no significant differences in native species richness, cover, and abundance among treatments during most sampling periods. The shift in species composition was therefore due to removal of exotic species, not to increased native species richness. These results suggest that repeated removal of exotic species must be incorporated into restoration plans, especially if the site is in close proximity to a source of exotic species propagules. Yet, native species may not recover sufficiently without supplemental plantings. Therefore, restoration plans for urban forests should incorporate both long-term monitoring and native plant re-introduction to achieve a diverse native community.

## INTRODUCTION

Urbanization is transforming landscapes worldwide. The fragmented natural areas that remain within these urban matrices may represent pockets of biodiversity (Miller and Hobbs 2002). Urban forests can provide wildlife habitat, pollution filtration, and many other ecological functions as well as recreational opportunities. Yet, the species compositions of these forests are often significantly altered by exotic plant invasions (e.g., Pyle 1995, Ehrenfeld 1997, Hutchinson and Vankat 1997, Kloor 1999).

Few studies have addressed the effects of exotic species on urban ecosystems (Byers et al. 2002, Miller and Hobbs 2002). Urban forests are often invaded by a suite of plant species, all of which may influence species interactions and environmental conditions. Exotic plants could displace native species and alter long-term successional trajectories of these urban forests (Fikes and Niering 1999, Collier et al. 2002, Gorchov and Trisel 2003), although they could also enhance the biodiversity of these forests (Kloor 1999). Exotics could also alter forest structure by providing more complex, continuous understory and shrub layers, affecting wildlife species both positively and negatively (e.g., Schmidt and Whelan 1999). Because the effects of most invasive exotic species on species diversity and ecosystem function remain unknown, most management decisions to control exotics are supported only by anecdotal evidence (Byers et al. 2002). Given the tremendous cost of removal and control efforts, the potential benefits of removing exotic species should be evaluated to better inform management and restoration decisions (Westman 1990, Myers et al. 2000).

The impacts of exotic species invasion on native species diversity have primarily been studied by comparing invaded to uninvaded areas (see Levine et al. 2003 for review). Yet, to determine if the exotics are actually suppressing native species instead of occupying space where

no natives previously existed, removal studies are necessary (Meekins and McCarthy 1999, Gould and Gorchov 2000). Removing exotic species allows us to determine whether the natives were indeed displaced by the exotics and to evaluate the potential for native species richness or abundance to recover.

We are interested in the response of these urban forest communities to removal of exotic species, particularly in terms of native species recovery. We asked the following questions: (1) Does removal of exotic species significantly increase native species richness, cover, and abundance in urban forests? and (2) What type of restoration is necessary to remove exotic species?

## **METHODS**

### **Study Site**

This study was conducted on the grounds of the North Carolina Museum of Art in Raleigh, NC. Within a heavily urbanized area, the 164 acre museum property includes facilities, forest, and pasture (Fig. 3.1). We are currently assisting museum staff in developing restoration plans for an Art and Environment Park. One element of the restoration effort could involve removal of an extensive exotic plant understory, dominated by *Ligustrum sinense* (Chinese privet) and *Elaeagnus umbellata* (silverberry), throughout approximately 45 acres of forested land.

This mixed-hardwood forest is representative of the remaining forest fragments in the Raleigh metropolitan area. Urbanization has transformed the landscape from mixed hardwood-pine forests to a matrix of development surrounding isolated forest patches. The remaining secondary successional forest patches suffer from recent anthropogenic disturbances, including heavy recreational use, runoff from surrounding developments, and invasion by a suite of exotic

plant species. Several exotic shrubs, such as *Ligustrum sinense* (Chinese privet), *Elaeagnus umbellata* (silverberry), and *Rosa multiflora* (multiflora rose) often dominate the understory of these forests. In addition, *Microstegium vimineum* (stilt grass) is quickly spreading throughout floodplains and uplands of this region (Miller 2003).

The Art Museum forest is a typical urban forest patch, surrounded by major roads and industrial and residential development. The canopy is composed of native mixed hardwoods, including *Liriodendron tulipifera* (tulip poplar), *Diospyros virginiana* (persimmon), *Liquidambar styraciflua* (sweetgum), and several *Quercus* (oak) species. The understory is dominated by exotic shrubs, which create an almost continuous thicket throughout large parts of this forest. This forest also bears some evidence of recent human settlement, including established ornamental plantings and small waste dumps.

## **Study Design**

Within this forested area, we delineated four sites of approximately 100 square meters. Each site featured a closed forest canopy, with no discernible canopy gaps, and was located at least 100m from the forest edge to minimize edge effects. Exotic understory density was visually consistent throughout these sites.

We established a replicated complete block experiment with three treatments: repeated removal, initial removal, and control. These treatments were applied to 2m x 2m plots in 3 blocks per site, for a total of 36 plots. Plot corners were marked with PVC pipes and flagging tape for relocation. Individual plots were separated by at least 1m and the sites were marked off with caution tape to reduce public interference with the experiment.

In July 2001, prior to experimental manipulations, we conducted vegetation surveys of each plot. We first divided each plot into 4 1m x 1m sections and then recorded the species, number of stems per species, and percentage cover of each species in the four sections. Nomenclature follows Radford et al. (1968). We then summed the number of stems by species and averaged the percentage cover of species for each 2m x 2m plot.

We initiated the three treatments on July 9, 2001. Control plots were left unaltered. The other plots were cleared of all vegetation, except native trees greater than 5 cm dbh, by clipping stems to ground level and removing this vegetation from the site. We chose to remove both native and exotic plants to avoid initial among-plot differences in above- and below-ground competition. However, the majority of biomass removed from the plots was of the two common exotic shrubs, *Ligustrum sinense* and *Elaeagnus umbellata*.

The “initial removal” treatment allowed for regeneration of all vegetation, except resprouting exotic plants. Where we could discern a previously cut stump, we continued to clip resprouts during the first growing season. Previous studies indicated that clipping of the exotic shrub, *Lonicera maackii*, resulted in mortality of the shrubs after 2 years and complete mortality of new stems on the shrub (Luken 1988, Luken and Mattimiro 1991). In our study, resprouting of clipped shrubs was minimal by the second growing season.

The “repeated removal” treatment required the clipping of any exotic plant seedling (*E. umbellata*, *L. sinense*, *R. multiflora*, *L. japonica*, and *M. vimineum*) that sprouted in the plots. These exotics were also removed by clipping back to the ground. Plots were visited every 2 weeks during the growing seasons to clip new exotic seedlings. We repeated the vegetation surveys of the plots in April and August of 2002 and 2003.

We minimized soil disturbance by avoiding trampling of the plots and by clipping vegetation (McClellan et al. 1995). We also cleared a 1m border around each plot, including controls, to reduce above- and below-ground competition from neighboring plants.

### **Seedbank Study**

To determine if native plant germination is suppressed by either the existing vegetation or by environmental conditions, we conducted a seedbank study. In February 2003, we collected soil samples from each of the four sites. Since we intend to continue monitoring recovery, we collected our soil samples from the area immediately adjacent to each plot instead of directly from the plot. We collected 4 soil cores, each 2.5 cm in diameter and 5 cm deep, from each side of every plot, for a total of 16 cores per plot and 144 cores per site. The soil substrate is quite rocky, which prohibited deeper soil samples.

We pooled the soil samples for each site and placed them in a freezer to undergo cold stratification for 8 weeks. In April 2003, we transferred the soil samples to an unheated greenhouse. One inch of soil was spread over a 2 inch substrate of Metromix, a commercially available potting soil. Six flats were used per site and an additional 4 flats of Metromix were interspersed between the study flats to measure any germination due to greenhouse contamination. An automatic watering system was used to ensure moist soil conditions. From April 2003 through January 2004, emerging seedlings were identified, counted, and removed from the flats.

## **Data Analysis**

We first used non-metric multidimensional scaling (NMS) with the Sorensen distance measure in PC-ORD (McCune and Mefford 1999) to detect differences in species composition among treatments and among time periods. NMS analysis was first run using the “autopilot” option to determine the best number of axes and then manually re-run 20 times. The scores were plotted with varimax rotation. For this analysis, we used categorical cover classes (Peet et al. 1998) instead of raw percentage cover to minimize variability in visual percent cover estimates made over two years.

Next, we used mixed model repeated measures analysis of variance (ANOVA) (SAS/STAT software, Version 8.1 of the SAS System for Windows, copyright 2001. SAS Institute, Inc.) to test for the effect of treatment and time on the following response variables: total species richness, exotic species richness, native species richness, native species cover, native species abundance, native tree seedling richness, and native tree seedling abundance. The mixed model allowed site and block to be treated as random effects and included an interaction between treatment and time. Differences of least square means, significant at the  $p < 0.05$  level, were compared to detect treatment and time effects.

## **RESULTS**

### **Treatment effects**

We selected a two-axis solution for ordinating species composition of plots (Fig. 3.2a), with the distance between plots representing the difference in species composition. The cluster of repeated removal plots is clearly separate from plots of the initial removal and control treatments, suggesting differences in species composition.

The repeated measures ANOVA results confirmed differences among treatments and among time periods for most response variables (Table 3.1). Prior to initiation of the treatments, the plots were not significantly different, in either exotic species richness or native species richness. Throughout time, the control and initial removal treatments had similar exotic species richness, while the repeated removal treatment had no exotic species (Fig. 3.3a). There were no significant differences in the number of native species (Fig. 3.3a) or native species cover (Fig. 3.3b) among treatments throughout this experiment. Native tree seedling richness was significantly higher in the control treatment until the August 2003 survey, when there were no significant differences detected among treatments (Fig. 3.3d). One tree species in particular, *Liriodendron tulipifera*, achieved higher percentage cover in the repeated removal treatment (Fig. 3.2b).

In August 2003, native species abundance, or the number of native stems, was significantly higher (mean = 75.9, standard deviation = 26.5) in the repeated removal treatment than either the initial removal (mean = 44.9, standard deviation = 31.3) or control (mean = 35.2, standard deviation = 27.7) treatments (Fig. 3.3c). Tree seedling abundance peaked in the repeated removal treatment in April 2003 but was not significantly different from the other treatments in August 2003 (Fig. 3.3e).

### **Interannual and seasonal variability**

The effect of time was significant in all repeated measures ANOVA models (Table 3.1). Mean native species richness, native species cover, and native species abundance all declined during the August 2002 period (Fig. 3.3 a-c). Most response variables recovered or increased during the April and August 2003 surveys (Fig. 3.3 a-e). Seasonal variability is confirmed by a

two-axis NMS solution for species composition by season indicates that spring surveys are more similar to each other than summer surveys (Fig. 3.4)

### **Seedbank results**

Several early successional species sprouted in the greenhouse flats from April to December 2003 (Table 3.2). In addition, several tree seedlings sprouted, primarily shade intolerant species such as *Liriodendron tulipifera* and *Liquidambar styraciflua*. One native shrub seedling, *Myrica cerifera*, sprouted while several exotic plant seedlings (*L. sinense*, *L. japonica*, and *M. vimineum*) were found in the majority of flats.

## **DISCUSSION**

### **Effect of exotic species removal on native species recovery**

The species composition of the repeated removal treatment is quite different from the other two treatments. This is attributed to an absence of exotic species, not to an increase in native species. The control plots retained the highest native species richness of all treatments throughout the study time, suggesting that the exotics are not suppressing the native flora. In August 2003, however, the number of native stems per plot in the repeated removal treatment was significantly higher than those in either of the other two treatments, indicating potential for long-term increases in native plants once exotics are removed.

There are several potential reasons for the lack of native species recovery in the repeated removal treatment plots. First, environmental conditions may not be amenable to native germination. The seedbank appears to have sufficient seed of shade intolerant species yet new plants may not have become established because of insufficient light availability or because of

belowground competition, despite maintenance of plot borders free of vegetation (Luken et al. 1997).

Gorchov and Trisel (2003) found that aboveground competition more significantly limits tree seedling establishment near the exotic shrub, *Lonicera maackii*, than belowground competition. We expected, therefore, that native species in general, and tree seedlings specifically, would respond more positively to the removal treatments. The increase of *Liriodendron tulipifera* cover in the August 2003 does represent some potential for regeneration of tree seedlings but the lack of other overstory species regeneration should be explored further.

The second explanation for the lack of native species recovery is that exotic species continue to outcompete native species. Large numbers of *Elaeagnus umbellata* and *Ligustrum sinense* seedlings continually resprouted throughout the growing seasons in all plots that initially contained smaller exotic shrubs, under 1 m tall. This phenomenon is consistent with Webb et al.'s (2001) finding of large increases of seedlings of the exotic *Acer platanoides* when relatively small seedlings of this species were removed. The new plants could be from mass of germinating seeds, provided by the abundant exotic shrubs surrounding these experimental plots, or from re-sprouting from existing roots. These abundant exotic seedlings may be competitively excluding the native seedlings from the plots.

Third, recruitment of native species may be limited. An informal survey of the site found no native shrubs over 1m tall (personal observation). Recruitment of native propagules from outside the site is also limited by the isolation of the NC Art Museum forest within a heavily urbanized matrix. A concurrent study suggests that landscape context in urban areas can significantly influence the extent of exotic species invasion, as adjacent neighborhoods provide a source of exotic propagules (Vidra et al., in review). The potential for native propagules to

disperse to this site, via animals, wind, or water, should be examined more closely in a future study.

Finally, the lack of native woody plants in the seedbank may be the result of “ghost of competition past.” While we have no historical records that document the time since invasion, the exotic understory may have persisted long enough to displace native species from the seedbank as well as the understory. To explore this explanation, future studies could compare the seedbank at this site with both uninvaded (or less invaded) sites and those sites that have only recently been invaded.

### **Comparison of restoration treatments**

The species composition of the initial removal plots was similar to the control plots, suggesting that this treatment is not sufficient to reduce exotic species invasion. Therefore, the initial removal effort must be followed by continued weeding of resprouting exotics and newly established seedlings. In this study, the surrounding forest retained a high density of exotic plants, supplying a large exotic propagule source throughout the study. Removing all exotic plants throughout the site may therefore reduce the reinvasion, although outside sources will likely continue to contribute new recruits (Vidra et al., in review).

