THE IMPACTS OF TERRESTRIAL INVASIVE PLANTS ON STREAMS AND
NATURAL AND RESTORED RIPARIAN FORESTS IN NORTHERN NEW ENGLAND

BY

CHAD F. HAMMER

B.S., State University of New York at Plattsburgh, 2017

THESIS

Submitted to the University of New Hampshire
In Partial Fulfillment of
the Requirements for the Degree of

Master of Science
in
Natural Resources: Forestry

September, 2019
This thesis was examined and approved in partial fulfillment of the requirements for the degree of Master of Science in Forestry by:

Thesis Director, John S. Gunn, Research Assistant Professor of Forest Management

Thomas D. Lee, Associate Professor of Forest Ecology

Wilfred Wollheim, Associate Professor of Aquatic Ecosystem Ecology

On July 15th, 2019

Approval signatures are on file with the University of New Hampshire Graduate School.
DEDICATION

This thesis is dedicated to my mother, father, and sister for their endless support and encouragement. My parents have taught us well the value of hard work, determination, doing what’s right, and having courage; all of which are fundamental building blocks for success. Also, my faculty advisors and mentors over the years who have shown great confidence in me. It’s what keeps me pushing forward.
ACKNOWLEDGEMENTS

I would sincerely like to give thanks to my advisor, Dr. John Gunn, for his consistent support and guidance, although allowing me the freedom to design much of this project and grow as a researcher and student. Dr. Gunn’s time and dedication for this project has been monumental in its success. His insight has been invaluable in teaching me the complexities of ecological data and how to navigate the research process. It has been a privilege and an honor to be his first graduate student and to work on this project under his direction.

I would also like to thank my graduate committee, Dr. Thomas Lee and Dr. Wilfred Wollheim. Their guidance has made me a better ecologist and has showed me the importance of prioritizing broader impacts and making my research accessible and beneficial to society. Their expertise in data analysis and insight into the research process has been invaluable. I am also thankful for Dr. Mark Ducey’s expertise in additional statistical analysis and field methods.

Great thanks are also due to the White River Partnership (WRP) and Mary Russ, Executive Director of the WRP. Mary had a pivotal role in the investigations of Chapter II, as it was focused on assessing the active restoration work that WRP conducted. Their dedication and passion for watershed management is inspiring and their work of restoring riparian areas directly makes a big difference to the ecosystems that we all cherish.

Finally, I am thankful to my field technicians, Nathan Rees and Monica Newton, for their hard work. Funding for this research was provided by NH Agricultural Experiment Station. This work was supported by the USDA National Institute of Food and Agriculture McIntire-Stennis project #1010675 (Dr. John S. Gunn, Principal Investigator).
# TABLE OF CONTENTS

DEDICATION.................................................................................................................................................. iii

ACKNOWLEDGEMENTS........................................................................................................................................ iv

TABLE OF CONTENTS........................................................................................................................................... v

LIST OF TABLES................................................................................................................................................... vii

LIST OF FIGURES................................................................................................................................................ viii

ABSTRACT............................................................................................................................................................. x

CHAPTER I: Introduction........................................................................................................................................... 1
  Invasion Patterns of Terrestrial Invasive Plants................................................................................................. 2
  Riparian Areas....................................................................................................................................................... 5
  Thesis Organization and Research Questions....................................................................................................... 7

CHAPTER II: INCREASING THE ABILITY OF NATIVE PLANT COMMUNITIES TO RESIST INVASION AFTER A NATURAL DISTURBANCE IN RESTORED RIPARIAN FORESTS OF NORTHERN NEW ENGLAND................................................. 10
  Introduction........................................................................................................................................................ 10
  Methods............................................................................................................................................................. 14
  Results.............................................................................................................................................................. 21
  Discussion......................................................................................................................................................... 25
  Conclusion....................................................................................................................................................... 31

CHAPTER III: THE IMPACTS OF TERRESTRIAL INVASIVE PLANTS ON STREAMS AND RIPARIAN FORESTS IN NORTHERN NEW ENGLAND.......................................................... 33
  Introduction..................................................................................................................................................... 33
  Methods.......................................................................................................................................................... 40
  Results............................................................................................................................................................ 53
Discussion .................................................................................................................. 68
Conclusion .................................................................................................................. 78

CHAPTER IV: SYNTHESIS ..................................................................................... 81
Intellectual Merit and Broader Impacts ..................................................................... 81
Management Implications ......................................................................................... 83
Limitations of the Study ........................................................................................... 85
Future Research ......................................................................................................... 86

REFERENCES ........................................................................................................... 87
LIST OF TABLES

Table 1. Tree and shrub species planted by the White River Partnership after Hurricane Irene in the planted area of the riparian study sites in the White River watershed, VT.................................................................18

Table 2. Mean, standard error (SE), and Wilcoxon signed rank test p-values for the response variables in the riparian paired non-planted (NP) and planted (P) study sites in the White River watershed, VT.................................................................22

Table 3. Native tree species observed colonizing after Hurricane Irene in riparian paired non-planted (NP) and planted (P) study sites in the White River watershed, VT.................................................................24

Table 4. Invasive plant species currently prioritized in the northern section of the upper Connecticut River watershed and the northern New England region........41

Table 5. Bank stability index scores for each category to be used on Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH........50

Table 6. Size chart to classify stream substrate particle size at the Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH........52

Table 7. Patch sizes and occurrences of terrestrial invasive plant species found along Garland Brook, Lancaster, NH.................................................................54

Table 8. Land cover classifications found within 500 m radius buffers around the Invaded, Lower Watershed, and Upper Watershed plot centers along Garland Brook in Lancaster, NH.................................................................55

Table 9. Mean, standard error, and MANOVA p-values for the forest structure response variables in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..........................66

Table 10. Mean, standard error, and MANOVA p-values for the stream habitat and soil behavior response variables in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites........67
LIST OF FIGURES

Figure 1. Study area containing paired riparian study sites along the main branch of the White River and its five major tributaries in Windsor and Addison Counties, VT........................................16

Figure 2. Plot layout used on paired riparian study sites in the White River watershed, VT.................................................................17

Figure 3. Plot A and subplot B design used on paired riparian study sites in the White River watershed, VT..............................................20

Figure 4. Total (native and invasive) stems/m² in the paired riparian study sites in the White River watershed, VT.............................................22

Figure 5. Total stems/m² (by functional group) colonizing paired riparian study sites after Hurricane Irene in the White River watershed, VT........................................23

Figure 6. Map of Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) plots along Garland Brook in Lancaster, NH..........................43

Figure 7. Plot layout used on systematically placed Lower Watershed (LW), Upper Watershed (UW) plots and also Invaded (I) plots that are centered around invasive communities along Garland Brook in Lancaster, NH..........................45

Figure 8. Plot A and subplot B (streamside) and C (upland) design used on Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH..........................................................46

Figure 9. Open canopy angle measurement explanation used on Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) plots along Garland Brook in Lancaster, NH..........................................................48

Figure 10. Embeddedness and substrate sampling locations at Invaded (I) plots along Garland Brook in Lancaster, NH..................................................51

Figure 11. Map of terrestrial invasive plant occurrences found along Garland Brook in Lancaster, NH..........................................................54

Figure 12. Mean total native and invasive stems/m² at the streamside (subplot B) and upland (subplot C) in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..........................56

Figure 13. Mean total stems/m² (by size class 1-4) at the streamside (subplot B) and upland (subplot C) found in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..........................57
Figure 14. Boxplot of tree basal area m$^2$/ha in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites.................................................................58

Figure 15. Mean canopy closure measured at the stream edge and upland (center of A plot) in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites.................................................................59

Figure 16. Boxplot showing the amount of open canopy (%) measured at the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..................................................................................................................................................60

Figure 17. Mean native tree regeneration found in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..................................................................................................................................................61

Figure 18. Mean coarse woody debris (m$^3$/ha) observed along the stream and upland in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study site..................................................................................................................................................62

Figure 19. Combined mean litter and duff layers (cm) measured at the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites..................................................................................................................................................63

Figure 20. Bare ground measured at the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites.........................................................64

Figure 21. Boxplot of embeddedness (%) measured upstream and downstream of invasive plant communities within Invaded plots in the Lancaster, N.H. study sites.........65
ABSTRACT

THE IMPACTS OF TERRESTRIAL INVASIVE PLANTS ON STREAMS AND NATURAL AND RESTORED RIPARIAN FORESTS IN NORTHERN NEW ENGLAND

by

CHAD F. HAMMER

University of New Hampshire, September, 2019

Non-native invasive species are a major cause of ecosystem degradation and impairment of ecosystem service benefits in the United States. Riparian areas are at high risk for invasion because they are among the most human-disturbed ecosystems in the world. Forested riparian areas provide us with many ecosystem services and are vital to streams and rivers as they increase habitat complexity and available resources for organisms of many trophic levels. In this study, I quantified the impacts of terrestrial invasive plant invasions by Japanese knotweed and woody invasive plant species on riparian forest structure, stream physical habitat, soil structure, and soil functioning in northern New Hampshire. In addition, I assessed the effects of restoring native trees to disturbed riparian sites and their ability to resist invasive plants in central Vermont. Invaded plots had greater stems per hectare but were associated with reduced basal area of native trees. Invaded plots consistently had greater canopy closure upland from the stream (5m from the stream edge) but provided less shade at the stream channel with larger open canopy angles, therefore increasing the amount of solar radiation entering the stream. Native tree sapling densities were generally reduced in invaded sites when compared to non-invaded sites. Significantly less organic material was available at invaded sites, with less course woody debris.
(CWD) and less litter and duff on the forest floor. Invaded plots also had greater amounts of bare exposed mineral soil and higher amounts of embedded stream substratum. When comparing planted vs. non-planted riparian sites in Vermont, we found that non-planted sites had three times the amount of invasive plants and 43% greater stem density. The results of this study may assist conservation efforts of riparian forests to further understand the distribution of invasive plants and how to minimize the risk of invasion. This study will also provide insight on what ecosystem functions may be altered by invasive plants and may need to be restored.
Chapter I
INTRODUCTION

As global connectivity increases it is important to understand the ecological impacts of non-native species invasions and the mechanisms that facilitate their arrival, establishment, and spread. The colonization of non-native flora and fauna has become very common over the last century and with a warming climate the threat of invasion is projected to increase (Richardson and Rejmánek 2011). In addition, changing land-use patterns and increases in human development and modifications will facilitate the colonization of invasive plants in New Hampshire, with the potential for a northward expansion as new areas become suitable (Allen et al. 2013). Not all introduced non-native plants are considered invasive, as only a small subset is capable of overcoming biological, physical, and environmental barriers to colonize new areas. If colonization does occur and can result in large measurable impacts to the ecosystem or native species, these non-native plants are then classified as invasive (Luken 2003). Once invasive plants become established, they pose a major threat to native species diversity and ecosystem functioning in the recipient habitat. Non-native plant species are estimated to comprise 30% of New England’s flora, of which 3-5% of these are considered invasive (Mehrhoff et al. 2000; Allen et al. 2013).

Invasive plant species often outcompete native plants for resources and space through a suite of mechanisms including increased uptake of resources, differential timing of resource use, and habitat alterations to benefit the invader (Levine et al. 2003; Vila and Weiner 2004; Richardson et al. 2007; Vilà et al. 2011). Colonization by invasive plants can have many direct
and indirect impacts to ecosystem functioning, habitat physical structure, population dynamics, native species composition, and species richness (Levine et al. 2003; Vila and Weiner 2004). As invaded areas experience impacts to ecosystem functioning and changes to plant community structure, it is likely that invasions will also alter the many benefits that humans receive from natural ecosystem processes (e.g., water filtration, carbon sequestration, flood mitigation). These benefits are known as ecosystem services and since they are dependent on properly functioning ecosystems they may be hindered by disturbance, ecosystem degradation, and modifications to the landscape (Millennium Ecosystem Assessment 2005).

**Invasion Patterns of Terrestrial Invasive Plants**

Many non-native plant species were introduced to New England during the 17th through the 19th centuries when farmers and landowners would plant these species for ornamental and horticulture reasons, livestock fodder, and bank stabilization (Mehrhoff et al. 2000). Additionally, much of the northeastern United States experienced significant disturbance during this time as the landscape was heavily deforested for agricultural purposes such as cropland and pasture, which may have facilitated the spread of invasive plants (McDonald et al. 2008; Allen et al. 2013). Many studies have been conducted on land use practices and invasion dynamics such as the work by McDonald et al. (2008) and Allen et al. (2013) that assessed the historical and contemporary land cover patterns on the distribution of invasive plant species in New England. Both studies consistently found that areas with more intact forests around them were less likely to have invasive plants present and areas that were formerly plowed or used for agricultural purposes were much more likely to have invasive plants present than continuously forested areas. Johnson et al. (2006) found similar results and reported that in New England the arrival and
establishment of invasive shrubs is associated with local factors (e.g., land cover, soil characteristics, proximity to road, disturbance and human alterations) in addition to landscape scale factors (e.g., abundance of agriculture, roads, and disturbance in surrounding areas). More specifically, they found that historic agricultural land use practices (before 1974) were consistently the best predictor of woody invasive plant occurrences throughout New England and stated that agricultural land use practices can influence local soil characteristics, which may further influence invasion dynamics. A study in northeast Spain conducted by Gonzalez-Moreno et al. (2012) reported that invasions of non-native plants in forest edges were highly correlated to the distance to nearby roads. This was also the most significant predictor of increased non-native plant species richness, highest proportion of non-native species, and low overall (native and non-native) species richness. González-Moreno et al. (2012) also reported that local and adjacent landscape characteristics influence invasion dynamics, as adjacent areas may provide increased propagule pressure and experience higher disturbance regimes. In addition, they found that least disturbed habitats had the highest species richness of native plants.

Biotic and abiotic factors can play a large role in the intensity of invasion and also the magnitude of the impacts to the ecosystem. Furthermore, invasion dynamics and impacts can vary depending on the specific traits of non-native plant species when compared to the traits of the local native plants (Funk et al. 2008; Nunez-Mir et al. 2017). As these factors can vary greatly throughout a region, the implication is that some habitats are more at risk of colonization from invasive plants than others (Liendo et al. 2015). This is particularly true in the heterogeneous landscapes that are commonly found in New England.

Human settlement, development, and other natural and anthropogenic disturbance events also play a large role in invasion dynamics as they often result in a loss of intact, interior forests.
Recent work in the New England region by Allen et al. (2013) reported a significant positive relationship between the abundance of woody invasive species and the amount of forest edges on the landscape when compared to interior core forests. They stated that this may be attributed to a more favorable growth environment (i.e., increased temperature and light) and greater opportunities for dispersal along forest edges. Moreover, recent work in northern Spain by Liendo et al. (2015) reported a positive correlation between human population, industry, and the level of colonization by invasive plants. They found the highest density of non-native plants in heavily populated areas and the lowest levels of invasion in a mountainous headwater stream system with low human populations. Liendo et al. (2015) also reported a significant difference in the abundance of non-native plant species and the level of invasion with the amount of hydrological and morphological disturbance on the landscape with the highest levels of invasion occurring in areas with high anthropogenic pressure and low-quality habitats. Natural and anthropogenic disturbance and modifications not only strongly influence the risk of invasion but also the magnitude of invasion, as these disturbances often result in canopy gaps, newly available resources (e.g., light, nutrients, space) and reduced competition (Hobbs and Huenneke 1992; Parendes and Jones 2000; McDonald et al. 2008). These disturbed sites can be easily colonized by invasive species (Vila and Weiner 2004; Richardson et al. 2007). New England’s heterogeneous landscapes and vast amount of headwater stream systems, water bodies, and land use/land cover types allow for various factors to influence invasion dynamics of terrestrial invasive plants throughout many different habitat types. In these heterogeneous landscapes, it is important to understand the impacts of invasive plants on individual habitats and the factors that contribute to their spread, so that management decisions can be made to maintain ecosystem integrity and function and also minimize invasion to new areas. My research is focused on
describing the impacts of invasive plants on natural and restored forests along streams and rivers, which are referred to as riparian forests. These results will provide better understanding of how invasive plants may hinder ecosystem integrity in these ecologically important habitats and what ecosystem functions may be at risk. This study will provide insight on the distribution of invasive plants along streams in northern New England and also provide better understanding of how invasion may impact ecosystem integrity in these important habitats and what ecosystem functions may be at risk.

**Riparian Areas**

A riparian area is a terrestrial ecosystem that is adjacent to a body of water, typically a river or a stream. These habitats are of high ecological importance and provide us with many ecosystem service benefits. Ecosystem services provided from streams and their associated riparian areas include water filtration, nutrient removal, bank stabilization, erosion control, flood protection, carbon sequestration, and the provisioning of fish and wildlife habitat. Many riparian areas and streams face ongoing threats from land use practices and disturbance, which can impact cold water fisheries and other ecosystem services (Richardson and Danehy 2007; Wilkerson et al. 2010; Kanno et al. 2015). Riparian areas are considered at high risk of invasion as they are one of the most anthropogenically disturbed ecosystems in the world and are often highly modified (e.g., man-made slopes, culverts, roads, dams) with alterations for transportation, flow regulation, and drainage purposes (Allan and Flecker 1993; Liendo et al. 2015). These anthropogenic modifications often introduce invasive plant propagules into riparian areas by using contaminated landfill or heavy machinery (Vila and Weiner 2004; Richardson et al. 2007; McDonald et al. 2008). Once established, they are further spread throughout the watershed by
natural processes and disturbance such as flooding, flowing water, and hurricanes (Hood and Naiman 2000). Riparian areas are also at high risk of invasion as they are commonly found adjacent to residential development, road networks, and agricultural land use; all of which facilitate the dispersal of seeds and propagules (Hood and Naiman 2000; Vicente et al. 2013).

Riparian vegetation strongly influences the surrounding terrestrial and aquatic ecosystems by providing shade that regulates air and water temperatures also provides resources for organisms throughout many trophic levels. This is particularly important for cold headwater streams with small channel sizes. Riparian habitats are also well known for having high levels of plants species richness and diversity. Many ecosystem processes in riparian habitats are dependent on the high level of plant species richness and functional diversity found in these systems. However, changes to native species composition can hinder ecosystem processes and functioning (Pusey and Arthington 2003; Richardson et al. 2007).

Many questions remain unanswered regarding the diversity of riparian plant species and the relationship to ecosystem services such as bank stabilization and protecting water quality (Richardson et al. 2007). Riparian vegetation must be well adapted to withstand flooding, sediment deposition, physical abrasion, stem breakage, and dynamic hydrological patterns, thus passing through many ecological filters that determine which species from a regional pool may colonize (Naiman et al. 1998; Richardson et al. 2007). Woody plant species are the most common vegetation type found in riparian habitats, although herbaceous plants can dominate riparian areas when disturbance regimes and local conditions are less favorable to woody species (Richardson et al. 2007). The relationships between riparian areas, vegetation, flooding, development, forest loss, and the increasing presence of invasive species is important to understand in New England, as much area remains heavily forested and contains vast amounts of
waterbodies. This is particularly important in New Hampshire as it is one of the most forested states in the U.S., although forest fragmentation is increasing partly as a result of a growing population (Johnson et al. 2006). More specifically, Allen et al. (2013) reported that from 1992-2006, New England experienced a 5.3% reduction of forest cover, an 8% reduction of interior forests, a 1.6% increase of forest edge, and a 0.7% increase of developed areas. Unfortunately, these trend have continued as Ducey et al. (2016) reported an additional 144,000 ha of forest cover decline in New England from 2001-2011, mostly due to population growth and commercial timber harvesting. Maintaining intact riparian forests on the landscape is critical as they protect water bodies from land use practices by regulating inputs of excess nutrients and also mitigate the effects of natural and anthropogenic disturbance events (Wilkerson et al. 2006). Furthermore, healthy riparian buffers on the landscape may lower the risk of invasion and help to ensure the continued provisioning of ecosystem services.

**Thesis Organization and Research Questions**

This research fills an important knowledge gap on invasive plants in northern New England since little research has been done to investigate the interaction between invasive plants and native vegetation in riparian forest and headwater stream ecosystems. The chapters I present in this thesis are organized into two overall investigations, which are independent from one another and have been written as complete manuscripts. Some of the literature reviewed for the introductions of the following chapters is applicable for both investigations and therefore some redundancy occurs. Chapter II covers riparian forest restoration and investigates the role of planting native vegetation following a major disturbance. In this chapter, I address the research question-*does planting native species within a riparian buffer increase resistance to invasive*
woody plants following a major natural disturbance? To do this, I analyzed sites along the White River in Vermont (VT) that were planted with native trees and shrubs after Hurricane Irene caused significant bank scarification and destruction in 2011 and left these areas bare and denuded of most vegetation. These planted sites also had an adjacent non-planted area with similar habitat. This allowed me to quantify the differences between paired planted and non-planted sites, particularly the density of terrestrial, non-native invasive plants. Chapter III addresses the research questions—what are the impacts of terrestrial invasive plant communities on native riparian vegetation, riparian forest structure, soil composition, soil function, stream physical habitat quality, and ecosystem services? To do this, I conducted stream surveys for terrestrial invasive plants along Garland Brook in Lancaster NH to map the distribution of invasive plants in the Garland Brook watershed. I then conducted forest structure, vegetation, and stream habitat surveys to assess a suite of interrelated response variables and quantify the differences between invaded and non-invaded stands. Chapter IV is a synthesis of both investigations. I discuss the combined results and the relationships between these two investigations, as well as their broader impacts and management implications. In addition, I address the limitations of our data and study design and areas for future additional research.

Japanese knotweed (Fallopia japonica) is the most abundant invasive species in my study area. In New England, most knotweed plants are believed to be Japanese knotweed, although hybridization with giant knotweed (Fallopia sachalinensis) produces the hybrid bohemian knotweed (Fallopia x bohemica) (Gammon et al. 2007; Gammon and Kesseli 2010). Studies have shown that some knotweed plants in New England may be hybrids of the two parent species (F. japonica and F. sachalinensis) with some traits resembling the hybrid bohemian knotweed (Gammon et al. 2007; Gammon and Kesseli 2010). Morphological assessments used to
differentiate between *Fallopia* species in Europe and other regions of North America have proved to be unreliable in New England due to a large amount of variability caused from multiple introductions into the New England region and also extensive introgression and repeated backcrossing (Gammon et al. 2007; Colleran and Goodall 2014). It is for this reason, hereafter, I will refer to all invasive knotweed plants as *Fallopia* or knotweed to allow for the possibility that my study sites contained any hybrid of Japanese knotweed.

In both investigations, I was only interested in terrestrial non-native and invasive woody plant species. Hereafter, I will refer to the terrestrial non-native and invasive woody plant species in my study as only “invasive plant species” or “invasive plants.” In my study areas, knotweed (*Fallopia* spp.) is highly abundant and functions similarly to a woody plant species, although it is classified as an herbaceous perennial. Because of this, I also considered knotweed as a woody species and it was counted and included as such in all analysis for these two investigations.
Chapter II
INCREASING THE ABILITY OF NATIVE PLANT COMMUNITIES TO RESIST INVASION AFTER A NATURAL DISTURBANCE IN RESTORED RIPARIAN FORESTS OF NORTHERN NEW ENGLAND

Introduction

Disturbance and modifications of vegetation on the landscape not only strongly influence the risk of invasive plant colonization, but also the magnitude of the invasion. Natural and anthropogenic disturbance events often create open areas with newly available resources, (e.g., nutrients, light, and space) that are easily colonized by invasive plants (Vila and Weiner 2004; Richardson et al. 2007). Furthermore, disturbance events are often responsible for introducing invasive plant propagules to new areas through the use of contaminated landfill or heavy machinery and flowing water transporting invasive propagules (Vila and Weiner 2004; Richardson et al. 2007; McDonald et al. 2008). Once invasive plants are established, they are further spread throughout the watershed and into riparian habitats by natural disturbance and processes such as flooding, moving water, and hurricanes (Hood and Naiman 2000).

Riparian areas are some of the most disturbed ecosystems in the world and are often highly modified for transportation, flow regulation, power generation, and drainage purposes (Allan and Flecker 1993; Liendo et al. 2015). Because of this disturbance tendency, riparian areas are considered at high risk of invasion (Allan and Flecker 1993; Liendo et al. 2015). Riparian areas are often adjacent to residential development, flowing water, road networks, and agricultural land use, all of which facilitate the dispersal of seeds and propagules and also
increase invasion risk (Hood and Naiman 2000; Vicente et al. 2013). Natural disturbances in riparian areas (e.g., storms, flooding, erosion, bank collapse) can cause damage, inundation, uprooting, or death to existing plant communities and can facilitate soil movement and sediment deposition. This creates newly exposed areas with available resources that allow plant propagules to establish (Richardson et al. 2007). The complex hydrology of streams in riparian areas also influence invasion dynamics. High water levels have the potential to remove existing native plants and low water levels expose soil, both creating newly available niche space and resources for invading species (Richardson et al. 2007). Once established, invasive plant communities in riparian areas can exist in isolated patches and their propagules may be dispersed by animals (e.g., birds and mammals) and natural events through corridors such as roads, streams, and rivers (Vicente et al. 2013). In this study, I analyze the effects of restoring native riparian plant communities after a major disturbance by evaluating the success of planting native trees and shrubs in disturbed riparian areas and their ability to resist invasive plants.

**Riparian Forest Restoration and Resistance to Invasive Plants**

Restoring and maintaining intact and diverse riparian areas on the landscape is critical as they mitigate the effects of disturbance (Wilkerson et al. 2006). The soils found here regulate excess nutrient inputs to the waterbody by reducing overland flow and allowing water to infiltrate downward into the soil (Wilkerson et al. 2006; Sun et al. 2018). Typically, soils with less organic material have slower infiltration rates (USDA-NRCS 2014; Sun et al. 2018). This increases the potential for greater overland flow, which facilitates the loss of topsoil from the terrestrial habitat and also can allow sediment and excess nutrients to enter the stream. Furthermore, restoring and maintaining intact riparian areas may lower the risk of future invasion
and help to ensure proper functioning of the ecosystem (Hood and Naiman 2000; McDonald et al. 2008; Hale et al. 2018). If invasion does occur, invasive species can be very challenging to control, and management can be costly and labor intensive (NYSDEC [date unknown]). An effective strategy to combat invasive species is to prevent their establishment and spread on the landscape (Funk et al. 2008). This may be accomplished through plant interactions between native and invasive species. These interactions can strongly influence the likelihood of invasion and also the ability of native communities to resist invading species (Nunez-Mir et al. 2017). To successfully resist invasive plants and reduce the likelihood of invasion, native plant communities must have sufficient biomass and occupied niche space (Nunez-Mir et al. 2017). This concept is known as the biotic resistance hypothesis, which states that habitats or native communities with greater species richness and functional diversity have less niche space and resources available (i.e., space, light, and nutrients) to invaders, therefore increasing the community’s resistance to invasion (Funk et al. 2008; Nunez-Mir et al. 2017). Native plant communities comprised of similar functional groups or low species richness may have strong competition for resources but may be at high risk of invasion if their traits are not diverse, therefore leaving available niche space and resources available for potential invaders (Funk et al. 2008). This is important in riparian areas where frequent disturbances can reduce biomass and compromise native plant communities prior to the arrival of invasive species (Richardson et al. 2007). In these disturbed areas, native plant propagule pressure may not be adequate for recolonization of riparian communities, therefore planting or seeding native species is often required to successfully restore riparian areas (Richardson et al. 2007). Understanding the role that biotic resistance plays in a forest ecosystem and how to increase a native plant community’s
biomass and competitive ability is imperative to effectively increase their resistance to invasion (Funk et al. 2008).

**Research Questions & Objectives**

Many questions remain unanswered regarding the role that biotic resistance has in forest ecosystems since the majority of studies on biotic resistance have focused on herbaceous plants in grasslands or aquatic systems (Nunez-Mir et al. 2017). In this study, I address the research question- *does planting native trees and shrubs within a riparian buffer increase resistance to invasive woody plants following a major natural disturbance?* This research fills an important knowledge gap on invasive plants in northern New England as little has been done to assess the competitive ability of native riparian plant communities and quantify the effectiveness of active restoration following a disturbance. The goal of my study was to investigate the effects of restoring native plant communities in riparian forests and their ability to resist invasive plants. More specifically, I assessed whether planting native species within a riparian buffer can increase resistance to knotweed and other woody invasive plant species following a major natural disturbance. To do this, I analyzed sites along the White River in VT, where much work has been done to restore riparian areas and their native plant communities after Hurricane/Tropical Storm Irene caused substantial damage to the area in August of 2011. Many riparian areas were left bare from intense flooding and massive erosion. These areas with newly available resources and niche space, combined with an abundance of moving water greatly facilitated the establishment and spread of invasive species. Restoration efforts were often focused on planting native trees and shrubs in sites where Hurricane Irene denuded almost all vegetation. Some of these planted areas had an adjacent non-planted area with similar habitat.
This allowed me to quantify the differences in the abundance of invasive plants between paired planted and non-planted sites. My hypothesis is that riparian sites that were planted following a major disturbance will have lower densities of invasive plant species as more niche space will be filled and fewer resources will be available to invading species.

A better understanding of the interactions between native and invasive species and biotic manipulations that can increase resistance to invasion is important to understand in the heterogeneous landscapes of New England, as invasion dynamics and interspecific plant interaction outcomes can vary depending on the specific traits of the invading species when compared to the traits of the local native species (Funk et al. 2008; Nunez-Mir et al. 2017). Additionally, increasing the native plant communities ability to resist invasive plants is imperative as disturbance from human activity and storm events are becoming more frequent and invasion is projected to increase (Richardson and Rejmánek 2011; Allen et al. 2013).

**Methods**

**Study Site**

This research took place in the White River Watershed in central VT (Figure 1). Our study sites were located in Windsor and Addison Counties (n=5), which have a population of 23 people per square km and 18 per square km, respectively (2010 census). Both counties are mostly covered by forest and farmland and also contain parts of the Green Mountain National Forest. The White River watershed encompasses 1,839 square km, which include 20,234 ha of the Green Mountain National Forest. The main stem of the White River is 90 km and has five major tributaries: The first branch, the second branch, the third branch, the west branch, and the
Tweed River\textsuperscript{1}. The White River Partnership (WRP) has done much work in central Vermont to restore riparian areas and plant native riparian plant communities after Hurricane Irene delivered 10-20 cm of rainfall and caused substantial damage and severe flooding to the area in August of 2011. Hurricane Irene is considered Vermont’s second most significant natural disaster in the 20\textsuperscript{th} and 21\textsuperscript{st} century as it was the most damaging to infrastructure and caused several human fatalities (NOAA 2012). Hurricane Irene left many areas bare from intense flooding, massive erosion, and uprooted many riparian plant communities. Furthermore, after the water receded, in some areas up to 1 m of sand was left deposited and removed with heavy machinery (Mary Russ, personal communication, June 1, 2018). These bare areas with reduced competition and ample niche space were heavily invaded by invasive species and the abundance of moving water greatly facilitated their spread and colonization throughout the White River watershed. The WRP has done extensive work planting over 60,000 native trees and shrubs along the White River and surrounding tributaries on over one hundred sites, some of which served as study sites for this research. The vegetation surveys described below confirmed that knotweed is by far the most abundant invasive plant species found in our paired riparian study sites. Knotweed was quickly able to spread and colonize many riparian areas after Hurricane Irene, as knotweed reproduced primarily by vegetative propagation and its propagules are mostly dispersed through moving water. Knotweed also has a very aggressive rhizome structure and a fast-growing dense canopy that allowed this species to quickly outcompete native vegetation after Hurricane Irene and establish large monospecific stands throughout central Vermont.