### **Effect of time on native species recovery**

Interannual variability in precipitation was significant during this study. All response variables were lower in August 2002, perhaps due the severe drought that affected the southeastern region. 2003 was an abnormally wet year and native species recovery during this growing season suggests that given appropriate environmental conditions, the understory species

composition could include a richer native flora. However, the cover of *Microstegium vimineum* expanded greatly throughout floodplains and adjacent forests in the area during the 2003 growing season, potentially intensifying exotic competition (personal observation).

This study took place over two complete growing seasons. Other experiments have shown responses of annual plants to removal over the course of just one growing season (Meekins and McCarthy 1999, Gould and Gorchoy 2000). Woody plants will likely take longer to recover and may require changes in environmental conditions as well, such as an increase in light availability. While the native species richness has not significantly increased in our treatment plots, we are encouraged by the increasing number of native stems in the repeated removal plots. While the initial results of this study are presented here, we plan to continue monitoring these plots for native species recovery over a longer time frame.

### **Recommendations for restoration**

Removing exotic species from urban natural areas requires tremendous time and effort (e.g., Cramer 1993, Holloran 1996). Yet, at the 2003 Society for Ecological Restoration International meeting, no fewer than 16 presentations featured urban restoration projects that include removal of exotic species. Because these fragmented forests remain susceptible to invasion (Vidra et al., in review), long-term intervention is necessary but expensive.

Removal of exotic species is incorporated into restoration plans in order to maintain native species composition, enhance wildlife habitat, and improve the aesthetic appeal of forests. Even if exotic plants are adding to the overall species richness of a site, as seen in this study, their presence could have long-term implications for forest health (Fikes and Niering 1999, Gorchoy and Trisel 2003). Continuous exotic herbaceous layers have been shown to suppress

native species germination, thereby reducing canopy tree regeneration (e.g., Meekins and McCarthy 1999). Dense understory shrub layers could have the same effect by reducing light availability and outcompeting tree seedlings for other resources (Luken et al. 1997, Gorchov and Trisel 2003). Therefore, the current thicket of exotic shrubs in this forest may preclude future canopy regeneration.

If removal is integrated into restoration plans, supplemental plantings of desirable species may be necessary to restore a diverse forest community. These plantings may also help to resist future invasion by occupying space and resources, thereby edging out aggressive exotic species. This is particularly important in urban areas, where native species recruitment may be minimized by extreme habitat fragmentation (Miller and Hobbs 2002). The landscape context, or surrounding land use, plays an important role in invasion dynamics of forest fragments in urban areas (Vidra et al. in review). Therefore, attention must be paid to both sources of exotic seeds and native seeds. In this study, there were plenty of exotic seeds due to the abundance of exotic plants left in the surrounding forest. However, native shrubs were absent at this site and in the seedbank, suggesting that recruitment from outside the site or intentional planting of native shrubs may be necessary to create a diverse forest understory.

In this study, we clipped the plants to minimize soil disturbance. Thus, the roots were left intact initially. Other studies have shown that repeated clipping will result in death of woody plants in one growing season (Luken and Mattimiro 1991). However, it is possible that some of the exotic plants in the initial removal and control treatments were resprouts from intact roots of adjacent shrubs (Webb et al. 2001). To reduce this possibility and to minimize the continued weeding efforts, we suggest that restorationists consider the use of a target herbicide. A small pilot spraying project undertaken in another area of this forest appears to have completely killed

exotic vegetation with very little resprouting (personal observation). Any restoration treatment should incorporate continued maintenance to remove exotic seedlings.

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**Table 3.1.** Mixed model repeated measures ANOVA results for effects of treatment, time, and the treatment by time interaction on response variables (bold indicates significance at  $p < 0.05$ ).

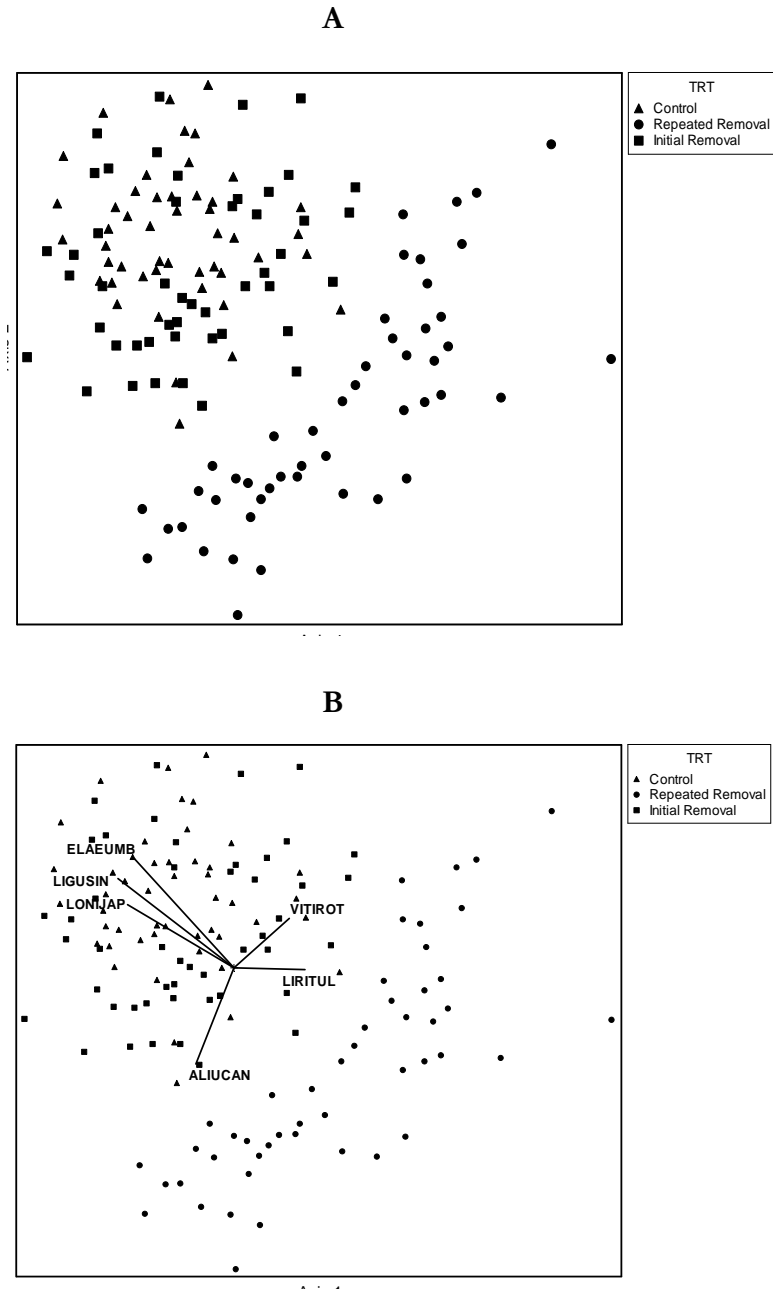
<b>Response Variable</b>	<b>Treatment</b>	<b>Time</b>	<b>Treatment*time</b>
<b>Species richness</b>	<b>F=15.23, p&lt;0.0001</b>	<b>F=34.14, p&lt;0.0001</b>	<b>F=2.74, p=0.02</b>
<b>Exotic richness</b>	<b>F=157.69, p&lt;0.0001</b>	<b>F=7.64, p=0.0001</b>	<b>F=4.41, p=0.0005</b>
<b>Native richness</b>	F=0.53, p=0.59	<b>F=27.38, p&lt;0.0001</b>	<b>F=2.87, p=0.01</b>
<b>Tree seedling richness</b>	<b>F=3.36, p=0.04</b>	<b>F=3.12, p=0.03</b>	F=1.57, p=0.16
<b>Tree seedling abundance</b>	F=2.73, p=0.07	<b>F=7.79, p=0.0001</b>	<b>F=4.89, p=0.0002</b>
<b>Native cover</b>	F=0.08, p=0.93	<b>F=7.73, p=0.0002</b>	F=1.10, p=0.37
<b>Native species abundance</b>	F=0.09, p=0.42	<b>F=14.43, p&lt;0.0001</b>	<b>F=6.66, p&lt;0.0001</b>

**Table 3.2.** Number of seedbank seedlings by species and site that emerged between April 2003 and January 2004 (\* indicate exotic species).

<b>Species</b>	<b>Site 1</b>	<b>Site 2</b>	<b>Site 3</b>	<b>Site 4</b>
<i>Duchesnea indica</i> *	8	7	5	14
<i>Rubus sp.</i>	7	13	2	6
<i>Lonicera japonica</i> *	6	3		1
<i>Liriodendron tulipifera</i>	5	5	6	6
<i>Microstegium vimineum</i> *	5	5		
<i>Liquidambar styraciflua</i>	4	6	3	1
<i>Cyperus odoratus</i>	4	12		11
<i>Eupatorium capillifolium</i>	4	1		1
<i>Viola sp.</i>	3	4	3	1
<i>Phytolacca americana</i>	3	4	1	6
<i>Vitis rotundifolia</i>	3			1
<i>Huechera sp.</i>	2	2	1	1
<i>Ipomea lacunosa</i>	1	2	1	
<i>Gnaphalium obtusifolium</i>	1	1	5	12
<i>Sida rhombifolia</i>	2	1	1	2
<i>Ligustrum sinense</i> *	1	1		
<i>Ipomea purpurea</i>	1		4	
<i>Pinus taeda</i>	1		4	3
<i>Ambrosia artmesifolia</i>	1			2
<i>Plantago sp.</i>	1			
<i>Carex crinita</i>		6	5	12
<i>Baccharis halimifolia</i>		5	3	3
<i>Scutellaria elliptica</i>		2		2
<i>Eragrostis hirsuta</i>		2		
<i>Geranium carolinianum</i>		1	1	
<i>Campsis radicans</i>		1		2
<i>Myrica cerifera</i>		1		
<i>Prunus serotina</i>		1		
<i>Lespedeza cuneata</i> *		1		
<i>Erigeron canadensis</i>			1	
<i>Dichanthelium sp.</i>				1
<i>Solanum americanum</i>				1
<b>Total species richness</b>	<b>20</b>	<b>24</b>	<b>16</b>	<b>21</b>



**Figure 3.1.** Aerial photograph (1999) showing Art Museum property outlined in black with study area noted by \*. Forest covers approximately 45 acres and is bordered by major roads, residential developments, and pasture.



**Figure 3.2.** Non-metric multidimensional scaling (NMS) two-axis solution for species composition of three experimental treatments (A) and with species vectors overlaid (B). (ELAEUMB = *Elaeagnus umbellata*, LIGUSIN = *Ligustrum sinense*, LONIJAP = *Lonicera japonica*, VITIROT = *Vitis rotundifolia*, LIRITUL = *Liriodendron tulipifera*, and ALIUCAN = *Alium canadense*).

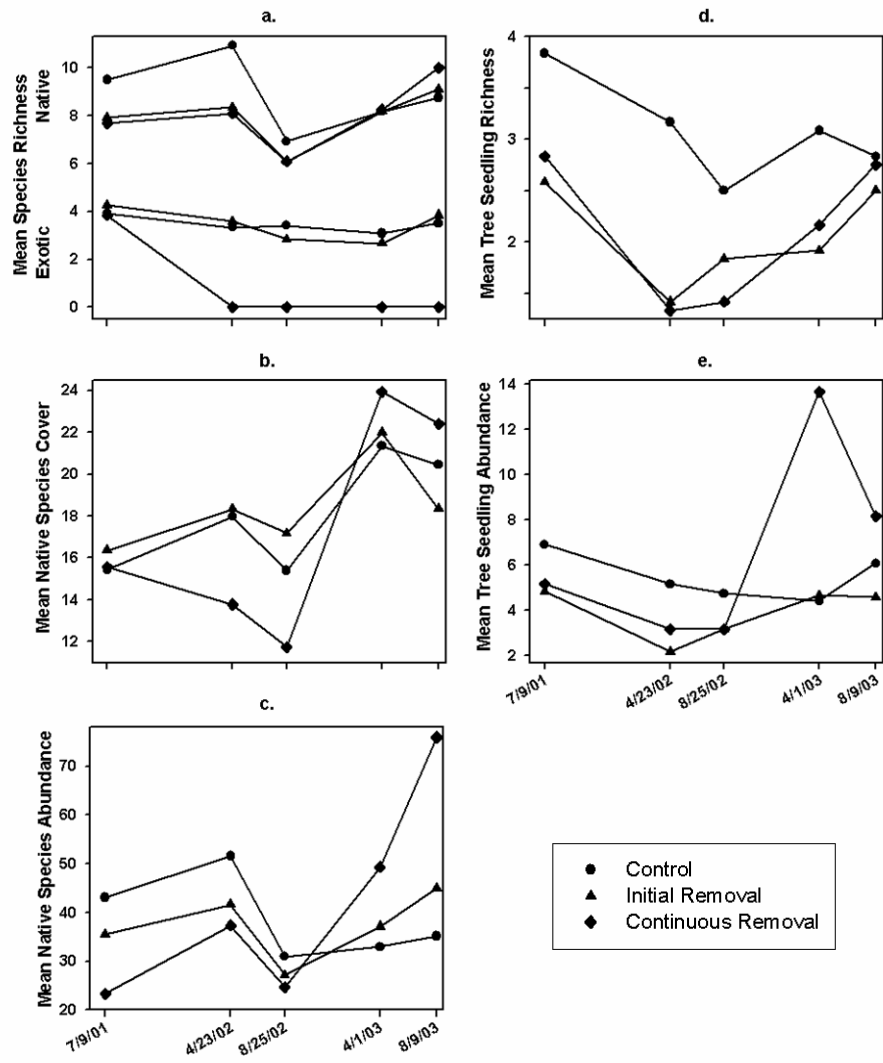
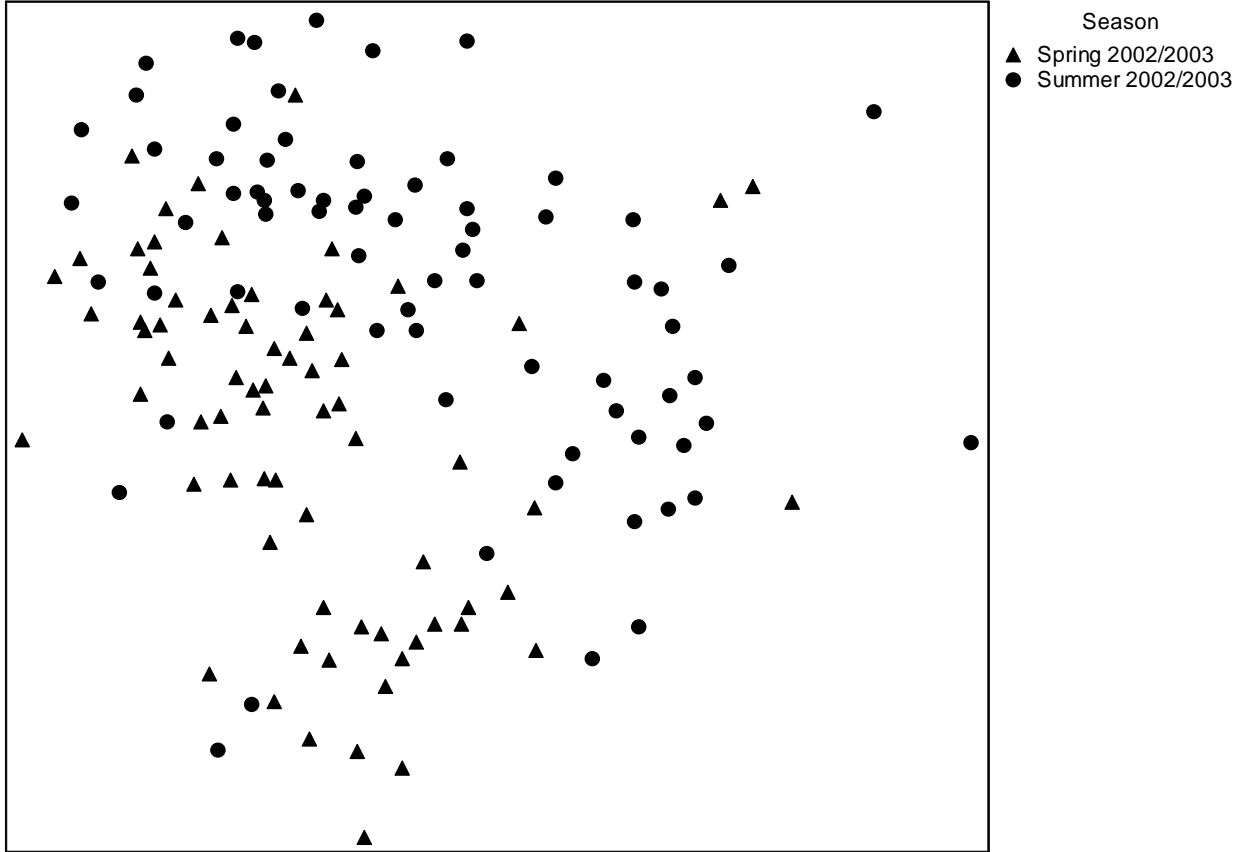


Figure 3.3. Mean values for response variables for each sampling period.



**Figure 3.4.** Non-metric multidimensional scaling (NMS) two axis solution for species composition by season.

## CHAPTER 4

### URBAN LANDSCAPE STRUCTURE INFLUENCES EXOTIC SPECIES INVASION OF RIPARIAN BUFFERS

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Submitted to *Conservation Biology*

## **ABSTRACT**

Protection of riparian buffers is becoming increasingly common in urban areas to provide flood protection, sediment filtration, wildlife habitat, and recreational opportunities. Unfortunately, these long, linear strips of forest are susceptible to invasion by exotic species, which can compromise both the native species composition and ecological functions of these buffers. We investigated the influence of landscape structure on buffer invasion in central North Carolina, USA. We assessed the effect of buffer width and landscape context on the exotic species richness, individual species cover, and total exotic species cover in 23 riparian buffers distributed across a rapidly urbanizing region. We characterized landscape context as the proportion of the adjacent landscape that fell into two land cover types: building cover and canopy cover. The most significant predictor variable of exotic species invasion is the amount of canopy cover in the adjacent landscape. Increasing canopy cover is positively correlated to both exotic cover and species richness. Building cover is positively correlated to percentage cover of ornamental exotics but not to total species richness. While the relationship between buffer width and cover and richness of exotic species is negative, this relationship is only significant for buffers narrower than 150m. These results suggest that buffer width, while important, is not the only important factor in determining susceptibility to invasion. The adjacent landscape context significantly influences the extent of invasion, likely by serving as both a source of exotic propagules and by providing external disturbances. We suggest that managers and others interested in conserving native riparian buffer communities focus on the landscape structure of the site to prevent future invasion by exotic plant species.

## RODUCTION

Riparian buffers represent a significant proportion of protected lands in urban areas and can serve as refuges of biodiversity in these otherwise degraded landscapes (Miller and Hobbs 2002). Ordinances to protect the ecological functions of forested riparian buffers, including sediment and pollution filtration (Welsch 1991), erosion control (Beeson and Doyle 1995), and wildlife habitat (Keller et al. 1993, Kilgo et al. 1998), are becoming increasingly common in urban areas. These buffers can also provide recreational opportunities as greenways and benefit adjacent neighborhoods with enhanced property values and flood protection.

In addition to their many benefits, riparian buffers are susceptible to invasion by exotic plant species because of the large proportion of edge, frequent disturbance by flooding, and accessibility to water-dispersed seeds (Pysek and Prach 1993, DeFerrari and Naiman 1994, Hood and Naiman 2000, Stohlgren et al. 2000). In urban areas, these buffers are further disturbed by the presence of sewer lines and recreational trails. Exotic species invasion of these buffers may negatively impact native species communities (e.g., Pysek and Prach 1993, Brown and Peet 2003) and compromise ecological functions (Vitousek 1986, Schmidt and Whelan 1999). Therefore, recommendations are urgently needed to predict and prevent future invasions of these communities.

Few studies have examined the effects of urban development on the establishment and spread of exotic species (Miller and Hobbs 2002). In general, an exotic species is expected to colonize a site if there is suitable habitat, dispersal opportunity, source populations of propagules, and a disturbance that opens up space or other resources (Baker 1986, Crawley 1987, Hobbs and Humphries 1995, Davis et al. 2000). Riparian buffers provide suitable habitat for many exotic plant species and are frequently disturbed by flooding (Hood and Naiman 2000,

Brown and Peet 2003). However, propagule pressure and dispersal opportunities are influenced by the spatial position of the site within a larger landscape (With 2002). Thus, landscape structure likely influences patterns of invasion across riparian buffers within urban areas.

Landscape structure has been linked to the spread of exotic species in both experimental (Bergelson et al. 1993) and observational studies (Hutchinson and Vankat 1998, McKinney 2002). Larger, isolated patches tend to be less invaded than smaller, connected patches, presumably because of limited dispersal of exotic propagules to the center of larger patches (Bergelson et al. 1993, Cadenasso and Pickett 2001) and their lower edge to area ratios (e.g., Brothers and Spingarn 1992). The landscape type surrounding a patch may also influence the spread of exotics. Invasion may be limited if the patch is surrounded by unsuitable habitat (Hutchinson and Vankat 1998, Cadenasso and Pickett 2001). Yet, nearby disturbance, such as human settlement, may enhance invasion by providing a source of exotic propagules or degrading patch habitat (e.g., McKinney 2002). To understand the influence of landscape structure on patterns of invasion, we must therefore examine both the patch size and the surrounding landscape context.

Significant debate surrounds the appropriate buffer width necessary to protect valuable ecological functions (Wenger and Fowler 2000). For example, to enhance their sediment trapping abilities, recommendations from 5 to 80 meters of undisturbed buffer on each side of a stream have been made (Cooper et al. 1987, Desbonnet et al. 1994). A 100m wide buffer is commonly recommended to provide sufficient wildlife habitat (Keller et al. 1993) although some urge protecting at least 500m to ensure protection of diverse wildlife communities within riparian forests (Noss 1993, Kilgo et al. 1998). Therefore, the ecological functions that drive the riparian buffer protection will also influence the appropriate width of that protection. In terms

of exotic species invasion, we are interested in identifying the influence that width has on resistance to invasion.

Because recruitment of plant propagules from outside the buffer will ultimately play a role in the establishment of exotic species, the character of the surrounding landscape matrix must be taken into account. Urban landscapes are heterogeneous mixtures of land use types (Luck and Wu 2002) and forested riparian buffers are not often surrounded solely by one type of dense development. In fact, small remnant forest patches can be found throughout these landscapes, especially in residential developments, and may increase connectivity between larger forest patches. This connectivity may facilitate dispersal of exotic propagules by animals into riparian buffers.

Propagules of some invasive exotic species are widespread in urban landscapes. In North Carolina, for example, the water- and wind-dispersed *Microstegium vimineum* is found in almost all riparian zones and is quickly becoming a noxious weed of playing fields, lawns, and adjacent forests (Barden 1987, Miller 2003). However, there are many popular exotic ornamental plants, such as *Vinca* spp., *Wisteria* spp., and *Nandina domestica* that have recently become invasive and are beginning to spread outside of their initial planting area. Thus, the vegetative spreading and animal dispersal of ornamental exotic propagules from the adjacent landscape into nearby riparian buffers could be substantial.

All riparian buffers within urban areas may not be equally susceptible to invasion. If we understand the role that landscape structure plays in the distribution of exotic species across riparian buffers (With 2002), we can enhance site selection and management efforts and reduce the probability of future invasion (Hobbs and Humphries 1995). In this study, we address the influence of landscape structure, measured as buffer width and adjacent landscape context, on

the distribution of exotic species in riparian buffers within a rapidly urbanizing portion of central North Carolina. We assessed this distribution using exotic species richness and percentage cover, which are both commonly used to describe the invasion status of habitat (e.g., DeFerrari and Naiman 1994, Parendes and Jones 2000, Gelbard and Belnap 2003). We asked the following questions: (1) Is there a relationship between width of riparian buffers and invasion? (2) Does the landscape context of the riparian buffer influence the extent of invasion? Our goal is to provide management recommendations for both appropriate buffer widths and adjacent land use practices that will reduce future invasion by exotic plant species.

## **METHODS**

### **Study Sites**

We focused on forested riparian buffers in Raleigh and Cary, North Carolina. Driven by tremendous population growth and urban sprawl, forest cover has been dramatically reduced over the past 30 years and currently occupies 45% of the region (Costa and Peterson 2002). The majority of remaining secondary successional forest is in various stages of recovering from the widespread agricultural abandonment of the late 19<sup>th</sup> and early 20<sup>th</sup> centuries (Christensen 1977).

Intact riparian forests are the most diverse plant community of the Piedmont of North Carolina (Peet and Christensen 1980). These mixed hardwood forests often have diverse shrub and vine subcanopies with rich herbaceous layers. Yet, in urban areas, these forests are heavily degraded by adjacent and upstream development, stream channelization, recreational trails, and sewer system installation and maintenance. These forests are also commonly invaded by a suite of exotic plant species (Peet and Christensen 1980, Barden 1987, Miller 2003).

Riparian buffer protection in the study area was recently enhanced by a new ordinance that protects a 36m buffer of any intermittent or perennial tributary of the major watershed, the Neuse River, from development. Many existing riparian buffers have been integrated into the Raleigh and Cary greenways systems, which include over 65 miles of recreational trails. Greenway managers are interested in managing these buffers for wildlife habitat, as well as for recreational opportunities. Exotic plants have the potential to significantly alter wildlife habitat by creating dense thickets of herbaceous and shrub layers or by displacing native food sources (e.g., Schmidt and Whelan 1999). Therefore, we chose to focus on riparian buffers within the greenways system and to investigate elements of landscape structure that managers and city planners may possibly control.

Using aerial photographs, we delineated 300m long segments of each greenway in Raleigh and Cary. We defined the intact forest canopy surrounding the stream as the riparian buffer, although the full extent of the buffer might extend beyond the protected greenway. We stratified these segments first by average width (narrow (<75m), medium (75m-300m), and wide (>300m), and then by surrounding development density (low density residential <7.5units/ha, high density residential >7.5 units/ha, and commercial) (Novotny 2003). Twenty-three of these segments, each at least 75m from another, were selected to reflect a broad range of widths and development densities, although fewer wide sites were available in this study area. The center of each segment was located in the field using a global positioning system (GPS).

Every greenway segment has a similar management regime, including maintenance of recreational trails and sewer easements. Mixed hardwoods, including sycamore (*Platanus occidentalis*), ironwood (*Carpinus caroliniana*), tulip poplar (*Liriodendron tulipifera*), sweetgum (*Liquidambar styraciflua*), and a variety of oak species (*Quercus* spp.), dominate the canopy of these

riparian forests. While the soils are severely eroded in many sites, soil cations and organic matter are sufficient for plant growth (unpublished data).

### **Characterizing Landscape Context**

Because of the heterogeneous nature of urban landscapes, we characterized landscape context by identifying the types of land uses in the area immediately adjacent to these buffers. Using 1999 leaf-off aerial photographs, we overlaid a grid of 100 evenly spaced points on the 300m by 300m zone on each side of the riparian buffer (Fig. 4.1). We then measured the proportion of points that fell into the following land use classes: canopy cover, building, pavement, bare earth, lawn and water (Hull 2003).

Time since development may be a confounding factor in this study as the disturbance caused by newer developments may affect the adjacent riparian buffer differently than that of older developments. Using property tax records maintained by Wake County, NC (2003), we estimated the age of the surrounding development by averaging the age of 5 randomly chosen structures immediately adjacent to the riparian buffer.