\footnote{\url{http://whiteriverpartnership.org/white-river-watershed-2/}}
Study Design

To analyze the success of restoring native plant communities where invasive plants are present, I conducted vegetation surveys at previously planted areas (n=5) by the WRP that were restored post Hurricane Irene in 2011-2012. I then gained access to a directly adjacent site with similar habitat characteristic to serve as the non-planted area. Together, these planted and non-planted areas served as my paired riparian study sites (n=5). The WRP generally plants 988 stems/ha of early successional and fast-growing native species in an attempt to successfully
compete with fast growing invasive species (Table 1). Vegetation surveys were conducted on these planted and adjacent non-planted sites to allow paired comparisons (Fig. 2).

**Figure 2.** Plot layout used on paired (planted and non-planted) riparian study sites (n=5) in the White River watershed, VT. Plot width was fixed at 10 m, but plot length varied with the distance between the stream bank and the extent of the restored site (plot A). A 2 m wide transect (subplot B) was established through the center of plot A. Light circles indicate planted vegetation with slightly larger planted trees at end of riparian buffer. Dark circles indicate naturally recruited vegetation throughout entire riparian buffer.
Table 1. Tree and shrub species (by proportion) planted by the White River Partnership (WRP) after Hurricane Irene (2011/2012) in the planted area of the riparian study sites (n=5) in the White River watershed, VT.

<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific name</th>
<th>Bagley</th>
<th>Clifford Park</th>
<th>Floyd</th>
<th>Mill Brook</th>
<th>Peavine Park</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Trees</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>American sycamore</td>
<td>Platanus occidentalis</td>
<td>-</td>
<td>0.03</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Balsam poplar</td>
<td>Populus balsamifera</td>
<td>-</td>
<td>0.01</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Black cherry</td>
<td>Prunus serotina</td>
<td>-</td>
<td>0.04</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Box elder</td>
<td>Acer negundo</td>
<td>0.17</td>
<td>0.10</td>
<td>0.07</td>
<td>0.08</td>
<td>0.13</td>
</tr>
<tr>
<td>Eastern cottonwood</td>
<td>Populus deltoides</td>
<td>-</td>
<td>0.10</td>
<td>0.07</td>
<td>0.05</td>
<td>0.16</td>
</tr>
<tr>
<td>Gray birch</td>
<td>Betula populifolia</td>
<td>0.03</td>
<td>0.04</td>
<td>-</td>
<td>0.08</td>
<td>-</td>
</tr>
<tr>
<td>Quaking aspen</td>
<td>Populus tremuloides</td>
<td>-</td>
<td>0.08</td>
<td>-</td>
<td>0.05</td>
<td>-</td>
</tr>
<tr>
<td>Red maple</td>
<td>Acer rubrum</td>
<td>0.20</td>
<td>0.09</td>
<td>0.27</td>
<td>0.08</td>
<td>0.16</td>
</tr>
<tr>
<td>Speckled alder</td>
<td>Alnus incana</td>
<td>-</td>
<td>0.04</td>
<td>0.07</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Tamarack</td>
<td>Larix laricina</td>
<td>-</td>
<td>0.03</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Willow spp.</td>
<td>Salix spp</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>0.36</td>
<td>-</td>
</tr>
<tr>
<td>White ash</td>
<td>Fraxinus americana</td>
<td>0.03</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Yellow birch</td>
<td>Betula alleghaniensis</td>
<td>0.17</td>
<td>0.06</td>
<td>-</td>
<td>0.01</td>
<td>-</td>
</tr>
<tr>
<td><strong>Shrubs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>American witch-hazel</td>
<td>Hamamelis virginiana</td>
<td>-</td>
<td>0.04</td>
<td>0.07</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Elderberry</td>
<td>Sambucus nigra</td>
<td>-</td>
<td>0.04</td>
<td>-</td>
<td>0.04</td>
<td>-</td>
</tr>
<tr>
<td>Highbush cranberry</td>
<td>Viburnum opulus</td>
<td>0.07</td>
<td>0.04</td>
<td>0.03</td>
<td>0.03</td>
<td>-</td>
</tr>
<tr>
<td>Nannyberry</td>
<td>Viburnum dentago</td>
<td>0.07</td>
<td>-</td>
<td>-</td>
<td>0.03</td>
<td>-</td>
</tr>
<tr>
<td>Red osier dogwood</td>
<td>Cornus sericea</td>
<td>0.10</td>
<td>0.07</td>
<td>0.17</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Serviceberry</td>
<td>Amelanchier spp</td>
<td>0.07</td>
<td>0.06</td>
<td>0.07</td>
<td>0.10</td>
<td>0.13</td>
</tr>
<tr>
<td>Silky dogwood</td>
<td>Cornus amomum</td>
<td>0.10</td>
<td>0.16</td>
<td>0.20</td>
<td>0.11</td>
<td>0.44</td>
</tr>
</tbody>
</table>
Plot Layout & Design

Ten-meter wide plots (plot A) were established and spaced 15 m apart along the adjacent stream or river and extended through the riparian zone (perpendicular to the stream) to the back of the restoration site of variable length (Fig. 2). I established as many plots (plot A) as the length of the riparian buffer allowed for a particular site with a target of 40% sampled area of the entire site. A 2 m wide transect (subplot B) was established through the center of plot A to achieve a 20% sampled area of plot A. Subplot B extended perpendicular to the adjacent stream or river through the riparian zone to the back edge of plot A (Fig. 2).

Vegetation Surveys

Within plot A, tree species and DBH (cm) was collected for all trees > 10 cm DBH. Height (m) was also measured on one dominant and one co-dominant tree using a Haglof Vertex IV and Transponder III (Haglöf Sweden AB, Långsele, Sweden). Visual vegetation estimates were completed to characterize vegetation structure. Visual estimates of percent cover were made within plot A to characterize: overstory canopy (> 5 m in height); understory (0.5 to 5 m in height); and herbaceous and non-herbaceous ground cover (< 0.5 m in height) (Kaufman and Robison 1998). Ground cover was further characterized into 11 categories by estimating percent cover of the following: (1) hay-scented fern (*Dennstaeditia punctilobula*); (2) various other fern; (3) horsetail; (4) leaf litter; (5) moss; (6) club moss (*Lycopodium*); (7) various herbaceous; (8) bare ground; (9) grasses (*Poaceae*), sedges (*Cyperaceae*), and rushes (*Juncaceae*); (10) invasive species; (11) and knotweed (live and dead). At all plot locations GPS coordinates, bank facing direction, any signs of disturbance or modifications, and dominant land use/land cover was documented. Within subplot B (nested 2 m transect), species and stem density of all woody
vegetation were collected and binned into height classes 1-4: 1 (0-1 m), 2 (1.1-2 m), 3 (2.1-5 m), or 4 (5 m and up). All native stems were documented as a non-planted stem or being a potentially planted stem based on appropriate spacing, species, and size. Invasive species percent cover was also estimated, as well as their distance to the river.

**Soil Composition and Function**

To analyze soil composition and functioning differences between planted and non-planted sites, the leaf litter layer and duff layer (i.e., decomposed organic soil layer above the mineral A horizon) depth was measured to the nearest 0.1 cm. Shallow soil profiles were dug on the forest floor with a small trowel 2 m from the front edge and 2 m from the back edge of subplot B, where litter and duff depths were measured and soil was characterized (Fig. 3). Infiltration rates of water into soil (cm/min) were also measured at these two locations in subplot B to assess overland flow potential by using a 15.24 cm diameter cylinder (single-ring infiltrometer made from plastic PVC) to perform an infiltration test (USDA 2014). Any presence of erosion at the bank was also documented.

![Figure 3. Plot A and subplot B design used on paired (planted and non-planted) riparian study sites (n=5) in the White River watershed, VT. Symbols indicate where soil was analyzed.](image-url)
**Statistical Analysis**

I used the statistical program JMP Pro 13 (SAS, Cary, NC USA) to perform a Wilcoxon Signed Rank Test to examine the differences between paired planted and non-planted riparian study sites for abundance of native and invasive vegetation (stems/m$^2$), regeneration (stems/m$^2$), and soil characteristics. A non-parametric test was needed because sample size was small (n=5) and the data did not meet the assumptions of normality, even after log and arcsine square root transformations were completed.

**Results**

**Plant Community and Soil Composition**

The density (stems/m$^2$) of invasive stems was substantially higher ($p=0.06$) in non-planted sites ($\bar{x}=4.1$) compared to planted sites ($\bar{x}=1.3$) (Table 2). All paired study sites had less invasive species present in the planted area when compared to its non-planted counterpart (Fig. 4). The invasive plant species composition in the paired riparian study sites (planted and non-planted) was comprised mostly of knotweed (92% and 95% of all stems, respectively) and the remainder was Morrow’s honeysuckle (*Lonicera morrowii*). There were no significant differences in the density of native stems between planted and non-planted riparian sites (Table 2). Total stem density was slightly higher in non-planted sites when compared to planted sites, although not statistically significant at $\alpha = 0.05$ level ($p=0.31$) ($\bar{x}=16.3$, 11.4, respectively). Other measured response variables such as native tree regeneration, soil properties and soil function showed no statistically significant differences or trends in the paired riparian study sites (Table 2).
Table 2. Mean, standard error (SE), and Wilcoxon signed rank test $p$-values for the response variables in the riparian paired non-planted (NP) and planted (P) study sites in the White River watershed, VT (n=5). Bold $p$-value indicates significance at $\alpha = 0.05$ level.

<table>
<thead>
<tr>
<th>Response Variable</th>
<th>Non-planted Mean (SE)</th>
<th>Planted Mean (SE)</th>
<th>$p$-value NP-P</th>
</tr>
</thead>
<tbody>
<tr>
<td>knotweed stems/m$^2$</td>
<td>3.9 (1.4)</td>
<td>1.2 (0.6)</td>
<td>0.06</td>
</tr>
<tr>
<td>Invasive stems/m$^2$</td>
<td>4.1 (1.5)</td>
<td>1.3 (0.6)</td>
<td>0.06</td>
</tr>
<tr>
<td>Native stems/m$^2$</td>
<td>12.2 (4.2)</td>
<td>10.1 (1.7)</td>
<td>1.00</td>
</tr>
<tr>
<td>Total stems/m$^2$</td>
<td>16.3 (3)</td>
<td>11.4 (1.6)</td>
<td>0.31</td>
</tr>
<tr>
<td>Native tree regeneration (stems/ha)</td>
<td>5030 (3235)</td>
<td>5546 (2369)</td>
<td>1.00</td>
</tr>
<tr>
<td>Upland litter &amp; duff depth (cm)</td>
<td>1.6 (0.5)</td>
<td>1.4 (0.4)</td>
<td>0.63</td>
</tr>
<tr>
<td>Bank litter &amp; duff depth (cm)</td>
<td>0.6 (0.3)</td>
<td>0.9 (0.3)</td>
<td>0.31</td>
</tr>
<tr>
<td>Upland infiltration rate (cm/min)</td>
<td>6.3 (2.6)</td>
<td>8.5 (2.9)</td>
<td>0.06</td>
</tr>
<tr>
<td>Bank infiltration rate (cm/min)</td>
<td>7 (3.3)</td>
<td>4.2 (1.3)</td>
<td>0.63</td>
</tr>
</tbody>
</table>

Overall, invasive plants comprised 23% of the stems, with the highest proportion observed at Bagley (63%), followed by Peavine Park (23%), Mill Brook (18%), Clifford Park (10%), and Floyd (1%). Mill Brook had the largest difference in abundance of invasive plants between non-planted and planted riparian sites, followed by Bagley and Peavine Park (Fig. 4). Planted sites had an average of 61% fewer invasive stems compared to non-planted sites.

Figure 4. Total (native and invasive) stems/m$^2$ in the paired planted (P) and non-planted (NP) riparian study sites (n=5) in the White River watershed, VT.
Natural regeneration

Although there was no difference observed in natural regeneration between planted and non-planted areas \((p=1.00)\), small shrubs and low-lying woody plants such as *Rubus* spp., *Ribes* spp., *Spiraea* spp., Virginia creeper (*Parthenocissus quinquefolia*), and highbush cranberry (*Viburnum opulus*) were colonizing the paired study sites most frequently after Hurricane Irene, followed by trees and tall shrubs (Fig. 5). With the exception of site Floyd, all planted areas had greater tree species richness when compared to its non-planted counterpart (Table 3). Boxelder (*Acer negundo*), eastern cottonwood (*Populus deltoides*), red maple (*Acer rubrum*), and willow (*Salix* spp.) were the most common and widespread tree species observed across all sites (Table 3).

**Figure 5.** Total stems/m² (by functional group) colonizing paired (planted and non-planted) riparian study sites \((n=5)\) after Hurricane Irene in the White River watershed, VT.
Table 3. Native tree species observed (stems/ha) colonizing after Hurricane Irene (2011) in riparian paired non-planted (NP) and planted (P) study sites in the White River watershed, VT (n=5). Stems/ha values include seedlings, saplings, and established mature trees of all size classes.

<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific Name</th>
<th>Bagley NP</th>
<th>Bagley P</th>
<th>Clifford Park NP</th>
<th>Clifford Park P</th>
<th>Floyd NP</th>
<th>Floyd P</th>
<th>Mill Brook NP</th>
<th>Mill Brook P</th>
<th>Peavine Park NP</th>
<th>Peavine Park P</th>
</tr>
</thead>
<tbody>
<tr>
<td>American basswood</td>
<td><em>Tilia americana</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,042</td>
<td>-</td>
<td>1,064</td>
<td>888</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>American elm</td>
<td><em>Ulmus americana</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>600</td>
<td>-</td>
<td>-</td>
<td>532</td>
<td>-</td>
<td>178</td>
</tr>
<tr>
<td>American sycamore</td>
<td><em>Platanus occidentalis</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>160</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Balsam fir</td>
<td><em>Abies balsamea</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>625</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Balsam poplar</td>
<td><em>Populus balsamifera</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>9,375</td>
<td>7,176</td>
<td>177</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Black birch</td>
<td><em>Betula leucodora</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>22,083</td>
<td>-</td>
<td>709</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Black locust</td>
<td><em>Robinia pseudocacia</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>200</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>323</td>
<td>-</td>
</tr>
<tr>
<td>Boxelder</td>
<td><em>Acer negundo</em></td>
<td>983</td>
<td>93</td>
<td>2,723</td>
<td>3,603</td>
<td>13,125</td>
<td>926</td>
<td>6,560</td>
<td>2,367</td>
<td>19,840</td>
<td>14,113</td>
</tr>
<tr>
<td>Butternut</td>
<td><em>Juglans cinerea</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>148</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Crabapple</td>
<td><em>Malus spp.</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,426</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Eastern cottonwood</td>
<td><em>Populus deltoides</em></td>
<td>-</td>
<td>-</td>
<td>44,554</td>
<td>7,606</td>
<td>35,417</td>
<td>30,556</td>
<td>-</td>
<td>592</td>
<td>2,313</td>
<td>3,226</td>
</tr>
<tr>
<td>Eastern hemlock</td>
<td><em>Tsuga canadensis</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>417</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Eastern hop hornbeam</td>
<td><em>Ostrya virginiana</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>3,241</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Eastern white pine</td>
<td><em>Pinus strobus</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>833</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Gray birch</td>
<td><em>Betula populifolia</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>155</td>
<td>80</td>
<td>-</td>
<td>148</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Green ash</td>
<td><em>Fraxinus pennsylvanica</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>161</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Hawthorn</td>
<td><em>Crataegus spp.</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>45</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Northern white cedar</td>
<td><em>Thuja occidentalis</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>10,417</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Quaking aspen</td>
<td><em>Populus tremuloides</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>80</td>
<td>833</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Red maple</td>
<td><em>Acer rubrum</em></td>
<td>1,251</td>
<td>217</td>
<td>-</td>
<td>120</td>
<td>5,208</td>
<td>694</td>
<td>532</td>
<td>148</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Red oak</td>
<td><em>Quercus rubra</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>40</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>3,306</td>
<td>-</td>
</tr>
<tr>
<td>Speckled alder</td>
<td><em>Alnus incana</em></td>
<td>402</td>
<td>-</td>
<td>-</td>
<td>240</td>
<td>833</td>
<td>1,852</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sugar maple</td>
<td><em>Acer saccharum</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>417</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>1,532</td>
</tr>
<tr>
<td>Tamarack</td>
<td><em>Larix laricina</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>40</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>White ash</td>
<td><em>Fraxinus americana</em></td>
<td>-</td>
<td>899</td>
<td>-</td>
<td>320</td>
<td>-</td>
<td>4,142</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>White spruce</td>
<td><em>Picea glauca</em></td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>417</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Willow</td>
<td><em>Salix spp.</em></td>
<td>2,636</td>
<td>248</td>
<td>4,950</td>
<td>10,889</td>
<td>28,333</td>
<td>44,907</td>
<td>-</td>
<td>444</td>
<td>23,132</td>
<td>14,194</td>
</tr>
<tr>
<td>Yellow birch</td>
<td><em>Betula alleghaniensis</em></td>
<td>-</td>
<td>93</td>
<td>-</td>
<td>320</td>
<td>117,708</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>
Discussion

Plant Community and Soil Composition

This study revealed a substantial difference and a clear trend when comparing the mean stem density of invasive plants between planted and non-planted sites. All paired sites had a lower stem density of invasive plants in the planted area when compared to its non-planted counterpart (Fig. 4). Although not statistically significant at an $\alpha=0.05$ level, this difference may be explained by the reduction of resources and space available to the invaders (i.e., niche availability) as native communities begin to establish. Conversely, non-planted sites may have greater niche space available to invading species, with the exception of the Floyd site that received more native propagule pressure in the non-planted area (discussed below) (Fig. 5).

These findings are consistent with the biotic resistance hypothesis and other community ecology theories that state native plant communities with sufficient biomass and higher species richness can successfully lower the probability of invasion when compared to communities with low species richness (Funk et al. 2008; Nunez-Mir et al. 2017). Recent work by Maron and Marler (2007) reported that diverse native plant communities were able to increase resistance to the invasive herbaceous perennial spotted knapweed (*Centaurea maculosa*) by decreasing nitrogen, light, and soil moisture available to the invader and stated that diverse communities can lower the probability of invasion over communities with low species richness.

In these sites, the riparian forests were mostly in the stand initiation stage and low biomass may be the limiting factor to fully increase resistance to invasion in these young native communities. If planted species and native communities are allowed more time to establish before receiving high propagule pressure from knotweed, statistical analysis may have revealed more significant results. The White River Watershed in Central Vermont is highly invaded by
knotweed and propagules are consistently being dispersed to new areas through moving water after high-flow and rain events. In these newly developing stands, some large legacy trees survived the hurricane and were dispersed equally in the paired study sites. These residual trees with mature canopies, in conjunction with a diverse array of planted fast-growing trees and shrubs may have been just enough biomass, cover, and competition to reduce the vigor of invasive knotweed. Furthermore, over time as these young trees and shrubs develop mature canopies, they may further slow the growth of invaders, particularly shade intolerant species.

The WRP has learned through trial and error that it is critical to use early successional species such as willow (*Salix* spp.), poplar (*Populus* spp.), red maple (*Acer rubrum*), birch (*Betula* spp.), and cottonwood (*Populus* spp.) when restoring and planting riparian habitats. These fast-growing species have the best chance at increasing the competitive ability of establishing native plant communities and slowing the growth of highly competitive invasive species. In addition to the planted species being functionally diverse and fast growing, a trait-based community framework can also be applied. To do this, the traits and resource use of the planted species should be similar to the species that is predicted to invade, therefore limiting the resources to the invaders and increasing resistance to invasion (Funk et al. 2008). Nevertheless, these diverse and competitive planted species should be well adapted to pass through the many ecological filters that riparian vegetation has to overcome before establishing. In riparian habitats, these typically include a high disturbance regime (i.e., high frequency and intensity), flooding, and thriving in an often-mesic environment.

Statistical analysis did not detect any significant differences in native tree regeneration (1.1-2 m native tree saplings) or total stem density of native vegetation between planted and non-planted sites (Table 2). Native stem density was similar across all paired planted and non-planted
sites, except at the Floyd site (Fig. 4). At the Floyd site, the non-planted area had much higher stem density of native species when compared to the planted area. This is most likely because the non-planted area was directly adjacent to a mature forest, receiving much higher seed rain and native species propagule pressure than the planted area that was adjacent to a field. This is the only paired site that had distinct differences in vegetation in the adjacent habitat. The similarity of native tree regeneration and total density of native vegetation between planted and non-planted sites is best explained by these stands still being early in successional development as they were denuded of topsoils and almost all vegetation after Hurricane Irene until plantings took place in 2012. In these sites, the highly competitive invasive species that established here most likely have not had enough time to fully establish and displace native species. Therefore, these stands have yet to experience all of the potential direct and indirect impacts to habitat physical structure, local species richness, and native species composition caused from invasive plant colonization (Frappier et al. 2003; Levine et al. 2003; Vila and Weiner 2004; Gerber et al. 2008; Lavoie 2017). Studies consistently show that when a habitat is invaded, the strong competitive ability of non-native invaders can lead to significant reduction of biomass, growth, reproduction, and resource allocation of native plant communities (Levine et al. 2003; Vila and Weiner 2004). Furthermore, a meta-analysis by Vilà et al. (2011) reported that the colonization of invasive plant species significantly impacted native plants by reducing fitness (41.7%) and growth (22.1%), although increased the total community plant production (56.8%). They also reported that plant community structure was negatively impacted by reducing native species abundance and diversity by 43.5% and 50.7%, respectively. This implies that over larger time scales, the negative impacts of these highly productive and competitive species can be seen at the species, community, and even ecosystem levels.
The negative effects to habitat physical structure, ecosystem functioning, and natural processes caused from invasion (Levine et al. 2003; Vila and Weiner 2004) may also reduce an ecosystem’s ability to provide the many benefits that humans receive from natural processes (e.g., water filtration, carbon sequestration, flood mitigation). These are known as ecosystem service benefits and are dependent on properly functioning ecosystems, which can be hindered by disturbance, modifications, ecosystem degradation and changes to plant communities (Millennium Ecosystem Assessment 2005). In the early successional stands in my study, niche space and resources may still be available for native and invasive species to co-exist, with native plant communities yet to experience the deleterious outcomes of knotweed invasions. Lavoie (2017) recently conducted a literature review on the effects of Japanese knotweed and reported that all studies showed negative impacts to biomass, cover, and species richness of native plant communities in invaded areas, with up to 10 times fewer species in areas where Japanese knotweed had established. Furthermore, in a study on the responses to stream ecosystems from riparian invasions of Japanese knotweed, Lecerf et al. (2007) reported that Japanese knotweed can negatively impact ecosystem processes and regeneration, mainly by reducing light with its dense canopy and also providing large amounts of lower quality leaf litter when compared to native species.

**Natural Regeneration**

Native low woody plant forms were naturally establishing the quickest at the paired riparian study sites (Fig. 5). This is common in early successional stages of northeastern riparian forests but was also different at the Floyd site. At this site, the adjacent forest’s propagule pressure again explains why trees were the most abundant growth form that quickly regenerated
naturally in these stands. The adjacent mature forest supplying seed rain and native species propagules in higher densities to the non-planted area at site Floyd also explain why tree species richness was higher in the non-planted area when compared to the planted area at this site (Table 3). When quantifying the stem density of regenerating tree species, all stems were counted from identifiable seedlings to stems > 5 m. In sites where large amounts of small seedlings were present, this resulted in high values, although many will not survive past the seedling developmental stage (Table 3). This was often the case with large amounts of yellow birch (*Betula alleghaniensis*) and boxelder (*Acer negundo*) seedlings present at the study sites (Table 3).

**The Role of Disturbance**

Immature riparian forests were mostly in the stand initiation stage following the stand-replacing disturbance of Hurricane Irene. This best explains why soil composition and soil functioning response variables showed no significant differences in mean values (Table 2). These study sites currently have poorly developed sandy soils and most likely had their uppermost soil layers washed away from Hurricane Irene and heavy annual flooding thereafter. Most of these riparian sites had large amounts of silt and sand left deposited after the flood waters receded and in some cases were further disturbed by heavy machinery needed to remove the sand and/or restore the stream banks. Along with natural disturbance and flowing water, heavy machinery also increases the risk of invasion by further disturbing and scarifying the soil. Furthermore, heavy machinery is often responsible for introducing outside seed sources to the area, as propagules may be transported on the machinery itself or in the soil that is used to reconstruct the habitat (McDonald and Urban 2006; McDonald et al. 2008). Like the young
vegetation communities in these stands, the soils here have not had enough time to mature and long-term effects are most likely not yet realized. This was most apparent in the Floyd and Mill Brook sites, both of which experience ongoing high natural disturbance regime due to ice scouring of the banks and frequent flooding. This further prevents soils from accumulating litter and duff and forming upper horizons, particularly an organic soil layer. The soils at the Mill Brook site were comprised mostly of pure sand and soils at the Floyd site were comprised of sand, gravel, and a small amount of silt. Establishing a healthy soil layer is imperative in riparian habitats, as soils play a large role in regulating the transfer of nutrients and sediment into waterways (Likens et al. 1970). Having organic material present in riparian soils can be an important factor in reducing overland flow by increasing infiltration rates (USDA-NRCS 2014; Sun et al. 2018). Soil organic matter increases infiltration rates by creating stable aggregates and suitable habitat for soil biota such as earthworms, both of which increase porosity and improve soil structure (USDA-NRCS 2014; USDA 2014).

The Mill Brook site had the greatest difference in density of invasive stems between planted and non-planted sites (Fig. 4). This site was located at the confluence of Mill Brook and the White River. The reason for this large difference is not clear but may be explained by the frequent ice scouring of the banks and intense flooding this section of the main branch of the White River often experiences. Planted trees at the Mill Brook site were struggling to establish with the frequent high intensity disturbance events, and continually getting knocked down by large sections of moving ice during the annual thawing of the river and ice scouring of the banks. Coincidentally, the section planted with native trees (though many were knocked down but still growing) was completely dominated by native ostrich fern (*Matteuccia struthiopteris*) and the adjacent non-planted section was completely taken over by invasive knotweed. Both of these
plants are perennials that are well adapted to flooding and can quickly spread through rhizomes. Furthermore, the planted section was ~1 m lower in elevation and therefore may be inundated periodically and experience more frequent disturbance from ice scouring and flooding.

**Conclusion**

Natural disturbance in riparian habitats can play a large role in creating available niche space for invaders to establish and further facilitate invasion by transporting plant propagules to these new sites, thus allowing many invasive species to overcome dispersal limitations. Our results illustrate that planting can make a difference in reducing the abundance of invaders following a major disturbance, but soil properties may be slower to respond as they are often dependent on vegetation, canopy, and stand development stages. Priorities for management and areas where plantings should take place are areas where seed rain and native propagule pressure may be insufficient to restore native communities naturally. Conversely, disturbed areas adjacent to mature forests may be able to naturally recruit native communities more successfully and would be less of a priority for management. Successful invaders are often functionally different than existing species, so communities that are functionally diverse with saturated niche space comprised of native species with similar traits to the expected invaders may have greater resistance to invasion.

This study provides important short-term baseline data for future studies investigating longer-term responses to active restoration and the success of increasing resistance to invasion and natural succession processes involving invasive species. Furthermore, monitoring is critical to evaluate the success of riparian restoration efforts and how well ecosystem functioning responds to restoration. This is particularly important to soil development and soil functioning as riparian soils are responsible for many of the ecosystem services that riparian areas provide, such
as nutrient regulation. Regulating excess nutrients has large implications to water quality, eutrophication, and aquatic organisms, which is particularly important in the agricultural landscapes that are often found in the state of Vermont. Over longer time scales, as native communities develop significant biomass and canopy cover, their competitive ability would most likely increase. Non-planted sites with higher densities of invasive plants may experience more harmful impacts to biodiversity, ecosystem functioning, and ecosystem services in the future.

**Management**

Since disturbance events are expected to increase and often lead to ecosystem degradation, active habitat restoration of native plant communities may be an essential tool in maintaining ecological integrity, biodiversity, and ecosystem functioning. To effectively increase resistance in native plant communities it may be most beneficial to evaluate their species diversity, functional diversity, and resource use. Additionally, the traits of potential invading species in riparian communities should be assessed to fully increase resistance to invasion. Even if high resistance is occurring, invasion may be inevitable. It is crucial to restore riparian habitats and plant native species in areas with low seed rain or native propagule pressure immediately after a disturbance to restore biomass and give the native species a head start. After a disturbance event or the removal of an invasive plant community, both of which may create openings, available resources, or niche space, it is important to take restoration efforts to prevent new invasive species from colonizing or re-invading an area (Clements et al. 2016). If invasion or re-invasion does occur, it may take a suite of management techniques to successfully combat invasive plants. With aggressive knotweed, this may take a repetitive rotation of removing aboveground biomass, herbicide treatment, planting native species, and monitoring.
Chapter III

THE IMPACTS OF TERRESTRIAL INVASIVE PLANTS ON STREAMS AND RIPARIAN FORESTS IN NORTHERN NEW ENGLAND

Introduction

As increasing global connectivity continues to facilitate the homogenization of Earth’s biota, it is important to understand the long-term impacts of non-native species invasions on ecosystem functioning. As invaded areas experience changes to ecosystem functioning and plant community structure is altered, it is likely that invasions will also impact the many benefits that humans receive from natural ecosystem processes (e.g., water filtration, carbon sequestration, flood mitigation). These are known as ecosystem service benefits and typically depend on properly functioning ecosystems, which can be hindered by disturbance, ecosystem degradation, and modifications to the landscape (Millennium Ecosystem Assessment 2005). Not all introduced non-native plants are considered invasive, as only a small subset is capable of overcoming biological, physical, and environmental barriers to colonize new areas. If colonization does occur and can result in large measurable impacts to the ecosystem or native species, these non-native plants are classified as invasive (Luken 2003). Once invasive plants become established, they often pose a major threat to native plant species diversity and ecosystem functioning in the recipient habitat. Invasive plant species often outcompete native plants for resources and space through a suite of mechanisms that include more efficient uptake and use of resources and habitat alterations to benefit the invader (Levine et al. 2003; Vila and Weiner 2004; Richardson et al. 2007; Vilà et al. 2011). These modified, novel habitats allow aggressive invaders to form dense monospecific stands that often outcompete native species
early in the growing season (Vila and Weiner 2004; Richardson et al. 2007). In addition, Levine et al. (2003) reported that changes to the ecosystem were often caused by changes to species composition, plant phenology, root structures, transpiration rates, and differences in functional traits between the invading species and the native species being displaced. These changes can have many impacts to species richness, native plant population dynamics, and forest structure (Levine et al. 2003). In this study, my objective was to describe and quantify the impacts of terrestrial invasive plants on ecosystem characteristics of riparian forests (i.e., forests along streams and rivers) and stream habitat in northern New Hampshire.