### **Sampling Design**

We defined our species of interest using a list of invasive exotic plant species in this area maintained the North Carolina Botanical Garden (2003). Fourteen of these species are commonly found within riparian buffers (Table 4.1). The majority of these species are still used as ornamental plants, with the exception of *Microstegium vimineum*, a C<sub>4</sub> grass and *Lonicera japonica*, a woody vine.

Within each greenway segment, we established belt transects adjacent to a randomly chosen side of the stream, referred to as riparian transects. Belt transects are commonly used to rapidly inventory forest composition (e.g., Gentry 1988) and have also been used to assess exotic species invasion (Parendes and Jones 2000). These transects, which ran parallel to the stream, encompassed the area beginning at the top of the stream bank and extending 2m out into the adjacent floodplains. Each transect was 300m long and broken into 60 5m subsegments. In each subsegment, we noted the presence and visually estimated percentage cover of each target exotic species. We sampled from July 3 to September 3, 2002 to ensure that plants had reached their full cover of the growing season.

To assess the within-buffer distribution of exotic plants, we surveyed additional transects in the floodplain of the 10 widest greenway sites. We first estimated the width of the floodplain and then chose a random number within that range, representing the number of meters from the stream that the transect would be placed. Within a site, riparian and floodplain transects were at least 10m apart. While these transects were placed parallel to the stream, they often were intersected by trails and sewer easements. Therefore, we truncated the floodplain transects to 150m, the maximum length of continuous uninterrupted floodplain transects for all 10 sites.

## **Data Analysis**

To assess the invasion of buffers, we calculated exotic species richness and mean percentage cover by species for each transect. We summed the mean percentage cover of all species to represent the “extent” of invasion by transect. We also evaluated the invasion of ornamental species by summing percentage cover of 12 species that are common ornamental

plants in our study area (Table 4.1). These data were distributed normally and met the assumptions of parametric tests.

We first examined the variability of invasion within segments in two ways: paired t-tests to test for the difference in response variables between riparian and floodplain transects, and a simple linear regression to test for the effect of distance to stream on the response variables.

We developed forward stepwise regression models (SAS/STAT software, Version 8.1 of the SAS System for Windows, copyright 2001. SAS Institute, Inc.) to reduce the number of independent variables used to describe landscape context. The proportions of canopy cover and building cover in the adjacent landscape were consistently the most significant predictor variables but were not correlated to each other ( $r = -0.23$ ,  $p = 0.22$ ). We also calculated Pearson correlation coefficients to describe the relationship between age of development and these context variables.

We built multiple regression models to test for the effect of width, measured as a continuous variable, percentage canopy, percentage building, and their interactions on all response variables. Non-significant interactions were dropped from the final models.

## **RESULTS**

### **Assessing invasion of riparian buffers**

There was no difference between total percentage cover of exotics in riparian and floodplain transects ( $t = 0.57$ ,  $p = 0.57$ ) and no relationship between distance to stream and invasion ( $r^2=0.03$ ). To maintain equal sample size between sites, we used only the riparian transect data in the multiple regression models.

### **Influence of buffer width**

To assess the influence of width, we need to examine the significance of both width and the interactions between width and the other response variables (Table 4.2). For buffers narrower than 150m, there was a negative relationship between width and invasion, measured as richness (Fig. 2a) and total percentage cover (Fig. 2b) of exotics. However, this relationship was not significant for buffers wider than 150m.

Several exotic species achieved higher percentage cover in narrower buffers. While most individual species' covers were lower in wider buffers, the most common species, *Microstegium vimineum*, *Lonicera japonica*, and *Ligustrum sinense*, exhibited no relationship with width (Table 4.2).

### **Influence of landscape context**

The proportion of canopy cover in the adjacent landscape was the most significant variable in all models (Table 4.2). As canopy cover increased, the exotic species richness ( $p = 0.01$ ) (Fig. 4.2c) and total percentage cover ( $p < 0.0001$ ) (Fig. 4.2d) of all exotics increased. Higher proportions of buildings were not correlated to percentage cover of exotics (Fig. 4.2e) but were positively correlated to total percentage cover of exotics in buffers less than 150m wide (Fig. 4.2f). Neighborhood age was highly correlated to the amount of canopy cover ( $r = 0.66$ ,  $p = 0.001$ ) but not the proportion of building ( $r = 0.11$ ,  $p = 0.61$ ) in the surrounding landscape.

### **Significance of interactions**

Some of the relationships depicted in figure 4.2 appear to be non-linear because of the interaction between response variables. While width and canopy cover are not correlated to each other ( $r = -0.30$ ,  $p = 0.19$ ), the interactions between them are often significant (Table 4.2).

This interaction term indicates that, for buffers wider than 150m, the relationship between the independent variables and the response variables changes (Fig. 4.3a). Because we sampled fewer sites wider than 150m, we focused on these relationships for buffers less than 150m wide (Fig. 4.3b).

The interaction between width and building cover also results in changing relationships between invasion and these response variables with increased width, most notably for buffers narrower than 150m (Fig. 4.4).

## **DISCUSSION**

### **Influence of buffer width on invasion**

Narrower buffers had higher total percentage cover of exotic plants than wider buffers, confirming previous studies that indicate negative correlations between patch size and invasion (Bergelsen et al. 1993, Hutchinson and Vankat 1998, McKinney 2002). For buffers narrower than 150m, these negative relationships are significant with exotic species richness and percentage cover swiftly declining with increasing width. However, once the buffer is 150m or wider, the additional width is not significant.

We attribute these results to several factors. First, while several individual exotic species are more abundant in narrow sites, some species are found in the majority of sites. Two common exotic species, *Microstegium vimineum* and *Lonicera japonica*, have invaded all buffers in this study. Streams may render all of these buffers susceptible to invasion by serving as sources for water-dispersed seeds and maintaining a disturbed environment through flooding (Pysek and Prach 1993, Johansson et al. 1996, Barden 1987, Stohlgren et al. 2000). These buffers are within the same larger watershed and are likely equally exposed to water-dispersed seeds of some

ubiquitous species like *Microstegium vimineum*. Wider buffers are thus no less susceptible than narrower buffers to invasion by a set of exotic species.

Second, the interaction between width and the other response variables warrants further investigation. The narrowest buffers in our dataset are also the most invaded and tend to be surrounded by high proportions of canopy cover and low proportions of building cover. One might also expect narrow buffers to be more influenced by surrounding land use than the wider buffers. Repeating this study in an area with several wide buffers surrounded by different landscape contexts is necessary to fully understand these interactions

Few opportunities exist to protect buffers wider than 150m in urban areas. In fact, the common recommendation for the southeastern United States is to protect 60 meters (200 feet), which is a relatively wide buffer for many densely developed areas (Wenger and Fowler 2000). Therefore, we suggest that these results clearly indicate the need for extending buffer protection to the widest extent possible, with a target of 150m, to resist invasion by exotic species.

### **Influence of landscape context on invasion**

The invasion of forested riparian buffers within urban areas is heavily influenced by the adjacent landscape context. These buffers are surrounded by heterogeneous landscapes that differ in terms of disturbance, density of development, and amount of forest canopy. This heterogeneity may also account for differences in propagule pressure and dispersal opportunities of certain exotic plant species. The amount of canopy cover in the adjacent landscape is the most significant predictor of both the number of exotic species and the total amount of invasion. The amount of canopy cover is much higher in older, residential neighborhoods than in newer subdivisions, commercial complexes, or school grounds.

The positive relationship between exotics and older, more forested neighborhoods could be the result of several processes. Canopy in the adjacent areas essentially extends the size of the riparian buffer and sometimes connects forest patches, thereby facilitating dispersal of exotic species into buffers by providing habitat for seed-dispersing birds. The riparian buffers surrounded by these neighborhoods have higher bird abundances (Hull 2003) and 10 out of 14 target exotic species are bird-dispersed (Table 1). Several exotic species that were most abundant in these sites are ornamental vines (*Hedera helix*, *Vinca* spp., *Wisteria* spp.), which may have spread vegetatively from the adjacent backyards. The vegetation of these yards is generally more established than the new plantings in newer residential developments and therefore may have had enough time to extend into the riparian buffers.

The relationship between the proportion of buildings and invasion could reflect several processes. Denser residential developments may serve as a source for exotic species (McKinney 2002, Pysek et al. 2002), especially common ornamental plants. Dense development could also be related to increased recreational use of greenways, larger sewer easements, or other within-site disturbances. Buildings may also serve as a surrogate for external disturbances such as runoff, sedimentation, and more intense recreational use (McKinney 2002).

Studies that have previously examined the effect of landscape context on patterns of invasion categorize land-use types (DeFerrari and Naiman 1994, Hutchinson and Vankat 1998, Stohlgren et al. 2000). We examined the heterogeneity within urban landscapes and found that land cover types differentially affected the invasion of riparian buffers. Therefore, we suggest that future studies carefully examine the effects of this heterogeneity on natural communities within urban areas (Luck and Wu 2002, Byers et al. 2002).

## **Invasion within buffers**

There are few methods to characterize the general invasion status of a particular site. Previous studies often use number of exotic species to describe the extent of invasion (e.g., Gelbard and Belnap 2003). However, with limited resources, managers often must target certain sites or species where the most damage is occurring (Byers et al. 2002). Therefore, we were interested in the use of belt transects as a relatively efficient way to quickly assess the extent of invasion along streams.

To understand how the level of invasion adjacent to the stream reflected the total invasion of the site, we examined the within-site variability of exotic species distribution. There were no differences between the riparian and floodplain transects, in terms of exotic species richness or total percentage cover. Managers of riparian buffers may be able to quickly survey the extent of invasion and identify the most abundant exotic species by conducting similar belt transects.

We were unable to sample the edges of the wider buffers, due to lack of access to private property. Previous studies have indicated that edges of forest patches may be more invaded (Brothers and Spingarn 1992, Hutchinson and Vankat 1998, Cadenasso and Pickett 2001). It is possible that vegetatively-spreading or wind-dispersed exotic species may have colonized the edge of these forests and have not been able to spread into the interior, because of too little time, or environmental constraints (Brothers and Spingarn 1992, Cadenasso and Pickett 2001). These edges should be examined in future studies, perhaps using remotely sensed images to avoid the complexities of accessing private property.

## CONSERVATION IMPLICATIONS

Sprawling urbanization continues to transform landscapes in the US and throughout the world (Ewing et al. 2002). Riparian buffers, which often cannot be built on because of steep slopes and flooding, provide an opportunity to protect valuable native communities for a suite of ecological functions. These functions may be compromised by the invasion of exotic plants, which can significantly alter both the physical structure (e.g., Schmidt and Whelan 1999) and native species diversity (e.g., Wilcove et al. 1998) in these communities. Understanding how landscape structure influences the invasion of riparian buffers allows us to recommend management efforts to prevent future invasion and perhaps restore currently invaded buffers.

Wider buffers tend to be less invaded by exotic plants but are still susceptible to invasion, especially by common water-dispersed plants such as *Microstegium vimineum*. More importantly, there is a strong relationship between the character of the surrounding landscape and the amount of invasion of these buffers. In our study area, several exotic species are abundantly planted in developments for hedges (e.g., *Elaeagnus pungens*) or ground cover (*Vinca* spp.). These new developments will become future sources of propagules as these exotic ornamentals become established and spread. Regulating the planting of these invasive plants, especially in new developments, may be possible through restrictive covenants, public education and municipal ordinances.

A major challenge for invasive species research is to assist managers in predicting and preventing future invasion (Byers et al. 2002). With's (2002) conceptual model of invasive spread highlights the importance of landscape structure. We specifically defined landscape structure as both width and landscape context because these factors could potentially be manipulated by riparian buffer managers. Emphasizing the impact of landscape structure on the

potential spread of exotic species, some models have been developed (e.g., Higgins et al. 1996, Campbell et al. 2000) but remain largely heuristic. This type of study will inform these modeling efforts, particularly by emphasizing the role of landscape context in heterogeneous urban areas, which are relatively under-studied (Miller and Hobbs 2002). These models can then be used to more accurately predict which sites are susceptible to future invasion.

Few intact forested riparian buffers remain in urban areas. The collocation of greenways within riparian buffers provides an opportunity to manage the buffer for wildlife habitat and native plant communities. Greenway managers are faced with the challenge of prioritizing protection efforts both to enhance the ecological functions and to provide recreational opportunities for the growing urban population. We urge managers and conservationists to protect the widest buffers possible in order to conserve a broad range of ecological functions (e.g., Wenger and Fowler 2000, Hull 2003, Novotny 2003). However, these results highlight the importance of considering the processes that occur within the adjacent landscape when developing conservation and restoration plans. The structure of these heterogeneous landscapes does influence the extent of invasion in riparian buffers. Riparian buffers are valuable but vulnerable ecological communities, and should be protected from future invasion by exotic plant species.

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**Table 4.1.** Invasive exotic species targeted in this study. All are common ornamental plants except for *Microstegium vimineum*, a C4 grass, and *Lonicera japonica*.