Many studies show that when a habitat is invaded by non-native plants, the strong competitive ability of these invaders can greatly reduce growth, reproduction, and resource allocation of native plant communities. In a meta-analysis of the ecological impacts from invasive plants, Vilà et al. (2011) reported that their colonization often leads to negative impacts to native plant communities by significantly reducing their fitness and growth, although consistently increasing the total community plant production. This implies that invasive plants are typically highly competitive and productive species. Furthermore, in a meta-analysis on the competitive ability of invasive plants, Vila and Weiner (2004) reported that the presence of invasive plants reduced native plant biomass by an average of 46.6%. Vilà et al. (2011) also reported that plant community structure was negatively impacted by reducing native species abundance and diversity by 43.5% and 50.7%, respectively, and stated that negative impacts can be seen at the species, community, and ecosystem levels. In a study on the responses to stream ecosystems from riparian invasions, Lecerf et al. (2007) reported that invasive plants can negatively impact ecosystem processes, mainly by reducing light with their dense canopy and also providing large amounts of lower quality leaf litter when compared to native riparian species.
Colonization by invasive plant communities can also have many direct and indirect impacts to ecosystem functioning and can affect habitat physical structure, population dynamics, native species composition, and species richness (Levine et al. 2003; Vila and Weiner 2004; Pimentel et al. 2005).

**Riparian Forest Vegetation and Structure**

Riparian forests are of high ecological importance and provide many ecosystem services. The vegetation found in the riparian zone strongly influences the surrounding terrestrial and aquatic ecosystems (Hood and Naiman 2000; Pusey and Arthington 2003; Richardson et al. 2007). Many ecosystem processes in riparian habitats are dependent on the high level of plant species richness and functional diversity found in these systems, however, changes to native species composition can hinder ecosystem processes and functioning (Richardson et al. 2007). The structural characteristics of riparian forest vegetation are what allow these habitats to mitigate the effects of disturbance and land use practices by stabilizing stream banks, reducing sedimentation, and limiting runoff into adjacent water bodies.

Forest structure refers to the distribution of horizontal and vertical layers in a forest, specifically the composition of trees, shrubs, ground cover, and coarse woody debris (CWD) (Good Forestry of the Granite State 2010). Maintaining intact riparian forest structure and native plant communities with sufficient canopy cover is critical to preserve aquatic and terrestrial ecosystem processes and integrity. This is particularly important for cold headwater streams with small channel sizes. Intact riparian forests provide sufficient canopy cover to shade streams, which reduces solar radiation, moderates temperature fluctuations, and maintains cool water temperatures and dissolved oxygen levels (Richardson and Danehy 2007). These low-light
habitats also maintain cool air temperatures and humidity of the terrestrial environment, which creates suitable habitat for many species of plants, amphibians, fungi, and bryophytes (Richardson and Danehy 2007). Maintaining intact riparian forests also increases habitat complexity in stream ecosystems and can provide more resources for organisms throughout many trophic levels. Sweka and Hartman (2006) found that forested riparian areas supply seeds, leaf litter, and other plant organic materials and energy inputs into streams. These inputs provide food and habitat for macroinvertebrates and is also vital to stream fish populations. In headwater streams of Connecticut, recent work by Kanno et al. (2015) reported that forest canopy cover was the most fundamental variable for the occurrence of brook trout (*Salvelinus fontinalis*). Riparian forests also input large woody debris into streams that create log jams and pools. These pools are important refuge for stream fish during flooding and low-flow events and may also be correlated with trout abundance (Sweka and Hartman 2006). In addition to providing critical habitat, log jams and pools store fine sediment and organic matter, which is beneficial to water quality (Sweka and Hartman 2006). Furthermore, maintaining intact riparian forests can reduce the risk of invasion by terrestrial invasive plants and help to ensure ecological integrity (Funk et al. 2008; Nunez-Mir et al. 2017), which may also facilitate the provisioning of ecosystem service benefits in stream and riparian habitats.

**Headwater Streams**

Headwater streams are of great importance in the United States as they are the most abundant stream or river habitat in total length and number (Allen and Castillo 2007). According to Richardson and Danehy (2007), there are many ways to define a headwater stream, but the most accepted definition is that it must have a stream channel width < 3 m and a mean annual
discharge of <57 l/s. Because of their abundance and small channel size, headwater streams are highly vulnerable to land use practices (e.g., forestry and agriculture) and other human activities (Kanno et al. 2015). Management and conservation efforts of these small streams are often overlooked as they are easily hidden by forest cover and may even be unmapped (Richardson and Danehy 2007). Headwater streams provide many ecosystem services such as water filtration and nutrient removal, although the extent of these services provided in both natural and managed areas are not yet fully understood (Richardson and Danehy 2007). Brown and Swan (2010) have shown that across three watersheds in Maryland, local habitat conditions directly impacted community structure of macroinvertebrates more so in headwater streams than in larger streams and rivers. This would suggest that local habitat conditions of headwater streams are highly important to structuring assemblages of stream fish populations, which feed primarily on macroinvertebrates that originate in headwater streams. These important habitats need to be better understood to guide proper management and conservation efforts to ensure their ecological integrity, the persistence of species that depend on them (Kanno et al. 2015), and the many ecosystem services that they provide.

The Role of Natural and Anthropogenic Disturbance

Natural and anthropogenic disturbance and modifications not only strongly influence the risk of invasive plant colonization but also the magnitude of the invasion, as these disturbances often result in canopy gaps, newly available resources (e.g., light, nutrients, space), and reduced competition. These disturbed sites can be easily colonized by invasive species (Vila and Weiner 2004; Richardson et al. 2007). Disturbed areas act as source populations and the surrounding habitats may experience greater risk of invasion as they receive inputs of invasive plant
propagules (McDonald and Urban 2006; Gonzalez-Moreno et al. 2012). Heavy equipment use can also increase the risk of invasion by soil scarification and inputs of outside seed sources that may be transported in soil or on log skidders and earth-moving machinery (McDonald and Urban 2006; McDonald et al. 2008).

Riparian habitats are considered at high risk of invasion because they are one of the most anthropogenically disturbed ecosystems in the world and are often highly modified (e.g., man-made slopes, culverts, roads, dams) with alterations for transportation, flow regulation, and drainage purposes (Allan and Flecker 1993; Liendo et al. 2015). These disturbances often introduce invasive plant seeds and propagules into riparian areas and facilitate their dispersal. Riparian habitats are also at high risk of invasion because they are commonly found adjacent to residential development, flowing water, and agricultural land use (Hood and Naiman 2000; Vicente et al. 2013). Once established, invasive plant communities can exist in isolated patches that may be connected through roads, streams, or rivers that act as corridors to facilitate movement of plants and propagules throughout the watershed by natural events and disturbance (Hood and Naiman 2000; Vicente et al. 2013). These corridors are often connected to urban environments with human settlement and industry, where many invasive plant propagules are introduced and transported into streams and riparian forests by moving water (Richardson et al. 2007). Natural disturbance events (e.g., storms, flooding, erosion, bank collapse) often compromise or uproot existing riparian plant communities and facilitate soil movement and sediment deposition. These open areas with newly available resources and complex hydrology that often expose soils in riparian areas are ideal environments for invasive plants to establish (Richardson et al. 2007). To reduce invasions of interior forests and protected areas we must
consider the surrounding areas that may be increasing invasive species propagule pressure and are facilitating invasion into these undisturbed areas (Gonzalez-Moreno et al. 2012).

**Research Questions & Objectives**

My research fills an important knowledge gap on invasive plants in northern New England because little has been done to quantify their impacts to riparian forests and headwater stream ecosystems and investigate their distribution along headwater streams in the New England region. The goal of this study was to quantify the impacts caused from terrestrial invasive plants on riparian forest structure, stream physical habitat, soil composition and functioning, and the provisioning of ecosystem services. To achieve this goal, I also set out to better understand the distribution of terrestrial invasive plants along headwater streams and riparian forests in northern New England. *The objective of this research was to quantify the impacts caused from terrestrial invasive plant invasions on riparian forest structure, native plant communities, stream physical habitat, soil composition, and soil functioning*. All of these can have direct impacts to ecosystem services such as water quality, carbon storage, and providing critical habitat that support cold water fisheries. My hypotheses were: (1) riparian sites dominated by invasive plants will have degraded forest structure and may experience negative impacts to soil functioning and composition. Impacts may include decreased litter and duff layers and possible erosion and silt into streams because invasive plants are fast growing and form dense monocultures that can quickly reduce diversity and ground cover, exposing soils to direct precipitation and moving water; (2) riparian sites dominated by invasive plant species will have less coarse woody debris (CWD) and less organic inputs because invasive plants may reduce tree basal area and regeneration of woody plants and native trees; and, (3) riparian sites with
established invasive species may have poorer stream habitat quality because of short, dense plant canopies allowing more light to enter the stream and fewer functional groups present in the plant community.

**Methods**

**Study Area**

This research took place along Garland Brook, a headwater stream of the Connecticut River in the town of Lancaster, New Hampshire, where it serves as the town water supply (Fig. 6). Lancaster is in Coös County and is the northernmost county of New Hampshire, which borders Canada. Lancaster has a population of 3,507 with 27.2 people per km² (2010 census) and is considered the gateway to the Great North Woods Region. Lancaster’s elevation is 263 m but the eastern part of the town falls in the White Mountain National Forest and is 1,000 m above sea level. Garland Brook is drained by the Israel River, which joins the Connecticut River just outside of Lancaster. The Connecticut River is widely known for its diverse fishery and drains 29,137 km². It also serves as the boundary between New Hampshire and Vermont. The Connecticut River Watershed includes more than 2.4 million residents, 44,000 road-stream crossings, and is home to several endangered and threatened flora and fauna species. Common land use practices in Coös County are agriculture and forestry. To the north of Lancaster, large tracts of land are primarily managed for timber products and directly to the south and east are the federally managed White Mountain National Forest (303,859 ha). Surveys for this study were conducted in a gradient beginning in an urban/agricultural landscape and moving towards rural, more forested and rugged topography. The upper Connecticut Cooperative Invasive Species  

---

2 [https://www.americanrivers.org/river/connecticut-river/](https://www.americanrivers.org/river/connecticut-river/)
Management Area (UCCISMA) has compiled a list of invasive and non-native plant species that are a priority for management in the northern section of the upper Connecticut River watershed (Table 4). In addition to this list, I have added multiflora rose (*Rosa multiflora*) and Japanese barberry (*Berberis thunbergii*) because they are among the most common invasive species found in the region, along with glossy buckthorn (*Rhamnus frangula*) and oriental bittersweet (*Celastrus orbiculata*) (McDonald et al. 2008).

**Table 4.** Major invasive plant species in the northern section of the upper Connecticut River watershed and the northern New England region.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th>Common Name</th>
<th>Scientific Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Common Buckthorn</td>
<td><em>Rhamnus cathartica</em></td>
<td>Japanese Barberry</td>
<td><em>Berberis thunbergii</em></td>
</tr>
<tr>
<td>Common Reed</td>
<td><em>Phragmites australis</em></td>
<td>Knotweed s.l. (sensu lato)</td>
<td><em>Fallopia spp.</em></td>
</tr>
<tr>
<td>False Spirea</td>
<td><em>Sorbaria sorbifolia</em></td>
<td>Multiflora Rose</td>
<td><em>Rosa rugosa</em></td>
</tr>
<tr>
<td>Garlic Mustard</td>
<td><em>Alliaria petiolata</em></td>
<td>Oriental Bittersweet</td>
<td><em>Celastrus orbiculatus</em></td>
</tr>
<tr>
<td>Glossy Buckthorn</td>
<td><em>Rhamnus frangula</em></td>
<td>Purple Loosestrife</td>
<td><em>Lythrum salicaria</em></td>
</tr>
<tr>
<td>Goutweed</td>
<td><em>Aegopodium podagraria</em></td>
<td>Wild Parsnip</td>
<td><em>Pastinaca sativa</em></td>
</tr>
<tr>
<td>Honeysuckle</td>
<td><em>Lonicera spp.</em></td>
<td>Yellow Flag Iris</td>
<td><em>Iris pseudacorus</em></td>
</tr>
</tbody>
</table>

**Site Description**

Garland Brook originates in the Kilkenny Mountain Range on a ridge between the summits of Terrace Mountain (1082 m), Mount Weeks (1185 m), and South Weeks Mountain (1183 m) (Fig. 6). Multiple first order headwater streams (~4.2 km stream length) converge at approximately 597 m elevation and flows in one continuous channel for ~11.3 km through undisturbed closed canopy forests, many of which are within the MWNF. Garland Brook then bifurcates at the base of the town’s drinking water supply treatment facility, which also served as the boundary to split the lower and upper watershed (described below). These two stream channels flow parallel for ~3.3 km, one to the north and one to the south. The land between them (i.e., river island) is on average ~400 m wide. This land is primarily agricultural with forested
riparian buffers and contains a rural, two-lane road with three stream/road crossings. In addition, the stream reach to the south contains an historic logging mill (Garland Mill) that is heavily invaded by knotweed. Here, knotweed has completely taken over both sides of the mill access road (480 m² patch), the storage yard (735 m² patch) and along the stream bank (132 m² patch). Downstream of the mill, Garland Brook begins to converge as it flows along both sides of a large pasture on the river island. Its confluence is ~160 m downstream of the pasture in a closed canopy forest. Garland Brook continues through this intact forest for ~1.3 km where it emerges and flows along another mowed pasture for ~0.7 km before meeting with Brook Road. Garland Brook then flows along Brook Road (bankside right), and a cattle-grazing pasture (bankside left) for ~1.5 km with an intermittent, narrow riparian buffer present. Garland Brook diverges from the road and flows through a matrix of agriculture, forests, and large homesteads for ~1.8 km until it meets a large plowed field. This is where the vegetation surveys for invasive plants ended, ~3 km before flowing into the Israel river.
Figure 6. Map of Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) plots along Garland Brook in Lancaster, NH (n=36).

**Riparian Invasion Along Headwater Streams**

*Stream surveys for invasive plants:* To map and analyze the distribution of invasive plants along Garland Brook and to establish plots around invasive communities, I surveyed both sides of the riparian zone of Garland Brook (~28 km of stream length) to locate all of the invasive plants present in Garland Brook’s riparian areas. When an invasive community or patch was detected, the length and width was measured to the nearest 0.25 m to calculate the area of the patch (m²). Species, bank side, and GPS coordinates were also recorded in the GAIA GPS (Trailbehind Inc.) app as a waypoint through an iPad mini 4 (Apple Inc., CA, USA).
The Impacts of Invasive Plants

Non-invaded plot layout: I conducted systematic forest structure, vegetation, and stream habitat surveys to obtain data for Garland Brook (~28 km total stream length) and adjacent riparian areas in non-invaded and invaded stands to quantify the differences between them. For this study, the Garland Brook watershed (~28 km of total stream length) was classified as either lower watershed or upper watershed based on differences in road density, development pressure, anthropogenic disturbance, and elevation. This distinction was made through visual inspection of satellite imagery via GAIA GPS app and verified through direct observations while conducting surveys. Generally, the upper watershed has no public roads, mostly closed canopy forests, and rapidly increasing elevation. The lower watershed contains public and private access roads, development (e.g., homesteads), and a high density of agriculture land use. Lower and Upper Watershed control plots (n=23) were established systematically every 800 m on alternating bank sides for the entire length of Garland Brook to get baseline data of vegetation, forest structure, stream habitat, soil structure, soil functioning, bank stability, and available coarse woody debris (CWD). These systematic plots are referred to as Lower Watershed (LW) or Upper Watershed (UW) (Fig. 6). To further characterize the watershed, elevation was documented at each plot to calculate the gradient of each stream reach using an iPad mini 4 (Apple Inc, CA, USA) and Gaia GPS (Trailbehind Inc.). Stream velocity and discharge was also measured (methods described below) at every other systematic plot (n=12).

Invaded plot layout: Once all invasive plant communities were mapped along Garland Brook, additional survey plots containing invasive species were established (n=15). These are referred to as Invaded plots (I). Invaded plot locations were established by taking the total stream length that
contained invasive species (i.e., the stream length between the first observed patch and the last; ~15.7 km), and equally spacing 15 plots in that amount of stream length. Once I knew the target vicinity for the plot, I selected nearby invasive patches residing on the bank based on size of patch and bank side. My goal was to capture a gradient of different stream channel widths, magnitudes of invasion, and to sample both bank sides equally. Overall plot numbers were limited by time and effort required to sample the entirety of Garland Brook.

![Diagram of plot layout](image)

**Figure 7.** Plot layout used on systematically placed Lower Watershed (LW), Upper Watershed (UW) plots (indicated by Syst.) and also Invaded (I) plots that are centered around invasive communities (indicated by Inv.) along Garland Brook in Lancaster, NH (n=36).

**Non-invaded and Invaded plot design:** Lower Watershed and Upper Watershed systematic plots were 800 m of stream length apart (plots pre-established in Gaia GPS using Gaia GPS measuring tool) and Invaded plots were centered around the invasive community on the stream bank (Fig. 7). Lower, Upper, and Invaded plots (Plot A) were 100 m² (10 m x 10 m), with the front stream edge starting where the vegetation begins at the stream bank (hereafter, stream bank or front...
A 2 m wide transect was established through the center of plot A, (perpendicular to the stream) and extended through the riparian zone to the back edge of plot A (Fig. 8). This transect was then divided into two 10 m² subplots, referred to as B and C plot (each 2 m x 5 m) to be able to capture changes within the riparian zone and characterize what is occurring along the stream within the first 5 m of the riparian buffer and also upland (5-10 m of the riparian buffer) (Fig. 8).

![Figure 8](image)

**Figure 8.** Plot A and subplot B (streamside) and C (upland) design used on Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH (n=36).

*Invaded threshold:* In order to quantify the impacts caused from terrestrial invasive plants, I set a 10% invaded threshold to be able to compare a more natural, uninverted habitat to an invaded habitat. Once all systematic and invasive vegetation surveys were complete, a systematic plot (LW or UW) with > 10% invasive stem density (stems/m²) was reclassified to an Invaded plot.
Conversely, if an Invaded plot contained < 10% invasive stem density, the plot would then be reclassified as a Lower or Upper Watershed systematic plot. After reclassification, I ensured that all plots were at least 50 m apart.

**Land cover classification:** I classified land cover within a 500 m radius buffer around all plots using ArcGIS 10.5 to further characterize and generalize the landcover of the three plot types: Invaded plots (n = 15), Lower Watershed plots (n = 13), and Upper Watershed plots (n=10). Land cover classification types were characterized using LANDFIRE (LF) data (www.landfire.gov) in ArcGIS 10.5.

**Vegetation surveys:** Within plot A, tree species and DBH (cm) was collected for all trees > 10 cm DBH (Fig. 8). Height (m) was measured on one dominant and one co-dominant tree using a Haglof Vertex IV and Transponder III (Haglöf Sweden AB, Långsele, Sweden) and species was documented. Visual estimates of percent cover were made within plot A to characterize: overstory canopy (> 5 m in height); understory (0.5 to 5 m in height); and herbaceous and non-herbaceous ground cover (< 0.5 m in height) (Kaufman and Robison 1998). Ground cover was further characterized into 15 categories by visually estimating percent cover: (1) hay-scented fern (*Dennstaedtia punctilobula*); (2) various other fern; (3) horsetail; (4) leaf litter (invasive vs. native); (5) moss; (6) club moss (*Lycopodium*); (7) various herbaceous; (8) bare ground; (9) grasses (*Poaceae*), sedges (*Cyperaceae*), and rushes (*Juncaceae*); (10) invasive species; (11) knotweed (live and dead); (12) shore; (13) boulder/cobble; (14) road; and (15) cut grass. At all plot locations, any signs of disturbance or anthropogenic modification were noted, and GPS coordinates, bank facing direction, and dominant land use/land cover was documented. Within subplot B and C (nested 2 m transect), species and stem density of all woody vegetation was
collected and binned into size classes 1-4: 1 (0-1 m), 2 (1.1-2 m), 3 (2.1-5 m), or 4 (>5 m).

Invasive species percent cover was also estimated within plot A.

**Canopy cover:** Canopy cover was obtained by using a concave spherical densiometer and modifying it by taping or “blocking off” sections of the grid to help eliminate bias by overlapping vegetation (USGS 1998). This modification only uses 17 of the 37 line intersections on the concave mirror. Canopy cover was measured at the center of plot A (upland) for baseline riparian forest canopy cover and also at the front edge of subplot B, at the stream, to compare stream shading ability of riparian zones between native and invasive communities.

**Open canopy angle:** To further characterize shading ability and the amount of solar radiation exposed to the stream, open canopy angle was determined using a Haglof Vertex IV and Transponder III (Haglöf Sweden AB, Långsele, Sweden) by measuring the right and left canopy angle while standing at the center of the stream channel (Fig. 9). The left and right angles were measured to the tallest object on each bank and the total was subtracted from 180 to give the open canopy angle (USGS 1998). The tallest object on each bank was also noted (e.g., bank, vegetation species, building, etc.).

![Figure 9. Open canopy angle measurement explanation (adapted from USGS 1998) used on Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH (n=36).](image)
Coarse woody debris (CWD): Coarse woody debris (CWD) was quantified in all systematic and Invaded plots using the line intersect sampling (LIS) method (Kershaw et al. 2017). LIS tallies all CWD that crosses a sample line placed systematically or randomly throughout an area of interest. For this study, two sample lines (transects) were systematically placed in all plots (plot A). The first transect (10 m long) was placed through the center of plot A, perpendicular to the stream and extending from the front of subplot B (stream bank) to the rear edge of subplot C. This transect was used to quantify the amount of upland CWD. The second transect was established (10 m long) along the front edge of plot A, parallel to the stream, and quantifies the amount of CWD at the stream edge. In addition to length (m) and diameter at intersection point (cm) for LIS calculations, decay class, species (if possible), and distance to the stream (m) was collected. Through visual inspection, the origin of the piece was also documented (onsite or offsite) for quantification and comparison purposes of CWD, because only pieces that originated onsite were included in data analysis and pieces that may have washed up or floated from elsewhere were excluded.

Soil characteristics and function: To analyze soil characteristics and functioning differences between invaded and non-invaded sites, the leaf litter and duff layer (i.e., decomposed organic soil layer above the mineral A horizon) depth were both measured to the nearest 0.1 cm (USFS 2011). Shallow soil profiles were dug on the forest floor with a small trowel 2 m from the front edge of subplot B, where litter and duff depths were measured, and soil type was characterized (Fig. 8). Infiltration rates of water into soil (cm/min) were also measured here to assess overland flow potential by using a 15.24 cm diameter cylinder (single-ring infiltrometer made from plastic
PVC) to perform an infiltration test (USDA 2014). Any presence of erosion at the bank was also documented.

**Bank stability index:** Bank stability index was calculated at each plot because it is useful for habitat assessments and can be correlated to land use/land cover patterns (Fitzpatrick et al. 1998). The bank stability index was also used as an indicator of bank conditions. I calculated the bank stability index by assigning a score to each of the four variables (bank angle, bank height, vegetation cover, and bank substrate) with a total maximum score of 22 (Table 5). Typically, total scores between 4-7 are considered stable, 8-10 are at risk, 11-15 are unstable, and 16-22 are very unstable (Fitzpatrick et al. 1998). If the species of vegetation along the bank was invasive, the species was noted and considered when assessing bank conditions.

Table 5. Bank stability index scores for each category (adapted from USGS 1998) to be used on Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH (n=36).

<table>
<thead>
<tr>
<th>Bank Characteristics</th>
<th>Measurement</th>
<th>Score</th>
</tr>
</thead>
<tbody>
<tr>
<td>Angle (degrees)</td>
<td>0-30</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>31-60</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>&gt; 60</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>&gt; 80</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>50-80</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>20-&lt;50</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>&lt;20</td>
<td>4</td>
</tr>
<tr>
<td>Vegetation Cover (%)</td>
<td>0-1</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>1.1-2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>2.1-3</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>3.1-4</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>&gt;4</td>
<td>5</td>
</tr>
<tr>
<td>Height (m)</td>
<td>Bedrock, artificial</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Boulder, cobble</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Silt</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Sand</td>
<td>8</td>
</tr>
<tr>
<td>Substrate</td>
<td>Gravel, sand</td>
<td>10</td>
</tr>
</tbody>
</table>
**Embeddedness**: Substrate embeddedness was measured to assess the health of the stream substratum and can also be an indicator of water quality (Fitzpatrick et al. 1998). Although the definition of embeddedness is the fraction of a particle’s surface at the stream bottom that is surrounded or embedded by sand or fine sediments (Fitzpatrick et al. 1998), we did not consider sand in this assessment. We only measured silt because much of the naturally occurring stream substratum was coarse sand and depending on the flow intensity, the sand may wash up onto the boulder/cobble/gravel or other stream substratum. We were only interested in embeddedness that was caused by silt from erosion and loss of topsoil from the riparian zone, so sand was not counted as embeddedness. Embeddedness was measured in Invaded plots to compare upstream and downstream of invasive communities (Fig. 10). Measurements were taken ~5 m upstream and ~5 m downstream of the Invaded plot. Instances when invasive communities were longer than 10 m, measurements were taken ~5 m upstream and ~5 m downstream of the invasive community. These measurements were taken at five sampling points along a transect through the stream channel (0, 25, 50, 75, and 100 percent of the measured wetted width). I averaged the values (%) from each of the 5 sampling points.

![Figure 10. Embeddedness and substrate sampling locations at Invaded (I) plots along Garland Brook in Lancaster, NH (n=13).](image)
The water depth was also measured at each substrate sampling location and averaged. Dominant and sub-dominant bed substrate (fines/silt/sand, small gravel, large gravel, cobble, boulder, bedrock, organic) was classified at these five sampling points and the stream reach was characterized as a pool, riffle, run, or cascade (Magee 2017) (Table 6).

Table 6. Size chart to classify stream substrate particle size (adapted from Magee 2017) at the Lower Watershed, Upper Watershed, and Invaded plots along Garland Brook in Lancaster, NH (n=36).

<table>
<thead>
<tr>
<th>Substrate</th>
<th>Code</th>
<th>Size (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fines/silt/sand</td>
<td>SA</td>
<td>&lt; 0.6</td>
</tr>
<tr>
<td>Small gravel</td>
<td>G1</td>
<td>0.6 to 7.6</td>
</tr>
<tr>
<td>Large gravel</td>
<td>G2</td>
<td>7.6 to 15.2</td>
</tr>
<tr>
<td>Cobble</td>
<td>CO</td>
<td>15.2 to 30.5</td>
</tr>
<tr>
<td>Boulder</td>
<td>BO</td>
<td>&gt; 30.5</td>
</tr>
<tr>
<td>Bedrock</td>
<td>BR</td>
<td>Solid mass</td>
</tr>
<tr>
<td>Organic</td>
<td>OR</td>
<td>Wood or herbaceous</td>
</tr>
</tbody>
</table>

**Discharge:** In the stream reach at every other systematic survey (n = 12), stream discharge was calculated to further characterize the stream reach and watershed using the time travel of a float technique (Peck et al. 2001). This technique measures stream velocity \( V = \left[ \frac{\text{travel distance}}{\text{average travel time}} \right] \times \text{substratum coefficient} \) and uses the cross sectional area \( A = \text{width} \times \text{average depth} \) to calculate discharge rates (discharge rate = \( \frac{\text{cross sectional area} \times \text{length} \times \text{coefficient}}{\text{time}} \)).

**Statistical Analysis**

To address the objective of quantifying differences in riparian forest structure, native plant communities, stream physical habitat, soil structure, and soil functioning, JMP Pro 13 (SAS, Cary, NC USA) was used to perform all statistical analyses. A non-parametric multivariate Kruskal-Wallis rank sums test was used to examine the medians of the dependent response.
variables and the different levels of the independent variable: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) plots. A non-parametric test was needed because data violated the assumptions of a parametric test, even after the data was log-transformed or arcsine square root transformed. A post-hoc non-parametric Wilcoxon test on each pair was used to compare response variables between two treatment types (e.g., LW-I).

**Results**

**Riparian Invasion Along Headwater Streams**

The 28 km of stream surveys for terrestrial invasive plants conducted in the riparian areas along both sides of Garland Brook detected 324 patches of invasive plants. All invasive plants were found in the lower watershed and made up approximately 3.2% of the total lower watershed riparian buffer land area (Fig. 11). Invasive plant species observed in the riparian areas along Garland Brook were knotweed (55% of all invasives detected), Morrow’s honeysuckle (43%), and glossy buckthorn (2%) (Table 7).
Figure 11. Map of terrestrial invasive plant occurrences found along Garland Brook in Lancaster, NH.

Table 7. Patch sizes and occurrences of terrestrial invasive plant species found along Garland Brook, Lancaster, NH.

<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific Name</th>
<th>Largest Patch Size (m²)</th>
<th>Mean Patch Size (m²)</th>
<th># of Occurrences</th>
<th>Total Patch Size (m²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Morrow’s Honeysuckle</td>
<td>Lonicera morrowii</td>
<td>440.0</td>
<td>9.0</td>
<td>140</td>
<td>1336.0</td>
</tr>
<tr>
<td>Knotweed s.l.</td>
<td>Fallopia spp.</td>
<td>3000.0</td>
<td>78.2</td>
<td>176</td>
<td>13763.0</td>
</tr>
<tr>
<td>Glossy Buckthorn</td>
<td>Rhamnus frangula</td>
<td>18.0</td>
<td>9.2</td>
<td>8</td>
<td>73.5</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td></td>
<td></td>
<td></td>
<td><strong>324</strong></td>
<td><strong>15172.5</strong></td>
</tr>
</tbody>
</table>

The Impacts of Invasive Plants

*Invaded threshold and land cover:* To comply with the pre-set invaded threshold, two Lower Watershed and two Invaded plots were reclassified. This meant that two Invaded plots (I-12 & I-15) that had a low-level of invasion (5%) were reclassified as Lower Watershed plots and two
systematic Lower Watershed plots (LW-1 & LW-2) that had a moderate level of invasion (20%) were reclassified as Invaded plots. Furthermore, Plot I-10 was discarded as it was unlike all other plots and was an outlier in all of the response variables. This is because it was located in the storage yard of Garland Mill and consisted of a 2 m wide strip of knotweed along the brook and then 8 m of cut grass. In addition, one LW plot was discarded (LW-1) as it was in close proximity to an Invaded plot after it was reclassified. After reclassification analysis and removal of plots, 36 remained. (LW=13, UW=10, I=13). Since all of the invasive plant communities were found in the lower watershed, many comparisons below are focused on the differences between Invaded plots (i.e., plots with > 10% invaded stems/m²) and Lower Watershed plots (i.e., plots with < 10% invaded stems/m²) to highlight the potential impacts of invasive plants (Invaded plots). Land cover analysis showed that Invaded plots had the least amount of forest cover and the most agricultural land use within a 500 m radius of the plot center (Table 8). Upper Watershed plots were almost entirely forested (98%) with a small amount of agriculture (1%) and riparian land cover (1%) (i.e., floodplains, marsh, swamp as per LF classification). Lower Watershed plots had more roads and development than Invaded plots (Table 8).

Table 8. Land cover classifications found within 500 m radius buffers around the Invaded, Lower Watershed, and Upper Watershed plot centers along Garland Brook in Lancaster, NH (n=36).