Species	Life form	Dispersal mode
<i>Albizia julibrissin</i>	Tree	Wind
<i>Elaeagnus pungens</i>	Shrub	Bird
<i>Elaeagnus umbellata</i>	Shrub	Bird
<i>Hedera helix</i>	Vine	Bird/Vegetative
<i>Ligustrum japonicum</i>	Shrub	Bird
<i>Ligustrum lucidum</i>	Shrub	Bird
<i>Ligustrum sinense</i>	Shrub	Bird
<i>Lonicera japonica</i>	Vine	Bird/Vegetative
<i>Mahonia bealii</i>	Shrub	Bird
<i>Microstegium vimineum</i>	Grass	Water/Wind
<i>Nandina domestica</i>	Shrub	Bird
<i>Rosa multiflora</i>	Shrub	Bird
<i>Vinca</i> spp. *	Vine	Vegetative
<i>Wisteria</i> spp. **	Vine	Vegetative

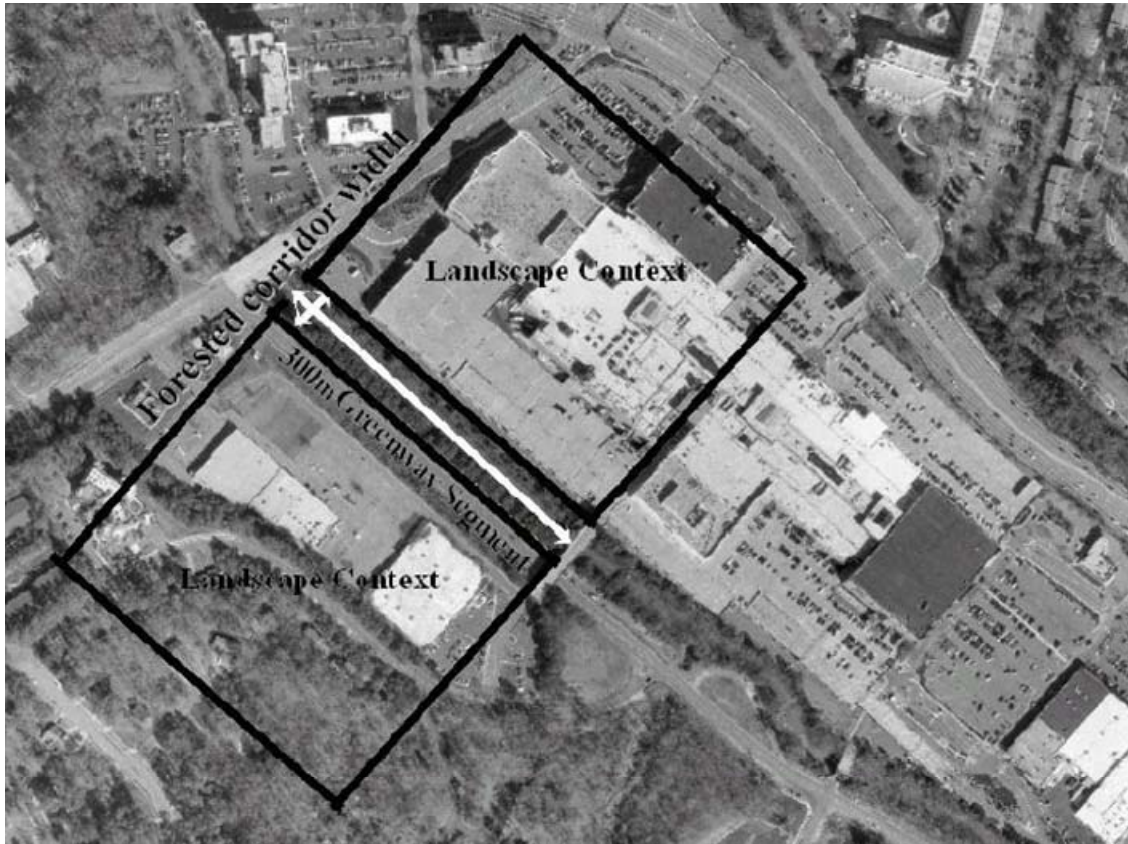
\* *Vinca* spp. includes *Vinca major* and *Vinca minor*.

\*\* *Wisteria* spp. includes *Wisteria floribunda* and *Wisteria sinense*.

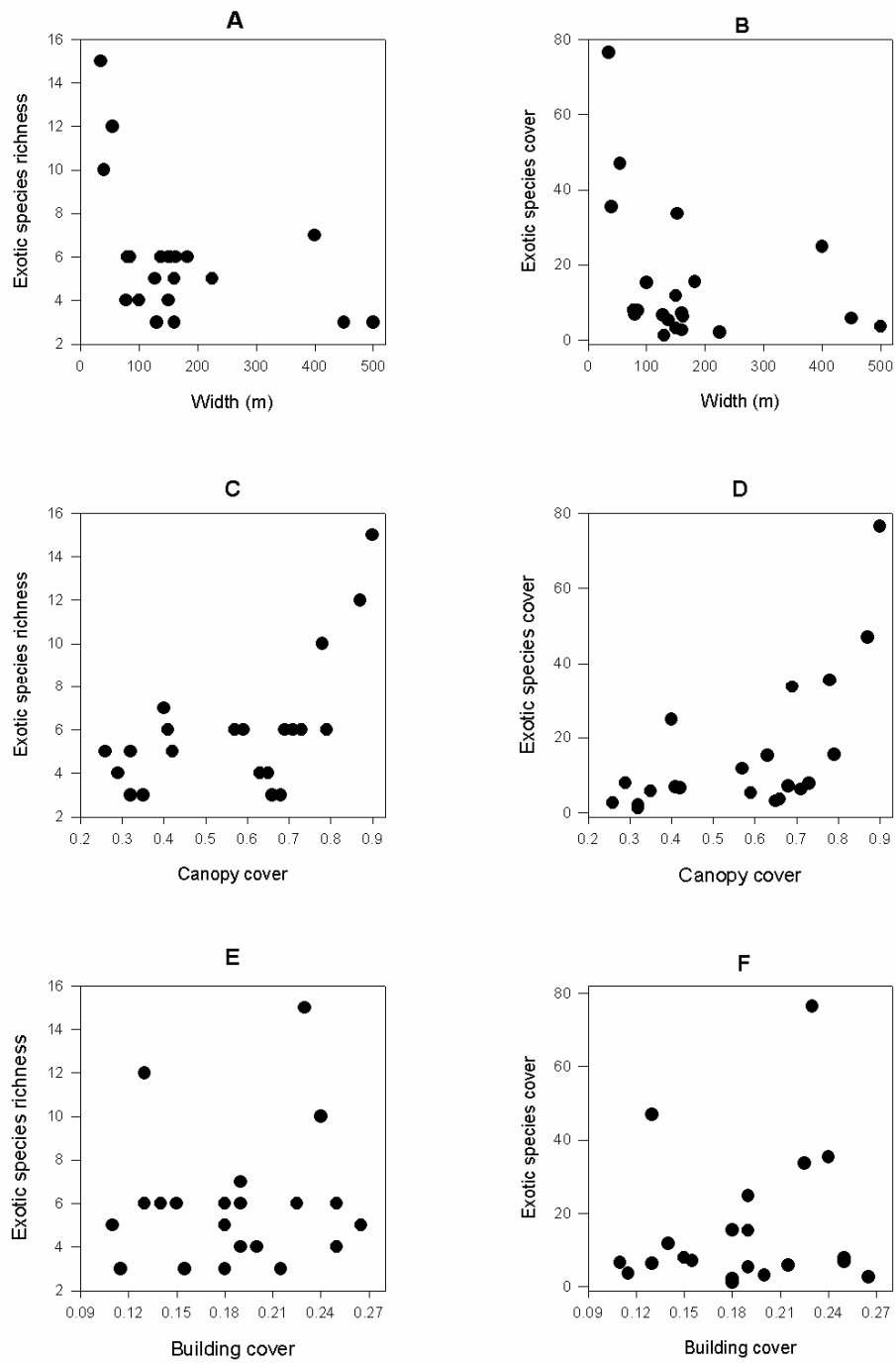
**Table 4.2.** *p* values from multiple regression models, including the width by canopy (W\*C) and width by building (W\*B) interactions.

<b>Response</b>	<b>R<sup>2</sup></b>	<b>Width</b>	<b>Canopy</b>	<b>Building</b>	<b>W*C</b>	<b>W*B</b>
Total exotic cover	0.72	<b>0.004</b>	<b>&lt;0.0001</b>	<b>0.17</b>	<b>0.001</b>	<b>0.02</b>
Species richness	0.36	0.82	0.01	0.98	0.49	0.84
Ornamentals*	0.74	<b>0.001</b>	<b>&lt;0.0001</b>	<b>0.01</b>	<b>0.003</b>	<b>0.01</b>
<i>Hedera helix</i>	0.86	<b>&lt;0.0001</b>	<b>&lt;0.0001</b>	<b>0.0001</b>	<b>&lt;0.0001</b>	<b>&lt;0.0001</b>
<i>Elaeagnus umbellata</i>	0.71	<b>0.0009</b>	<b>&lt;0.0001</b>	<b>0.007</b>	<b>0.0005</b>	<b>0.003</b>
<i>Ligustrum lucidum</i>	0.57	<b>0.01</b>	<b>0.0007</b>	<b>0.04</b>	<b>0.005</b>	<b>0.03</b>
<i>Wisteria</i> spp.	0.57	<b>0.009</b>	<b>0.001</b>	<b>0.01</b>	<b>0.007</b>	<b>0.02</b>
<i>Microstegium vimineum</i>	0.51	0.83	<b>0.02</b>	0.33	0.69	0.96
<i>Ligustrum japonicum</i>	0.50	<b>0.01</b>	<b>0.003</b>	<b>0.01</b>	<b>0.01</b>	<b>0.02</b>
<i>Nandina domestica</i>	0.50	<b>0.01</b>	<b>0.003</b>	<b>0.01</b>	<b>0.01</b>	<b>0.02</b>
<i>Vinca</i> spp.	0.50	<b>0.01</b>	<b>0.003</b>	<b>0.01</b>	<b>0.01</b>	<b>0.02</b>
<i>Elaeagnus pungens</i>	0.43	0.07	<b>0.02</b>	0.64	<b>0.03</b>	0.14
<i>Rosa multiflora</i>	0.43	0.10	<b>0.003</b>	0.66	<b>0.03</b>	0.44
<i>Albizia julibrissin</i>	0.41	0.34	<b>0.02</b>	0.76	0.09	0.71
<i>Mahonia bealii</i>	0.36	0.50	<b>0.03</b>	0.54	0.16	0.93
<i>Ligustrum sinense</i>	0.19	0.80	0.39	0.81	0.77	0.69
<i>Lonicera japonica</i>	0.11	0.82	0.50	0.96	0.97	0.63

\*Ornamentals include all species except *Microstegium vimineum* and *Lonicera japonica*.

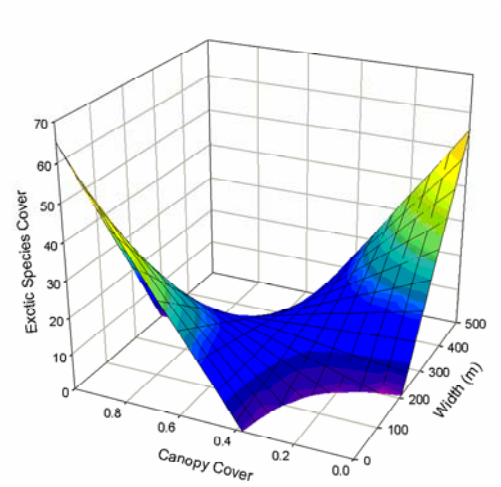


**Figure 4.1.** Aerial photograph (1999) with 300m long greenway segment delineated. Landscape context was measured in 300m x 300m region adjacent to both sides of segment. Figure adapted from Hull (2003).

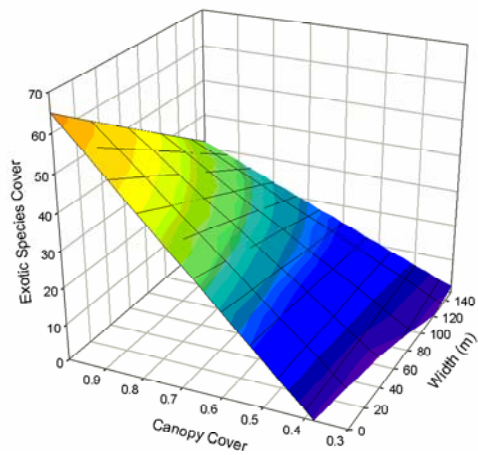


**Figure 4.2.** Relationships between independent variables and exotic species richness and percentage cover.

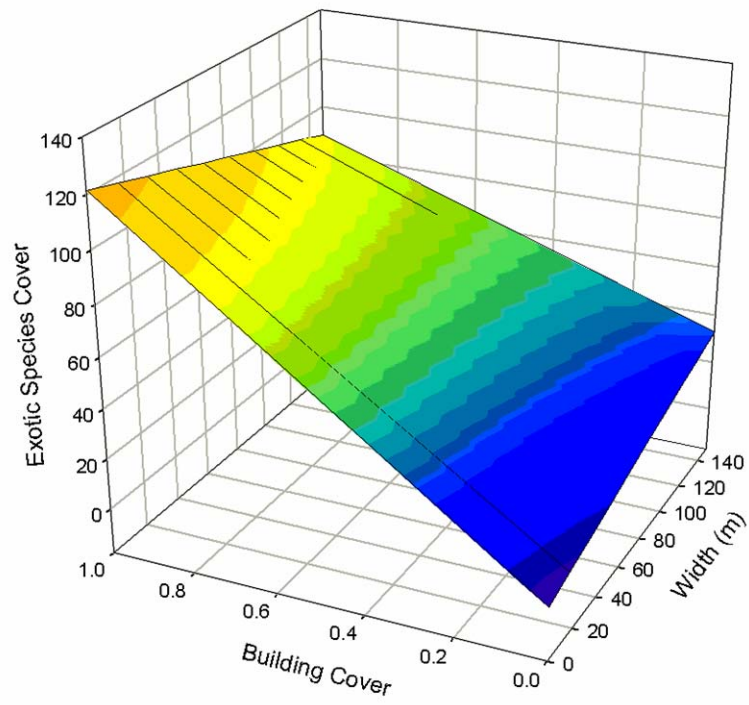
**A**



**B**



**Figure 4.3.** Modeled relationships between exotic species cover, buffer width (m) and surrounding canopy cover (A) with a closer examination of the response of exotic species cover to buffers up to 150m in width (B).



**Figure 4.4.** Modeled relationships between exotic species cover and buffer width (m) and surrounding building cover.

## APPENDICES

**Appendix A.** List of species encountered in vegetation surveys of urban riparian forests (reported in Chapters 2 and 4) and experimental plots (reported in Chapter 3). Nomenclature follows Radford et al. (1968). Growth form (group) and exotic status determined by the USDA PLANTS (2004) database.