<table>
<thead>
<tr>
<th>Plot Type</th>
<th>Forested</th>
<th>Agricultural</th>
<th>Riparian</th>
<th>Developed</th>
<th>Road</th>
</tr>
</thead>
<tbody>
<tr>
<td>Invaded (I)</td>
<td>64%</td>
<td>12%</td>
<td>17%</td>
<td>4%</td>
<td>3%</td>
</tr>
<tr>
<td>Lower Watershed (LW)</td>
<td>71%</td>
<td>8%</td>
<td>8%</td>
<td>8%</td>
<td>5%</td>
</tr>
<tr>
<td>Upper Watershed (UW)</td>
<td>98%</td>
<td>1%</td>
<td>1%</td>
<td>0%</td>
<td>0%</td>
</tr>
</tbody>
</table>
Plant composition (native and invasive):

When analyzing the total composition of invasive stems in the combined B (streamside) and C (upland) subplots, 40% of all stems in the Invaded plots were invasive species compared to 2% in the Lower Watershed plots, and zero invasive species in the Upper Watershed plots (Fig. 12). At all plot types, total stem densities were consistently higher within the first 5 m of the riparian zone (streamside, subplot B) when compared to the upland 5 m of the riparian zone (upland, subplot C). When comparing stem density between Invaded plots (B and C subplot $\bar{x}$=25.9, 20.6 stems/m², respectively) and Lower Watershed plots (B and C subplot $\bar{x}$=18.2, 15.6 stems/m², respectively), Invaded plots tended to have greater overall stem density (B and C subplot $p$=0.33, 0.22, respectively) (Fig. 12).

Figure 12. Mean total native and invasive stems/m² at the streamside (subplot B) and upland (subplot C) in the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36).
Plant composition (by size class): Invaded plots had a statistically significant difference in the density of size class 2 (1.1-2 m) and 3 (2.1-5 m) stems in the first 5 m of the riparian zone (streamside), with greater densities compared to both Lower and Upper Watershed plots (Fig. 13, Table 9). In the upland 5 m (subplot C), Invaded plots had significantly more size class 2 and 3 stems/m² when compared to Lower Watershed plots, although a less significant difference when compared to Upper Watershed plots (Fig. 13, Table 9). Size class 4 (>5 m) stem densities were low in all plot types but were almost non-existent in Invaded plots (streamside and upland) with lower size class 4 stem densities in Invaded stands compared to both Lower Watershed (B subplot \( p=0.22 \), C subplot 0.30) and Upper Watershed plots (B subplot \( p=0.26 \), C subplot 0.26) (Fig. 13, Table 9).

**Figure 13.** Mean total stems/m² (by size class 1-4) at the streamside (subplot B) and upland (subplot C) found in the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36).
**Tree Basal Area:** Tree basal area (m²/ha) was found to be statistically similar between Lower and Upper Watershed plots \((p=0.40)\), although Invaded plots were significantly lower \((\bar{x}=11.5 \text{ m}^2/\text{ha})\) when compared to both Lower \((\bar{x}=34.2 \text{ m}^2/\text{ha})\) and Upper Watershed plots \((\bar{x}=34.8 \text{ m}^2/\text{ha})\) \((p=0.03, <0.01, \text{ respectively})\) (Fig. 14).

![Boxplot of tree basal area m²/ha in the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36). Dots represent individual data points and dashed line indicates the mean.](image)

**Canopy closure:** Canopy closure along the stream (front edge of subplot B) was significantly lower in Invaded plots \((\bar{x}=45.7\%)\) when compared to both Lower Watershed \((\bar{x}=73.8\%)\) and Upper Watershed plots \((\bar{x}=85.4\%)\) \((p=0.02, <0.01, \text{ respectively})\) (Fig. 15). Measurements of upland canopy closure was similar between the three plot types \((p=0.50)\), although the mean upland canopy closure was slightly higher in Invaded plots \((\bar{x}=85.7\%)\) when compared to the Lower Watershed plots \((\bar{x}=78.6\%)\) (Fig. 15).
Figure 15. Mean canopy closure measured at the stream edge and upland (center of A plot) in the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36).

Open canopy angle: Open canopy angle and the percent open canopy was calculated for the three plot types. Invaded plots had significantly higher percent open canopy above the stream channel ($\bar{x}=45.7\%$) compared to both Lower Watershed ($\bar{x}=19.1\%$) and Upper Watershed plots ($\bar{x}=5.6\%$) ($p<0.01$, <0.01, respectively) (Fig. 16).
Figure 16. Boxplot showing the amount of open canopy (%) measured at the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36). Dots represent individual data points and dashed line indicates the mean.

**Tree regeneration:** Regeneration of native tree species (stems/ha of size class 2, 1.1-2 m native trees) was consistently lower in Invaded plots in both B and C subplots ($\bar{x}$=538, 1,308 stems/ha, respectively) compared to Lower ($\bar{x}$=2,000, 2,154 stems/ha, respectively) and Upper Watershed plots ($\bar{x}$=3,400, 3,700 stems/ha, respectively) (Fig. 17, Table 9). This was only statistically significant when comparing Invaded plots to the Upper Watershed (B and C subplot $p$=<0.01, 0.02, respectively), although a substantial difference was also observed when comparing Invaded plots to Lower Watershed plots (B and C subplot $p$=0.11, 0.21, respectively). (Fig. 17, Table 9).
Coarse woody debris (CWD): The volume of CWD measured along the stream channel was significantly lower in Invaded plots ($\bar{x}$=0.0 m$^3$/ha) when compared to Lower Watershed plots ($\bar{x}$=35.7 m$^3$/ha) and Upper Watershed plots ($\bar{x}$=22.9 m$^3$/ha) ($p$=0.02, $<$0.01, respectively) (Fig.18). The volume of upland CWD was consistently lower in Invaded plots ($\bar{x}$=38.2 m$^3$/ha) compared to Lower Watershed plots ($\bar{x}$=55 m$^3$/ha), although not statistically significant ($p$=0.17) (Fig.18). The volume of upland CWD was greatest in the Upper Watershed ($\bar{x}$=66.9 m$^3$/ha), which is significantly greater than what was found in Invaded plots, but not statistically different than the amount found in Lower Watershed plots ($p$=0.04, 0.45, respectively) (Fig.18).
Figure 18. Mean coarse woody debris (m$^3$/ha) observed along the stream and upland in the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study site (n=36).

**Litter and duff layers:** Leaf litter and duff layers in the soil were both measured and combined to further analyze the organic material present in the survey plots. Invaded plots had a statistically significant reduction of litter and duff layers ($\bar{x}$=1.0 cm) compared to both Lower Watershed ($\bar{x}$=2.5 cm) and Upper Watershed plots ($\bar{x}$=8.8 cm) ($p$=0.03, <0.01, respectively) (Fig. 19). Upper Watershed plots consistently had thicker litter and duff layers (Fig. 19).
Figure 19. Combined mean litter and duff layers (cm) measured at the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36).

*Bare ground:* Bare ground was generally higher in Invaded plots (\(\bar{x}=12\%\)) when compared to Lower Watershed plots (6.8%), although not statistically significant at \(\alpha = 0.05\) level (\(p=0.09\)) (Fig. 20). Upper Watershed plots had significantly lower amounts of bare ground present (\(\bar{x} =1.2\)) when compared to Lower Watershed and Invaded plots (\(p=0.02, <0.01\), respectively) (Fig. 20).
Figure 20. Bare ground measured at the three plot types: Invaded (I), Lower Watershed (LW), and Upper Watershed (UW) in the Lancaster, N.H. study sites (n=36). Dots represent individual data points and dashed line indicates the mean.

**Ground cover:** Ground cover percentage was similar among all three plot types \( (p=0.55) \), although slightly lower in Invaded plots \( (\bar{x}=45.8\%) \) when compared to Lower Watershed \( (\bar{x}=53.5\%) \) and Upper Watershed plots \( (\bar{x}=43\%) \) (Table 9).

**Infiltration rates:** Infiltration rates (cm/min) were also analyzed in the three plot types to determine the ability of water to percolate into the soil and reduce the potential for overland flow. Upper Watershed plots had significantly faster infiltration rates \( (\bar{x}=25.7\text{ cm/min}) \) compared to Lower Watershed \( (\bar{x}=8.1\text{ cm/min}) \) and Invaded plots \( (\bar{x}=12.6\text{ cm/min}) \) \( (p<0.01, 0.01, \text{ respectively}) \). When comparing infiltration rates between the Lower Watershed and Invaded plots, the differences in infiltration rates were not statistically significant \( (p=0.22) \) (Table 10).
**Embeddedness**: Embeddedness analysis revealed that the presence and abundance of silt covering stream substrate was four times higher downstream of Invaded plots ($\bar{x} = 16.0\%$, SE=6%) when compared to upstream of the Invasive plots ($\bar{x} = 4.0\%$, SE=1.0%) ($p=0.10$) (Fig. 21).

![Figure 21. Boxplot of embeddedness (%) measured upstream and downstream of invasive plant communities within Invaded plots in the Lancaster, N.H. study area (n=36). Dots represent individual data points and dashed line indicates the mean.](image-url)
Table 9. Mean, standard error, and Kruskal-Wallis (KW) p-values for the forest structure response variables in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites (n=36). Additional post-hoc Wilcoxon test on each pair to compare response variables between two treatment types. Bold p-value indicates significance at $\alpha = 0.05$ level.

<table>
<thead>
<tr>
<th>Response Variable</th>
<th>Lower (LW)</th>
<th>Upper (UW)</th>
<th>Invaded (I)</th>
<th>KW p-value</th>
<th>LW-UW-I</th>
<th>LW-I</th>
<th>UW-I</th>
<th>UW-LW</th>
</tr>
</thead>
<tbody>
<tr>
<td>Upland (A plot) canopy closure (%)</td>
<td>78.6 (6.8)</td>
<td>92.2 (1.3)</td>
<td>85.7 (5.1)</td>
<td>0.50</td>
<td>0.46</td>
<td>0.57</td>
<td>0.30</td>
<td></td>
</tr>
<tr>
<td>Bare ground (%)</td>
<td>6.8 (2.1)</td>
<td>1.2 (0.4)</td>
<td>12 (2.7)</td>
<td>&lt;0.01</td>
<td>0.09</td>
<td>&lt;0.01</td>
<td>0.02</td>
<td></td>
</tr>
<tr>
<td>Upland CWD (m$^3$/ha)</td>
<td>55 (21.2)</td>
<td>66.9 (15.8)</td>
<td>38.2 (19.1)</td>
<td>0.10</td>
<td>0.17</td>
<td>0.04</td>
<td>0.45</td>
<td></td>
</tr>
<tr>
<td>Ground cover (%)</td>
<td>53.5 (5.7)</td>
<td>43 (7.7)</td>
<td>45.8 (6.6)</td>
<td>0.55</td>
<td>0.47</td>
<td>0.73</td>
<td>0.32</td>
<td></td>
</tr>
<tr>
<td>Leaf litter (%)</td>
<td>46.5 (8.0)</td>
<td>82.5 (5.0)</td>
<td>29.6 (5.8)</td>
<td>&lt;0.01</td>
<td>0.15</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Tree basal area (m$^2$/ha)</td>
<td>34.2 (9.4)</td>
<td>34.8 (4.7)</td>
<td>11.5 (3.9)</td>
<td>&lt;0.01</td>
<td>0.03</td>
<td>&lt;0.01</td>
<td>0.40</td>
<td></td>
</tr>
<tr>
<td>B plot tree regeneration (stems/ha)</td>
<td>2,000 (641)</td>
<td>3,400 (1087)</td>
<td>538 (268)</td>
<td>0.03</td>
<td>0.11</td>
<td>&lt;0.01</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td>C plot tree regeneration (stems/ha)</td>
<td>2,154 (619)</td>
<td>3,700 (1359)</td>
<td>1,308 (702)</td>
<td>0.08</td>
<td>0.21</td>
<td>0.02</td>
<td>0.38</td>
<td></td>
</tr>
<tr>
<td>B plot stems/m$^2$ (total)</td>
<td>18.2 (3.2)</td>
<td>31.4 (9.3)</td>
<td>25.9 (25.9)</td>
<td>0.21</td>
<td>0.33</td>
<td>0.40</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td>B plot stems/m$^2$ (size class 1)</td>
<td>16.5 (3.1)</td>
<td>30.3 (9.3)</td>
<td>20.6 (6.1)</td>
<td>0.17</td>
<td>0.66</td>
<td>0.13</td>
<td>0.09</td>
<td></td>
</tr>
<tr>
<td>B plot stems/m$^2$ (size class 2)</td>
<td>1.1 (0.6)</td>
<td>0.8 (0.2)</td>
<td>2.7 (0.5)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>0.71</td>
<td></td>
</tr>
<tr>
<td>B plot stems/m$^2$ (size class 3)</td>
<td>0.5 (0.2)</td>
<td>0.4 (0.1)</td>
<td>2.5 (0.6)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>0.90</td>
<td></td>
</tr>
<tr>
<td>B plot stems/m$^2$ (size class 4)</td>
<td>0.1 (0.1)</td>
<td>0.1 (0)</td>
<td>0 (0)</td>
<td>0.38</td>
<td>0.22</td>
<td>0.26</td>
<td>0.70</td>
<td></td>
</tr>
<tr>
<td>C plot stems/m$^2$ (total)</td>
<td>15.6 (4)</td>
<td>20.4 (5.2)</td>
<td>20.6 (4.5)</td>
<td>0.29</td>
<td>0.22</td>
<td>0.64</td>
<td>0.18</td>
<td></td>
</tr>
<tr>
<td>C plot stems/m$^2$ (size class 1)</td>
<td>14.8 (4.1)</td>
<td>19.2 (5.2)</td>
<td>17.6 (4.6)</td>
<td>0.37</td>
<td>0.56</td>
<td>0.40</td>
<td>0.18</td>
<td></td>
</tr>
<tr>
<td>C plot stems/m$^2$ (size class 2)</td>
<td>0.3 (0.1)</td>
<td>0.8 (0.2)</td>
<td>1.3 (0.2)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>0.14</td>
<td>0.02</td>
<td></td>
</tr>
<tr>
<td>C plot stems/m$^2$ (size class 3)</td>
<td>0.3 (0.1)</td>
<td>0.3 (0.1)</td>
<td>1.6 (0.5)</td>
<td>0.04</td>
<td>0.02</td>
<td>0.07</td>
<td>0.58</td>
<td></td>
</tr>
<tr>
<td>C plot stems/m$^2$ (size class 4)</td>
<td>0.1 (0.1)</td>
<td>0.1 (0)</td>
<td>0 (0)</td>
<td>0.46</td>
<td>0.30</td>
<td>0.26</td>
<td>0.94</td>
<td></td>
</tr>
<tr>
<td>B plot native stems/m$^2$</td>
<td>17.6 (3.2)</td>
<td>31.4 (9.3)</td>
<td>12.3 (4.8)</td>
<td>0.01</td>
<td>0.06</td>
<td>0.01</td>
<td>0.08</td>
<td></td>
</tr>
<tr>
<td>B plot invasive stems/m$^2$</td>
<td>0.6 (0.3)</td>
<td>0 (0)</td>
<td>15.6 (2.7)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>C plot native stems/m$^2$</td>
<td>15.6 (4)</td>
<td>20.4 (5.2)</td>
<td>14.2 (3.8)</td>
<td>0.32</td>
<td>0.98</td>
<td>0.20</td>
<td>0.18</td>
<td></td>
</tr>
<tr>
<td>C plot invasive stems/m$^2$</td>
<td>0 (0)</td>
<td>0 (0)</td>
<td>6.3 (2.2)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
<td>0.43</td>
<td></td>
</tr>
</tbody>
</table>
Table 10. Mean, standard error, and Kruskal-Wallis (KW) p-values for the stream habitat and soil behavior response variables in the three plot types: Lower Watershed (LW), Upper Watershed (UW), and Invaded (I) in the Lancaster, N.H. study sites (n=36). Additional post-hoc Wilcoxon test on each pair to compare response variables between two treatment types. Bold p-value indicates significance at \( \alpha = 0.05 \) level.