SP. CODE	SPECIES	GROUP	EXOTIC
ACERBAR	<i>Acer barbatum</i> Michx.	woody	
ACERNEG	<i>Acer negundo</i> L.	woody	
ACERRUB	<i>Acer rubrum</i> L.	woody	
ACERSAC	<i>Acer saccharinum</i> L.	woody	
ALBIJUL	<i>Albizia julibrissin</i> Durazz.	woody	X
ALLICAN	<i>Allium canadense</i> L.	herb	
ALNUSER	<i>Alnus serrulata</i> (Ait.) Willd.	woody	
AMBRART	<i>Ambrosia artemisiifolia</i> L.	herb	
AMELARB	<i>Amelanchier arborea</i> (Michx. f.) Fern.	woody	
ANDRVIR	<i>Andropogon virginicus</i> L.	grass	
ARISSER	<i>Aristolochia serpentaria</i> L.	vine	
ARISTRI	<i>Arisaema triphyllum</i> (L.) Schott	herb	
ARUNGIG	<i>Arundinaria gigantea</i> (Walt.) Muhl.	grass	
ASARCAN	<i>Asarum canadense</i> L.	herb	
ASCLINC	<i>Asclepias incarnata</i> L.	herb	
ASCLSYR	<i>Asclepias syriaca</i> L.	herb	
ASIMTRI	<i>Asimina triloba</i> (L.) Dunal	woody	
ASPLPLA	<i>Asplenium platyneuron</i> (L.) Oakes	fern	
ASTEDIV	<i>Aster divaricatus</i> L.	herb	
ATHYASP	<i>Athyrium asplenoides</i> (Michx.) A.A.E	fern	
AUREVIR	<i>Aureolaria virginica</i> (L.) Pennell	herb	
BACCHAL	<i>Baccharis halimifolia</i> L.	woody	
BETUNIG	<i>Betula nigra</i> L.	woody	
BOEHCYL	<i>Boehmeria cylindrica</i> (L.) Sw.	herb	
BOTYVIR	<i>Botrychium virginianum</i> (L.) Sw.	fern	
BRACERE	<i>Brachyelytrum erectum</i> (Schreber) Beauvois.	grass	
BROMJAP	<i>Bromus japonicus</i> Thun. Ex Murr.	grass	X
CACAATR	<i>Cacalia atriplicifolia</i> L.	herb	
CAMPRAD	<i>Campsis radicans</i> (L.) Seem. exBureau	vine	
CARESP	<i>Carex sp.</i>	grass	
CARECRI	<i>Carex crinita</i> Lamb.	grass	
CARPCAR	<i>Carpinus caroliniana</i> Walt.	woody	
CARYSP	<i>Carya sp.</i>	woody	
CASSFAS	<i>Cassia fasciculata</i> Michx.	herb	
CASSNIC	<i>Cassia nictitans</i> L.	herb	
CELAORB	<i>Celastrus orbiculata</i> Thunb.	vine	X
CENTVIR	<i>Centrosema virginianum</i> (L.) Benth.	vine	
CERCCAN	<i>Cercis canadensis</i> L.	woody	
CHASLAX	<i>Chasmanthium laxum</i> (L.) Yates	grass	
CINNARU	<i>Cinna arundinacea</i> L.	grass	
CLEMVIR	<i>Clematis virginiana</i> L.	vine	
CLITMAR	<i>Clitoria mariana</i> L.	vine	
COMMCOM	<i>Commelina communis</i> L.	herb	X
CORNFLO	<i>Cornus florida</i> L.	woody	
CORYAME	<i>Corylus americana</i> Walt.	woody	
CRATSP	<i>Crataegus sp.</i>	woody	
CUPHSP	<i>Cuphea sp.</i>	herb	
CYPEODO	<i>Cyperus odoratus</i> L.	grass	
DENTSP	<i>Dentaria sp.</i>	herb	
DESM PAN	<i>Desmodium paniculatum</i> (L.) DC.	grass	
DICASP	<i>Dichanthelium sp.</i>	grass	
DIOSVIL	<i>Dioscorea villosa</i> L.	vine	
DIOSVIR	<i>Diospyros virginiana</i> L.	woody	
DUCHIND	<i>Duchesnea indica</i> (Andr.) Focke	herb	X
ELAEPUN	<i>Elaeagnus pungens</i> Thunb.	woody	X

Appendix A. continued

SP. CODE	SPECIES	GROUP	EXOTIC
ELAEUMB	<i>Elaeagnus umbellata</i> Thunb.	woody	X
ELEPTOM	<i>Elephantopus tomentosus</i> L.	herb	
ELYMSP	<i>Elymus spicatus</i> (Pursh) Gold	grass	
ELYMVIR	<i>Elymus virginicus</i> L.	grass	
EPIFVIR	<i>Epifagus virginiana</i> (L.) W. Bart.	herb	
ERAGHIR	<i>Eragrostis hirsuta</i> (Michx.) Nees	grass	
ERIGCAN	<i>Erigeron canadensis</i> L.	grass	
EUONAME	<i>Euonymus americana</i> (L.)	woody	
EUPACAP	<i>Eupatorium capillifolium</i> (Lam.) Small	herb	
FAGUGRA	<i>Fagus grandifolia</i> Ehrh.	woody	
FRAXSP	<i>Fraxinus</i> sp.	woody	
GALISP	<i>Galium</i> sp.	herb	
GAULPRO	<i>Gaultheria procumbens</i> L.	herb	
GELSSEM	<i>Gelsemium sempervirens</i> (L.) St. Hil.	vine	
GERACAR	<i>Geranium carolinianum</i> L.	herb	
GEUMSP	<i>Geum</i> sp.	herb	
GLECHED	<i>Glechoma hederacea</i> L.	herb	X
GNAPOBT	<i>Gnaphalium obtusifolium</i> L.	herb	
GYMNAMB	<i>Gymnopogon ambiguus</i> (Michx.) B.S.P.	grass	
HEDEHEL	<i>Hedera helix</i> L.	vine	X
HELISP	<i>Helianthus</i> sp.	herb	
HEMEFUL	<i>Hemerocallis fulva</i> (L.) L.	herb	X
HEPAAME	<i>Hepatica americana</i> (DC.) Ker.	herb	
HEXAARI	<i>Hexastylis arifolia</i> (Michx.) Small	herb	
HUMUJAP	<i>Humulus japonicus</i> Sieb. & Zucc.	vine	X
HUMULUP	<i>Humulus lupulus</i> L.	herb	
HYPESP	<i>Hypericum</i> sp.	herb	
HYSTPAT	<i>Hystrix patula</i> Moench.	grass	
ILEXDEC	<i>Ilex decidua</i> Walt.	woody	
ILEXOPA	<i>Ilex opaca</i> Ait.	woody	
ILEXVER	<i>Ilex verticillata</i> L.	woody	
IMPACAP	<i>Impatiens capensis</i> Meerb.	herb	
IPOMLAC	<i>Ipomoea lacunosa</i> L.	vine	
IPOMPUR	<i>Ipomoea purpurea</i> (L.) Roth	vine	
IPOMSP	<i>Ipomea</i> sp.	vine	
IRISSP	<i>Iris</i> sp.	vine	
JASMNUD	<i>Jasminum nudiflorum</i> Lindl.	herb	
JUNCSP	<i>Juncus</i> sp.	grass	
JUNIVIR	<i>Juniperus virginiana</i> L.	woody	
LEERVIR	<i>Leersia virginica</i> Willd.	grass	
LESPCUN	<i>Lespedeza cuneata</i> (Dumont) G. Don.	herb	X
LIGUJAP	<i>Ligustrum japonicum</i> Thunb.	woody	X
LIGULUC	<i>Ligustrum lucidum</i> Ait. f.	woody	X
LIGUSIN	<i>Ligustrum sinense</i> Lour.	woody	X
LINDBEN	<i>Lindera benzoin</i> (L.) Blume	woody	
LIQUSTY	<i>Liquidambar styraciflua</i> L.	woody	
LIRIMUS	<i>Liriope muscari</i> (Dcne.) Bailey	herb	X
LIRITUL	<i>Liriodendron tulipifera</i> L.	woody	
LONIJAP	<i>Lonicera japonica</i> Thunb.	vine	X
LYCOSP	<i>Lycopodium</i> sp.	club moss	
LYCOVIR	<i>Lycopus virginicus</i> L.	herb	
MAGNVIR	<i>Magnolia virginiana</i> L.	woody	
MAHOBEA	<i>Mahonia bealei</i> (Fortune) Carr.	woody	X
MENTSP	<i>Mentha</i> sp.	herb	
MICRVIM	<i>Microstegium vimineum</i> (Trin.) A. Camus	grass	X
MITCREP	<i>Mitchella repens</i> L.	herb	
MORUALB	<i>Morus alba</i> L.	woody	X
MYRICER	<i>Myrica cerifera</i> (L.) Small	woody	
NANDDOM	<i>Nandina domestica</i> Thunb.	woody	X
NYSSSLY	<i>Nyssa sylvatica</i> Marsh.	woody	
OPHISP	<i>Ophiopogon</i> sp.	grass	

Appendix A. continued

SP. CODE	SPECIES	GROUP	EXOTIC
OPHIVUL	<i>Ophioglossum vulgatum</i> L.	fern	
OSTRVIR	<i>Ostrya virginiana</i> (P. Mill.) K. Koch	woody	
OXALSP	<i>Oxalis</i> sp.	herb	
OXYDARB	<i>Oxydendrum arboreum</i> (L.) DC.	woody	X
PARTQUI	<i>Parthenocissus quinquefolia</i> (L.) Planch.	vine	
PASPDIL	<i>Paspalum dilatatum</i> Poir.	grass	X
PASSSP	<i>Passiflora</i> sp.	vine	
PHYLSP	<i>Phyllostachys</i> sp.	grass	X
PHYTAME	<i>Phytolacca americana</i> L.	herb	
PINUTAE	<i>Pinus taeda</i> L.	woody	
PLANARI	<i>Plantago aristata</i> Michx.	herb	
PLANRUG	<i>Plantago rugelii</i> Dcne.	herb	
POLYARI	<i>Polystichum acrostichoides</i> (Michx.) Schott	fern	
POLYPER	<i>Polygonum persicaria</i> L.	herb	
POLYHYD	<i>Polygonum hydropiperoides</i> Michx.	herb	
PRUNSER	<i>Prunus serotina</i> Ehrh.	woody	
PRUNVUL	<i>Prunella vulgaris</i> L.	herb	
PYRUCOM	<i>Pyrus communis</i> L.	woody	X
QUERALB	<i>Quercus alba</i> L.	woody	
QUERNIG	<i>Quercus nigra</i> L.	woody	
QUERPHE	<i>Quercus phellos</i> L.	woody	
QUERRUB	<i>Quercus rubra</i> L.	woody	
QUERSP	<i>Quercus</i> sp.	woody	
QUERVEL	<i>Quercus velutina</i> Lam.	woody	
RANUSP	<i>Ranunculus</i> sp.	herb	
RHODMAX	<i>Rhododendron maximum</i> L.	woody	
RHUSCOP	<i>Rhus copallinum</i> L.	woody	
RHUSGLA	<i>Rhus glabra</i> L.	woody	
ROSAMUL	<i>Rosa multiflora</i> Thub. Ex Murr	woody	X
RUBUARG	<i>Rubus argutus</i> Link	vine	
RUBUSP	<i>Rubus</i> sp.	vine	
SALVLYR	<i>Salvia lyrata</i> L.	herb	
SAMBCAN	<i>Sambucus canadensis</i> L.	woody	
SANICAN	<i>Sanicula canadensis</i> L.	herb	
SASSALB	<i>Sassafras albidum</i> (Nutt.) Nees	woody	
SCUTELL	<i>Scutellaria elliptica</i> Muhl. ex Spreng.	herb	
SIDARHO	<i>Sida rhombifolia</i> L.	herb	
SMILGLA	<i>Smilax glauca</i> Walt.	vine	
SMILROT	<i>Smilax rotundifolia</i> L.	vine	
SMILSP	<i>Smilax</i> sp.	vine	
SOLAAME	<i>Solanum americanum</i> P. Mill.	herb	
SOLASP	<i>Solanum</i> sp.	herb	
SOLISP	<i>Solidago</i> sp.	herb	
STAPTRI	<i>Staphylea trifolia</i> L.	woody	
STELMED	<i>Stellaria media</i> (L.) Vill.	herb	X
SYMPTIN	<i>Symplocos tinctoria</i> (L.) L'Her.	woody	
THALSP	<i>Thalictrum</i> sp.	herb	
THELHEX	<i>Thelypteris hexagonoptera</i> (Michx.) Weatherby	fern	
THELNOV	<i>Thelypteris noveboracensis</i> (L.) Nieuwl.	fern	
TIARCOR	<i>Tiarella cordifolia</i> L.	herb	
TIPUDIS	<i>Tipularia discolor</i> (Pursh) Nutt.	herb	
TOXIRAD	<i>Toxicodendron radicans</i> (L.) Kuntze	vine	
TRIFSP	<i>Trifolium</i> sp.	herb	
ULMUALA	<i>Ulmus alata</i> Michx.	woody	
ULMURUB	<i>Ulmus rubra</i> Muhl.	woody	
UNIOLAT	<i>Uniola latifolia</i> Michaux.	grass	
UVULPER	<i>Uvularia perfoliata</i> L.	herb	
UVULSES	<i>Uvularia sessilifolia</i> L.	herb	
VACCARB	<i>Vaccinium arboreum</i> Marsh.	woody	
VACCSTA	<i>Vaccinium stamineum</i> L.	woody	
VACCVAC	<i>Vaccinium vacillans</i> Torrey.	woody	

Appendix A. continued

SP. CODE	SPECIES	GROUP	EXOTIC
VERBOCC	<i>Verbesina occidentalis</i> (L.) Walt.	herb	
VIBUACE	<i>Viburnum acerifolium</i> L.	woody	
VIBUDEN	<i>Viburnum dentatum</i> L.	woody	
VIBUNUD	<i>Viburnum nudum</i> L.	woody	
VIBUPRU	<i>Viburnum prunifolium</i> L.	woody	
VIBURAF	<i>Viburnum rafinesquianum</i> J.A. Schultes	woody	
VICISP	<i>Vicia</i> sp.	herb	
VINCSP	<i>Vinca</i> sp.	vine	X
VIOLSP	<i>Viola</i> sp.	herb	
VITIAES	<i>Vitis aestivalis</i> Michx.	vine	
VITIROT	<i>Vitis rotundifolia</i> Michx.	vine	
VITIVUL	<i>Vitis vulpina</i> L.	vine	
WISTSP	<i>Wisteria</i> sp.	vine	X
WOODARE	<i>Woodwardia areolata</i> (L.) T. Moore	fern	
XANTSIM	<i>Xanthorhiza simplicissima</i> Marsh.	herb	
ZEPHATA	<i>Zephyranthes atamasco</i> (L.)	herb	
Unknown 1	Unknown 1	grass	
Unknown 2	Unknown 2	grass	
Unknown 3	Unknown 3	herb	
Unknown 4	Unknown 4	grass	
Unknown 5	Unknown 5	grass	
Unknown 6	Unknown 6	grass	
Unknown 7	Unknown 7	herb	
Unknown 8	Unknown 8	grass	
Unknown 9	Unknown 9	woody	
Unknown 10	Unknown 10	grass	
Unknown 11	Unknown 11	grass	
Unknown 12	Unknown 12	grass	
Unknown 13	Unknown 13	grass	
Unknown 14	Unknown 14	herb	
Unknown 15	Unknown 15	herb	
Unknown 16	Unknown 16	herb	

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Radford, A.E., H.E. Ahles and C.R. Bell. 1968. *Manual of the vascular flora of the Carolinas*. Chapel Hill: University of North Carolina Press.

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Appendix B. Species presence data by site from surveys of 23 urban riparian sites.

SP. CODE	OP1	SC3	W04	PW1	BC9	OP2	PC3	TA2	W2	BC8	DU3	W01	TA1	GA1	LW1	PC2	BC3	BD1	SM2	PC1	BD3	BC5	H12
ACERBAR	1	1				1			1							1	1			1	1	1	
ACERNEG			1																				
ACERRUB	1	1	1	1	1	1		1	1	1	1	1					1		1	1			
ACERSAC																						1	
AESCSLY																	1						
ALBIJUL																		1					1
ALNUSER			1				1					1					1						1
AMBRART																1							
AMELARB										1													
ANDRVIR											1												1
ARISSE	1	1		1					1														
ARISTRI										1	1								1				
ARUNGIG										1	1	1											
ASARCAN										1													1
ASCLINC			1																				
ASIMTRI	1																						
ASPLPLA	1			1																			
ASTEDIV	1	1									1												1
ATHYASP							1																
AUREVIR											1										1		
BACCHAL											1					1							
BETUNIG									1		1						1						
BOEHCYL					1	1					1					1		1					1
BOTYVIR			1					1	1				1										1
BRACERE										1													
BROMJAP																						1	
CACAATR			1																				
CAMPRAD	1			1	1	1		1	1				1							1			
CARESP		1		1		1					1						1						
CARPCAR		1			1	1		1	1	1		1				1	1	1	1	1			1
CARYSP	1			1	1	1		1	1	1	1		1				1	1	1				1
CASSFAS			1			1					1												
CASSNIC								1															
CELAORB																						1	
CENTVIR		1	1	1																			
CERCCAN				1	1		1			1							1						1
CHASLAX		1		1		1		1	1	1	1					1	1			1	1	1	1
CINNARUN																							
CLEMVIR									1					1	1	1			1				
CLITMAR	1		1	1	1	1	1		1	1	1	1		1		1	1	1					1
COMMCOM																1							
CORNFLO		1	1	1	1	1	1	1	1	1	1	1	1	1			1		1		1	1	1
CORYAME											1							1					
CYPEODO	1							1															
DENTSP																							1
DESPAN		1																					
DICHSP	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
DIOSVIR								1			1												
DUCHIND														1		1		1				1	1
ELAEPUN				1							1								1				1
ELAEUMB	1													1				1	1			1	1
ELEPTOM					1		1								1	1					1		
ELYMSPI		1									1				1								
ELYMVIR																1							
EPIFVIR			1																				
EUONAME		1	1	1	1	1		1	1	1	1	1	1	1		1	1		1	1	1	1	1
EUPACAP			1									1											
FAGUGRA						1	1				1						1		1	1			1
FRAXSP		1			1		1	1		1			1			1	1		1	1			

Appendix B. continued.

SP. CODE	OP1	SC3	W04	PW1	BC9	OP2	PC3	TA2	W2	BC8	DU3	W01	TA1	GA1	LW1	PC2	BC3	BD1	SM2	PC1	BD3	BC5	H12	
GALISP			1	1	1					1		1	1			1	1							
GAULPRO									1															
GEUMSO				1			1	1			1						1							
GLECHED		1													1	1			1				1	
GNAPOBT											1													
GYMNAMB	1			1													1				1	1		
HEDEHEL															1		1		1				1	
HELIANT	1																							
HEMEFUL															1								1	
HEPAAME						1		1		1							1						1	
HEXAARI				1	1			1		1		1	1			1	1			1		1	1	
HUMUJAP																1								
HUMULUP																1								
HYPESP		1	1					1			1										1			1
HYSTPAT			1															1						
ILEXDEC										1														
ILEXOPA		1					1	1			1	1									1			
ILEXVER		1										1									1			1
IMPACAP										1	1				1		1							1
IPOMSP	1				1											1						1		
IRISSP																							1	
JUNCSP								1				1	1			1							1	
JUNIVIR			1					1				1							1				1	
LEERVIR																1					1			1
LESPCUN			1	1		1		1	1		1		1										1	
LIGUJAP								1																
LIGUSIN							1		1			1			1	1	1		1			1		
LINDBEN	1						1	1	1	1		1					1	1						1
LIGUSTY		1	1		1	1		1		1	1	1							1		1			1
LIRIMUS	1		1	1	1	1		1	1	1		1			1	1	1		1			1	1	
LIRITUL	1	1	1	1	1						1		1											
LONIJAP	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
LYCOSP																							1	
LYCOVIR						1		1			1													1
MAHOBEA															1									
MICRVIM	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
MITCREP						1		1	1													1		
MORUALB														1						1			1	
MYRICER					1						1													
NANDDOM	1																							
NYSSSLY			1		1					1		1									1			
OPHIOSP															1									1
OSTRVIR						1						1						1	1				1	
OXALIS											1				1									
OXYDARB				1						1						1		1						
PARTQUI	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
PASS	1				1		1	1																
PHYLSP												1			1					1	1			
PHYTAME						1			1										1					1
PINUTAE		1	1	1		1		1			1	1					1							
PLANARI											1													
PLANRUG						1																		
POLYARI		1	1	1	1		1	1		1	1	1						1					1	
POLYHYD								1																
POLYPER															1									1
PRUNSER	1	1	1	1	1	1	1	1	1	1	1	1	1			1			1	1	1	1	1	1
PRUNVUL								1														1	1	
PYRUCOM																	1							

Appendix B. continued.

SP. CODE	OP1	SC3	W04	PW1	BC9	OP2	PC3	TA2	W2	BC8	DU3	W01	TA1	GA1	LW1	PC2	BC3	BD1	SM2	PC1	BD3	BC5	H12
QUERALB			1		1	1		1	1	1	1	1			1			1		1	1		1
QUERNIG	1	1	1				1			1									1			1	
QUERPHE			1		1			1	1	1	1	1			1		1	1	1	1	1	1	1
QUERRUB			1		1	1	1		1	1	1				1			1		1			1
QUERVEL																		1					
RANUSP										1							1	1			1		1
RHUSCOP												1											
RHUSGLA				1																			
ROSAMUL	1					1									1								1
RUBUSP			1	1	1		1	1	1	1		1		1	1	1		1				1	1
SALVLYR				1	1						1						1						1
SAMBCAN				1	1					1	1		1		1	1	1		1		1		
SANICAN																	1						
SASSALB				1						1			1										
SMILSP	1	1	1	1	1	1	1	1	1	1	1	1	1	1			1	1		1	1		1
SOLASP									1														
SOLISP			1	1	1	1	1	1	1		1	1	1	1			1	1	1	1	1	1	1
STAPTRI	1																						
THALSP									1										1				
THELHEX								1											1				
THELNOV					1			1				1					1			1			
TIARCOR											1								1				
TOXIRAD	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1
TRIFREP																1		1					1
ULMJALA					1																		
ULMURUB	1				1			1		1													
UNIOLAT	1									1									1				
UVULSES					1						1						1			1	1		1
VACCARB					1				1		1		1								1		
VACCSTA			1																				
VERBOCC	1						1			1													
VIBUACE			1	1				1			1		1					1					1
VIBUDEN											1												
VIBUNUD																			1				
VIBUPRU			1		1			1		1											1		1
VIBURAF			1	1	1	1	1	1	1	1			1	1				1					1
VINCA																1							
VIOLSP			1	1		1	1	1	1	1					1		1	1	1	1	1	1	1
VITIAES				1						1													
VITIROT	1	1	1	1	1	1	1	1		1		1	1	1				1		1	1		1
VITIVUL																							
WSTSP																	1		1				
WOODARE												1											
XANTSIM			1	1		1	1	1		1	1		1					1		1	1	1	1
unknown 1	1																						
unknown 2																			1				
unknown 3																		1					
unknown 4																			1				
unknown 5			1	1	1	1	1	1	1	1	1	1	1	1			1	1		1	1	1	1
unknown 6																							
unknown 7																							
unknown 8																							
unknown 9					1																		
unknown 10																							1
unknown 11																							
unknown 12														1									
unknown 13																							
unknown 14																							
unknown 15								1															1

Appendix C. Mean values for soil nutrients and canopy cover for 23 urban riparian sites.

	BC3	BC5	BC8	BC9	BD1	BD3	DU3	GA1	HI2	IW2	LW1	OP1	OP2	PC1	PC2	PC3	PW1	SC3	SM2	TA1	TA2	W01	W04
<b>P</b>	1.95	5.88	1.93	6.45	16.5	15.4	4.23	9.93	8.05	11.6	27.5	4.03	5.4	2.15	4.25	6.05	10.7	1.1	1.52	3.32	7.82	8.75	16.7
<b>K</b>	0.13	0.16	0.3	0.16	0.32	0.21	0.54	0.24	0.26	0.37	0.21	0.34	0.24	0.15	0.25	0.16	0.15	0.23	0.19	0.16	0.26	0.19	0.26
<b>Ca</b>	3.61	3.78	2.97	3.46	7.49	3.86	2.22	5.37	3.9	2.6	4.72	3.93	3	2.33	6.41	5.86	5.32	1.48	2.26	6.28	3.74	3.31	3.54
<b>Mg</b>	1.55	1.63	1.19	1.55	1.47	1.05	0.75	1.47	1.13	0.79	0.57	1.17	1.04	1.18	2	1.53	1.98	0.66	0.91	2.19	1.5	1.33	1.35
<b>Ila</b>	0.05	0.1	0.03	0.1	0.1	0.05	0.02	0.1	0.05	0	0.08	0.1	0.06	0	0.08	0.07	0.07	0.1	0	0.08	0.07	0.05	0.05
<b>WtVol</b>	1.17	0.96	0.94	0.89	1.02	1.23	1.11	0.88	1.06	1.14	0.93	1.03	1.26	1.2	1.18	1.06	1.06	1.17	1.02	0.84	1.02	1.04	1.04
<b>pH</b>	5.37	5.37	5.03	4.93	6	5.28	5.43	5.6	5.62	5.3	5.63	5.78	5.4	5.08	6.13	5.62	5.17	4.92	5	5.42	4.93	5.02	5.02
<b>BufAC</b>	1.95	1.87	2.17	2.72	1.38	2.1	1.13	1.8	1.52	1.45	1.25	1.22	1.53	2.18	1.17	1.92	2.08	1.93	2.35	1.82	2.38	1.93	2.17
<b>Sum</b>	5.4	5.44	3.02	5.13	9.33	5.09	3.16	5.6	5.13	3.37	5.53	5.34	4.22	3.6	8.63	7.51	7.52	2.34	3.36	4.25	5.3	4.87	5.24
<b>CEC</b>	7.1	7.15	6.45	6.25	9.03	5.62	4.23	8.75	6.5	4.78	6.65	6.45	5.67	5.7	9.7	6.25	8.03	4.12	5.72	10.4	7.63	5.27	7.18
<b>BaseSat</b>	50.3	72	55.7	56.3	84.2	65.2	61.8	78.7	54	58.3	69.5	67.7	70.3	58.7	69.2	63.8	75.3	35.5	51.7	79.3	66.8	55.3	68.8
<b>Mn</b>	52.7	74.5	79	94.3	68.2	49.5	39.6	63.3	43.2	47.5	15	63.5	47.3	38.7	64	71	100	57	81.1	60.8	66.5	36.1	43.8
<b>Zn</b>	4.7	5.52	2.87	3.15	10.6	13	2.68	13.9	9.63	4.03	35.2	4.97	4.07	8.38	8.58	6.85	3.57	2.87	3.78	3.73	4.95	2.75	6.37
<b>Cu</b>	1.12	1.27	0.73	0.88	1.95	2.35	1.38	2.75	1.05	0.88	13.7	1.35	1.02	1.38	1.98	1.5	1.02	0.78	1.23	1.77	1.32	0.72	0.75
<b>Humic</b>	0.3	0.48	0.34	0.37	0.35	0.47	0.36	0.42	0.39	0.32	0.34	0.33	0.28	0.29	0.31	0.38	0.33	0.65	0.34	0.4	0.43	0.33	0.41
<b>Canopy</b>	79.2	63.9	96.2	92	94	77.1	43.7	91.8	71.3	96	91.5	66.3	61.3	95	95.6	90.5	93.3	73.8	95.4	79.8	81.1	96.4	74.7



Appendix D. continued.

SP. CODE	1B	2B	3B	4B	5B	6B	7B	8B	9B	10B	11B	12B	13B	14B	15B	16B	17B	18B	19B	20B	21B	22B	23B	24B	25B	26B	27B	28B	29B	30B	31B	32B	33B	34B	35B	36B		
ACERRUB	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	2	3	0	0	0	6			
ALUCAN	2	4	2	2	4	4	3	4	2	3	4	2	4	4	4	2	4	3	2	0	0	4	0	2	0	4	2	2	0	0	0	0	0	0	0	0		
AMBRARY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
AMELARB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
ASPLFLA	0	0	0	0	0	0	0	0	0	2	0	0	2	2	2	2	2	0	2	0	0	2	0	0	2	0	0	2	2	2	4	3	2	0	3	0		
ASTERSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
BOEHCYL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
BOTRVIR	2	0	0	0	2	0	2	2	2	0	2	0	2	0	0	2	0	0	0	0	0	0	0	0	0	0	2	2	0	0	2	2	0	0	2	2	0	
CAMPRAD	0	0	0	2	2	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2	3	0	3	0	0	0		
CARYSP	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	0		
CORNFLA	0	0	0	2	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2	2	3	0	0		
CRATSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
CUPHSP	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
DIOSVIR	3	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	2	0	2	0	0	0	0	0	0	0	0	0	0		
DUCHND	0	0	0	4	0	2	2	0	0	3	2	4	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0		
ELAEUMB	7	0	3	7	0	4	6	4	0	2	0	6	0	7	5	2	0	6	0	2	6	2	6	0	0	2	5	4	0	8	4	0	7	0	2	7		
FAGUGRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0		
FRAXSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
GALISP	0	0	0	0	0	0	0	0	0	2	2	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	
GELSCAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GERACAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
HEDEHEL	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
ILEXOPA	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
POMPUR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
JASMNLD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LESPCUN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIGUSIN	4	0	3	3	0	2	0	3	0	5	0	4	0	6	6	0	5	0	5	4	3	4	0	0	4	6	2	0	2	0	0	0	0	0	0	4	0	
LINDBEN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIQUSTY	0	0	0	5	0	6	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	5	6	0	2	0	4	6	4	0		
LIRMUS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIRITUL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	7	0	0	
LONJAP	5	0	4	6	0	6	6	5	0	3	0	5	0	2	4	4	0	3	0	2	5	6	0	0	5	6	4	0	6	6	0	7	0	2	3	0	0	
MENTS	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
MICRYM	0	0	2	2	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	
NANDOOM	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
NYSSSLV	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
OPHVIR	2	0	0	0	0	0	0	2	0	2	2	0	0	0	2	0	2	0	2	0	0	0	0	0	0	0	2	2	0	2	0	2	0	2	0	2	0	0
PARTOLI	3	0	2	3	3	0	2	2	2	0	2	2	2	2	2	2	2	2	3	2	4	2	2	2	2	2	2	2	0	2	2	4	0	2	2	0	0	
PHYTAME	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	
PINUTAE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
POLYPER	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
PRUNSER	2	0	2	2	2	2	2	0	2	2	2	0	0	2	0	0	0	2	2	0	0	0	0	0	0	2	7	2	2	0	2	0	2	0	2	0	0	
QUERARB	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	4	0	0	0	0	0	0	0	0	0	
QUERCUS	2	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
QUERPHE	2	0	0	0	0	0	0	2	0	0	2	2	2	0	0	0	0	0	2	0	2	0	0	0	2	0	0	2	0	2	0	2	2	0	2	0	0	
QUERRUB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
RANUSP	4	3	4	2	3	2	2	2	2	2	2	0	2	2	2	2	0	0	0	0	2	0	0	2	0	2	0	0	0	2	2	2	0	2	0	2	0	
ROSAMUL	0	0	0	0	0	0	3	0	0	3	0	5	0	4	2	0	6	0	3	3	0	4	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	
RUBUARG	3	0	2	0	0	0	0	0	0	0	2	0	0	0	2	0	0	2	0	0	2	0	0	2	0	2	0	2	0	0	2	0	2	0	2	0	0	
SMLGLA	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2	0	0	2	0	2	0	0	0	0	0	0	0	2	0	2	0	0	
SMLROT	2	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0	
SMLSP	0	0	0	0	0	0	0	2	0	0	2	0	0	0																								





Appendix D. continued

SP. CODE	1E	2E	3E	4E	5E	6E	7E	8E	9E	10E	11E	12E	13E	14E	15E	16E	17E	18E	19E	20E	21E	22E	23E	24E	25E	26E	27E	28E	29E	30E	31E	32E	33E	34E	35E	36E				
ACERRUB	0	0	0	0	2	0	2	0	0	0	0	0	2	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	6			
ALUCAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0			
AMBRARY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0			
AMELARB	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0			
ASPLFLA	0	0	0	0	0	0	0	0	5	0	3	3	2	3	2	3	0	2	0	0	2	4	2	0	4	0	0	2	3	4	4	5	2	0	5	0	0			
ASTERSP	0	0	0	2	0	0	0	0	0	2	0	2	0	0	2	3	0	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0			
BOEHCYL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0			
BOTRVIR	0	0	0	0	2	4	2	3	2	2	2	0	2	0	0	0	0	2	0	0	0	0	0	0	0	2	2	2	2	2	0	2	0	2	2	4	0	0		
CAMPRAD	0	3	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	3	2	0	2	0	0	0	0			
CARYSP	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0			
CORNFLA	0	0	0	0	0	6	0	2	0	0	0	0	0	0	0	2	0	0	0	2	0	0	0	0	0	0	0	2	3	0	0	2	3	3	0	0	0			
CRATSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0			
CUPHSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
DIOSVIR	5	3	0	2	0	0	0	0	0	2	0	0	0	0	2	0	0	0	0	0	2	0	3	0	2	3	0	0	0	0	0	0	0	0	0	0	0	0		
DUCHND	0	0	0	4	2	2	0	0	0	4	4	5	2	2	0	5	2	2	2	0	0	0	0	0	0	0	0	0	2	0	0	0	0	3	0	0	0	0		
ELAEUMB	6	0	2	7	0	2	8	2	0	2	6	0	7	4	4	0	7	0	0	7	0	0	0	7	0	0	2	5	2	0	7	2	0	7	0	2	7			
FAGUGRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0		
FRAXSP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
GALISP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GELSCAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
GERACAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
HEDEHEL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
ILEXOPA	0	0	0	0	0	3	0	0	0	0	0	0	0	0	3	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
POMPUR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	2	0	0	2	2	0	0	0	0	0	0		
JASMNLD	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LESPCUN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIGUSIN	5	0	4	4	0	2	2	2	0	5	0	5	0	6	6	0	5	0	6	7	3	6	0	4	8	2	0	0	0	0	3	0	2	5	0	0	0	0	0	
LINDEN	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIQUSTY	0	0	0	6	0	6	0	0	0	0	0	0	0	0	0	0	2	0	0	0	6	0	0	0	0	0	0	0	0	4	0	6	0	5	0	0	0	0		
LIRIMUS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
LIRITUL	2	2	0	0	2	0	0	0	2	0	2	0	2	0	0	3	3	0	3	2	0	2	0	2	0	2	2	2	0	0	0	0	0	0	0	0	4	2	0	
LONJAP	4	0	4	6	0	4	0	4	0	2	0	4	0	2	2	5	0	3	0	2	0	6	4	0	6	6	5	0	6	6	0	5	0	0	3	0	0	3		
MENTS	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
MICRYM	3	0	7	6	0	4	0	0	0	6	0	6	0	4	0	0	3	0	0	0	0	0	0	0	0	0	2	0	0	0	5	0	3	0	2	0	0	0	0	
NANDOOM	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
NYSSSLV	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
OPHVIR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
PARTOLI	2	4	2	2	4	2	4	4	3	0	0	2	2	2	0	4	0	5	2	2	3	2	3	2	2	2	2	0	0	3	3	3	2	2	2	0	0	0	0	
PHYTAME	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
PINUTAE	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
POLYPER	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	3	0	2	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0
PRUNSER	3	2	3	5	2	2	4	2	2	2	0	2	2	0	0	2	0	0	2	2	0	2	2	0	2	2	2	0	6	3	0	2	0	0	2	0	0	0	0	
QUERALB	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	0	2		
QUERCUS	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
QUERPHE	1	0	0	0	0	0	0	2	0	0	0	0	2	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	2	0	2	2	0	2	2	0	0	0	0	
QUERRUB	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
RANUSP	2	4	4	2	4	2	2	3	4	4	0	2	2	3	4	2	2	0	0	2	2	0	0	2	2	0	2	0	0	0	2	2	0	0	2	0	0	2	0	0
ROSAMUL	0	0	0	0	0	5	2	0	0	5	0	5	3	0	0	0	0	2	2	0	2	2	0	2	0	0	0	0	0	0	0	0	0	0	3	0	0	0	0	
RUBUARG	5	0	2	0	0	0	0	0	3	0	4	0	2	2	0	2	0	2	0	2	0	2	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
SMLGLA	0	3	0	0	0	4	2	0	0	2	0	0	2</																											

**Appendix E.** Mean values by site for independent and dependent variables for greenway transects.

Variable	BC3	BC5	BC8	BC9	BD1	BD3	DU3	GA1	HI2	IW2	LW1	PC1	PC2	PC3	PW1	SC3	SM2	TA1	TA2	WO1	WO4
width (m)	225	163	500	160	55	40	80	35	160	183	400	150	153	150	77.5	450	100	85	138	128	130
canopy	0.32	0.71	0.66	0.26	0.87	0.78	0.41	0.9	0.68	0.79	0.4	0.65	0.69	0.57	0.29	0.35	0.63	0.73	0.59	0.42	0.32
age	5.8	5.8	19.2	19.2	55.6	55.6	13.4	61.2	35.2	28.6	42.8	18	18	18	13.4	9.6	25.2	28.6	28.6	6	6
Canopy %	0.32	0.71	0.66	0.26	0.87	0.78	0.45	0.9	0.68	0.79	0.4	0.65	0.69	0.4	0.29	0.35	0.63	0.73	0.59	0.42	0.32
Building%	0.18	0.13	0.12	0.27	0.13	0.24	0.25	0.23	0.16	0.18	0.19	0.2	0.23	0.14	0.25	0.22	0.19	0.15	0.19	0.11	0.18
Paved%	0.34	0.11	0.21	0.28	0.16	0.16	0.25	0.15	0.17	0.17	0.21	0.21	0.18	0.25	0.35	0.21	0.18	0.18	0.25	0.23	0.19
Lawn%	0.27	0.35	0.35	0.39	0.29	0.35	0.42	0.39	0.63	0.56	0.45	0.49	0.5	0.35	0.38	0.4	0.54	0.34	0.41	0.53	0.41
Water%	0	0	0	0	0	0	0	0	0	0	0.2	0	0	0	0.75	0	0	0	0	0.1	0.1
Earth%	0	0	0	0	0	0	0	0	0	0	0.5	0	0	0	0.5	0	0	0.5	0	0.2	0.5
MICROVIM	0.3	2.17	2.5	0.4	2.93	6.5	2.42	14.6	5.6	9.6	4.62	2.4	8.78	4.67	2.5	3.8	5.38	4.72	3.52	1.83	2.58
LIGUSIH	0	0.2	0.13	0	2.82	2.17	0.78	4.1	0	1.63	9.45	0.42	17.7	5.67	0.67	0.25	6.48	0.25	0.5	0.83	0.83
LOHIJAP	0.92	0.48	1.17	1.48	1.4	3.48	2	1.63	0.13	2.32	4.52	0.17	3.22	1.18	5.35	1.8	2.95	1.82	1.48	3.98	7.55
ELAEUMB	0.42	1.72	0	0.28	3.62	2.7	0.33	1.57	0	1.62	0	0	0	0	0	0	0	0	0	0	0
ELAEPUII	0.43	1.17	0	0	1.28	0.33	0.45	1.45	0	0	0	0	0	0.92	0.5	0	0	0	0.83	0.5	0
HEDEHEL	0	0	0	0	18.2	17.5	0	26.6	0	0.33	0	0	0.33	0	0	0	0	0.5	0.17	0	0
ROSAMUL	0	0	0	0.5	7.88	1.28	1.13	5.87	1.42	0.32	6.5	0	3.42	0.18	0	0	0.22	0	0	0.25	0
ALBIJUL	0.67	0.6	0	0.48	6.1	1.13	0	1.62	0	0	0.83	0.33	0	0.42	0	0	0	0.25	0	0	0
LIGULUC	0	0	0	0	0.42	0.1	0	2.2	0	0	0.33	0	0	0	0	0	0	0.83	0	0	0
WISTSP	0	0	0	0	0.83	0.18	0	4.35	0	0	0	0	0.53	0	0	0	0	0	0	0	0
MAHOBEA	0	0	0	0	0.92	0	0	0.25	0	0	0	0	0	0	0	0	0	0	0	0	0
VIIICSP	0	0	0	0	0	0	0	2.92	0	0	0.83	0	0	0	0	0	0	0	0.33	0	0
NAHDDOM	0	0	0	0	0	0	0	0.33	0	0	0	0	0	0	0	0	0	0	0	0	0
LIGUJAP	0	0	0	0	0	0	0	0.83	0	0	0	0	0	0	0	0	0	0	0	0	0
SP RICH	5	6	3	5	12	10	6	15	3	6	7	4	6	6	4	3	4	6	6	5	3
TOTAL	2.13	6.33	3.65	2.7	47	35.4	6.82	76.5	7.15	15.5	24.8	3.17	33.7	11.8	7.97	5.85	15.3	7.92	5.33	6.65	1.22
ORHAM	2.67	5.73	3.65	2.17	12.5	15.2	5.68	32.4	5.73	15.2	18.6	2.98	29.7	11.2	7.97	5.85	14.8	6.78	5.13	6.4	1.22