<table>
<thead>
<tr>
<th>Response Variable</th>
<th>Lower (LW) Mean (SE)</th>
<th>Upper (UW) Mean (SE)</th>
<th>Invaded (I) Mean (SE)</th>
<th>KW p-value</th>
<th>Wilcoxon p-value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stream edge (B plot) canopy closure (%)</td>
<td>73.8 (5.9)</td>
<td>85.4 (2.7)</td>
<td>45.7 (8.4)</td>
<td>&lt;0.01</td>
<td>0.02</td>
</tr>
<tr>
<td>Open canopy angle (%)</td>
<td>19.1 (6.1)</td>
<td>5.6 (2.1)</td>
<td>45.7 (6.6)</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>CWD at stream edge (m³/ha)</td>
<td>35.7 (20.6)</td>
<td>22.9 (9.4)</td>
<td>0 (0)</td>
<td>0.02</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>CWD total (stream &amp; upland) (m³/ha)</td>
<td>45.3 (15.8)</td>
<td>44.9 (10.8)</td>
<td>17.7 (8.9)</td>
<td>0.05</td>
<td>0.14</td>
</tr>
<tr>
<td>Litter &amp; duff depth (cm)</td>
<td>2.5 (0.5)</td>
<td>8.8 (3.1)</td>
<td>1 (0.4)</td>
<td>&lt;0.01</td>
<td>0.03</td>
</tr>
<tr>
<td>Infiltration rate (cm/min)</td>
<td>8.1 (2.5)</td>
<td>25.7 (5.8)</td>
<td>12.6 (3.8)</td>
<td>&lt;0.01</td>
<td>0.01</td>
</tr>
</tbody>
</table>
Discussion

The results from this study support my hypothesis that riparian sites dominated by invasive plants will have degraded forest structure and stream habitat quality and also experience negative impacts to soil functioning and composition. As I will discuss below, degradation of these ecologically important habitats has many implications to ecosystem functioning and natural processes that can also hinder the provisioning of ecosystem services. Many of the response variables showed significant differences when comparing Invaded (I) plots to non-invaded plots (LW and UW). The largest differences can be seen within the first 5 m of the riparian buffer, known as streamside (subplot B). This is because knotweed is typically found at the water’s edge where its propagules first come into contact with moist soil and begins to establish. From there its rhizomes spread, and the plant begins to occupy the surrounding areas. But at the streamside is where it has the highest stem density and the most deleterious effects (Fig. 8). An alternative hypothesis or explanation is that invasive plants have colonized these areas because of a disturbance that may have degraded forest structure and compromised native communities. Invasive plants may have been able to succeed in these degraded stands with increased light and available niche space instead of creating these disturbed conditions. Although, if the alternative hypothesis is true, invasive plants will most likely prohibit these stands from recovering and degraded conditions will most likely persist.

Impacts to Forest Structure

Riparian sites dominated by invasive plants experienced negative impacts to all levels of forest structure (i.e., soils, ground cover, CWD, stem composition, understory, and overstory canopy). Invaded plots generally had greater stem density than Lower Watershed plots (Fig. 13;
This most likely occurs because the abundant knotweed and honeysuckle in these stands are fast-growing and highly competitive. Another explanation is that greater stem density could be a result of higher levels of light entering these plots. The greater stem density is restricted to small stems (< 5 m) since knotweed, Morrow’s honeysuckle, and glossy buckthorn only grow to be 3-5 m tall. Therefore, these densely invaded stands have a lower stem density of large stems and overstory trees (> 5 m), and tree basal area in Invaded plots was much lower compared to all other plot types (Fig. 14; Table. 9). This shift in forest structure (e.g., canopy composition, basal area, and stem density) in invaded stands can have many negative impacts to ecosystem functioning and ecosystem services. Some of these include ability to shade streams, water temperature, water quality, tree regeneration, ground cover, CWD, bank stabilization, and carbon storage.

In Invaded plots, the total volume of CWD was significantly lower when compared to other plot types and no CWD was present along the stream channel (Fig. 18). Although not statistically significant, the amount of CWD upland from the stream was also lower in Invaded plots (Fig. 18). This reduction of CWD has many impacts to the habitat as large decaying logs create moist, suitable habitats for amphibians and decomposers and also act as nurse logs that supply moisture and suitable microsites for many species of regenerating trees. An alternative explanation is that if these invaded stands were previously disturbed with fewer overstory trees, these stands would have reduced tree basal area and less production of CWD as a result of the disturbance and not the invasion. Upland mean canopy closure was slightly higher in Invaded plots when compared to Lower Watershed plots (Fig. 15). This is largely because of the dense stands of knotweed and their large, broad leaves in the understory that may further reduce native tree regeneration. As a healthy, native canopy shifts to a reduced overstory canopy and a dense
understory, the quality of shade and available light also shifts from high-shade to low-shade. High-shade produced from a healthy overstory canopy results in more diffuse light that is beneficial to seedling establishment, tree regeneration, and plant growth (Ashton and Kelty 2018). Low-shade does not produce as much diffuse light as high-shade. This reduction of high-quality light can be detrimental to the regeneration of native trees by reducing seedling survival and plant growth (Ashton and Kelty 2018).

The hindrance of regenerating native tree species caused by terrestrial invasive plants is well documented, although studies in northeastern riparian forests are lacking and impacts may vary depending on the specific habitat, invading species, and the species being replaced (Frappier et al. 2003; Levine et al. 2003; Vilà et al. 2011; Hamelin et al. 2016). A clear trend emerges on Garland Brook when assessing the density of native tree regeneration (Fig. 17) and the spatial distribution of invasive plants (Fig. 12), although not statistically significant at $\alpha=0.05$ level. Native tree regeneration is much less at the streamside (subplot B) in Invaded plots than in any other plot or subplot type (Fig. 17). In Invaded plots, subplot C also had much lower densities of native tree regeneration compared to non-invaded plots (LW and UW) (Fig. 17). Less native tree regeneration in invasive stands is most likely caused by the aggressive growth of knotweed and Morrow’s honeysuckle and their dense understory canopies limiting available high-quality light. To further exacerbate this problem, the potential allelopathic properties of knotweeds can further influence the reduction of native tree regeneration and ground cover in invaded stands (Murrell et al. 2011; Abgrall et al. 2018). When compared to the Lower Watershed plots that were dominated by native vegetation, Invaded plots had much higher amounts of bare ground (Fig. 20) and also slightly less ground cover on the forest floor (Table 9). This combination of more exposed, bare mineral soil and less ground cover further increases the
potential for runoff, erosion, sedimentation, and future bank stabilization issues (Arnold and Toran 2018; Hale et al. 2018). Furthermore, having less organic material (CWD, leaf litter, duff) and less ground cover may create a drier habitat. This can be detrimental for many flora and fauna species that depend on mesic riparian habitats.

**Impacts to Soil Composition and Functioning**

Soil composition analysis revealed that Invaded plots have less litter and duff layers present in the soil (Fig. 19). Typically, soils with less organic material have slower infiltration rates (USDA-NRCS 2014; Sun et al. 2018). This increases the potential for greater overland flow, which facilitates the loss of topsoil from the terrestrial habitat and also can allow sediment and excess nutrients to enter the stream. Furthermore, the lack of organic material in upper soil horizons and less CWD acting as nurse logs in these stands, may further reduce species diversity and native riparian canopy trees. Upper Watershed plots were found to have significantly higher infiltration rates because of the substantial amount of litter and duff found in the mature, closed canopy forests of the upper watershed (Table 10). Here, the organic layers in the soil were three times greater than the Lower Watershed and up to eight times higher than Invaded plots (Fig 19; Table 9). This positive relationship between soil organic material and infiltration rates is well documented (USDA-NRCS 2014; Sun et al. 2018). Surprisingly, infiltration rates were higher in Invaded plots when compared to Lower Watershed plots, despite having less litter and duff present (Table 10). This is likely because knotweed has the ability to alter the soil composition, mostly through secondary metabolites and aggressive growth of root and rhizome structures (Murrell et al. 2011; Abgrall et al. 2018). In our field observations, knotweed creates an abundance of bare ground, although it has very loose soil structure. This would decrease soil
compaction and density, which would also increase soil infiltration rates (USDA-NRCS 2014; Sun et al. 2018).

**Impacts to Stream Habitat**

Riparian sites dominated by invasive plants also experienced many negative impacts to stream habitat quality, most of which were caused by degraded forest structure and soil composition. Mean canopy closure at the stream channel was significantly lower in Invaded plots when compared to the two other plot types (Fig. 15). This reduces the riparian buffers ability to shade streams, which increases water temperature and has cascading effects to dissolved oxygen levels, water quality, and degradation of habitat for brook trout and other aquatic organisms (Richardson and Danehy 2007; Wilkerson et al. 2010). Open canopy angle was significantly higher in Invaded plots compared to Lower Watershed and Upper Watershed plots (Fig. 16). This lower amount of stream canopy closure and increased open canopy angle is best explained by less overstory trees in invaded stands and the abundance of short overarching knotweed stems along the stream. Solar radiation entering the stream is the main factor for increases in water temperature, which is detrimental to the survival and fitness of brook trout (Pusey and Arthington 2003; Kratzer and Warren 2013; Kanno et al. 2015). This is particularly important in New England because Kratzer and Warren (2013) have identified water temperature and CWD as the two most important limiting factors for brook trout streams in the region, with reported negative effects to survival and mean weight occurring with water temperatures greater than 20° C and less than 200 pieces of CWD per ha.

In Invaded plots, no CWD was found along the stream channel that was believed to be generated on site. The lack of CWD has many direct and indirect negative effects in stream
ecosystems. CWD is responsible for increasing habitat complexity by creating log jams and deep pools that are important refuge for stream fish, especially during high and low flow events (Sweka and Hartman 2006; Kratzer 2018). These deep pools often contain colder water temperatures, which is imperative for cold water fisheries and other aquatic organisms (Kratzer and Warren 2013). Furthermore, pools formed by log jams allow sediment to settle out of the water column and be stored along with organic material that serves as a food source base for organisms of lower trophic levels such as aquatic macroinvertebrates (Sweka and Hartman 2006). Additionally, increasing the amount of CWD in northern New England streams has been shown to significantly increase mean brook trout biomass (abundance and weight) (Kratzer 2018).

Although Invaded plots have higher infiltration rates that are beneficial to reduce overland flow potential, substantial topsoil loss and sedimentation is still occurring in these stands. This can be seen when analyzing the amount of silt particles on and around stream substrate (embeddedness). Stream substrate downstream of large invasive communities had a much higher percentage of silt blanketing the substratum when compared to the amount of silt covering the substratum before these large communities of invasive plants (Fig. 21). This increase of silt on stream substrate can have many direct impacts to aquatic organisms such as brook trout and their primary food sources, aquatic macroinvertebrates. In addition, a lack of CWD and debris dams may further exacerbate embeddedness and sedimentation. Studies show that increases in embeddedness can reduce abundance, weight, and condition of cold-water stream fish and also the abundance and diversity of macroinvertebrates (Haro and Brusven 1994; Bolliet et al. 2005). Embedded and smothered substratum can reduce foraging habitat for trout and other stream fish, as many macroinvertebrates such as stoneflies and mayflies are found in unsedimented and unembedded cobble, with nymphs often found on the undersides of
unembedded cobble (Haro and Brusven 1994; Bolliet et al. 2005). Embedded cobble has also been linked to decreases in shelter for fish, therefore increasing intraspecific competition and territoriality (Bolliet et al. 2005). Ultimately, embedded streambeds may destabilize predator-prey dynamics and convolute ecological relationships between aquatic organisms that may have co-evolved in unembedded cobble environments (Haro and Brusven 1994).

In stands heavily invaded by knotweed, the combination of loose soil structure, exposed bare ground, and a lack of organic material in conjunction with aboveground growth that dies back in the fall and leaves banks exposed during snowmelt and heavy spring rain events is a large factor to the increased amounts of embeddedness and sedimentation downstream of invaded stands. Arnold and Toran (2018) also found similar results of increased amounts of sediment and embeddedness downstream of invasive plants, specifically knotweed. Furthermore, Hale et al. (2018) reported similar results when losses of surface coarse particulate organic matter (CPOM) were greater in riparian areas that lacked structure and diversity, particularly groundcover. Hale et al. (2018) also reported that CWD may play a role in retaining CPOM in the riparian zones and mitigating topsoil loss into streams. A large body of evidence suggests that increased amounts of bare ground is an important driver of topsoil and sediment entering the streams (USDA-NRCS 2014; Hale et al. 2018). Furthermore, the presence of knotweed has also been linked to increased erosion rates and bank collapse (van Oorschot et al. 2017; Arnold and Toran 2018).

A potential source of error with embeddedness measurements could have arisen when stands of invasive plants extended beyond the 10 m plot (e.g., some stands were >100 m long). This may cause the downstream habitat to differ from the upstream habitat (pool, riffle, run, cascade), which could influence the amount of sedimentation and embeddedness in the transect.
When applicable and whenever possible, this was avoided by strategic placement of embeddedness transects by shortening the distance between upstream and downstream transects in large communities to retain similar habitat.

**Riparian Invasion Along Headwater Streams**

All invasive plant occurrences along Garland Brook were in the lower watershed with 324 patches detected along both sides of the brook. Surprisingly, land cover analysis revealed that Invaded plots had less roads and development nearby than Lower Watershed plots (Table 8). Our ocular search width was an average of 15 m, therefore making the lower watershed surveyed land area was roughly 467,700 m². Invasive plant occurrences occupied approximately 3.2% of the total surveyed lower watershed riparian zone. The invasive plant species found furthest up the watershed was Morrow’s honeysuckle detected along the stream adjacent to the drinking water supply treatment facility access road (Fig. 11). Two individual plants were found at the facility on opposite banksides and were ~300 m upstream from the closest downstream individual. This was the last developed site in the Garland Brook watershed before entering intact and undisturbed forest of the upper watershed and the WMNF. This warrants concern as many invasive honeysuckles were found growing as advanced regeneration along trails in closed canopy forests. Any further distribution up the watershed would be approaching the closed canopy, intact forests of the WMNF and will most likely go undetected along these small headwater streams with infrequent human travel. Morrow’s honeysuckle is successful at dispersing long distances and traveling upstream as it produces large berries that contain multiple seeds and are dispersed primarily by birds (IPAUS 2018). A narrow strip of the WMNF lands extend down from its core area in the upper watershed and comes in close proximity to
residential property and a public road/stream crossing near Garland Mill (Fig. 11). In this lower section of the WMNF, downstream of the water treatment facility, six patches of Morrow’s honeysuckle were detected (6.5 m$^2$ mean size). Morrow’s honeysuckle was most abundant on the northern reach of the bifurcation of Garland Brook. This area was historically used for agriculture and contains the largest pasture in the study area. A 440 m$^2$ patch of Morrow’s honeysuckle was found here along with many other large patches. In these heavily invaded areas, beavers (*Castor canadensis*) were frequently using Morrow’s honeysuckle as building material for their dams, transporting branches filled with berries into the moving water and dispersing seeds further downstream.

Knotweed was by far the most abundant invasive species observed in our study area. Expansive monocultures of knotweed were found along Garland Brook’s lower watershed, usually in disturbed or open areas but not exclusively. Surprisingly, in a 75 m wide riparian forest between Garland Brook and the adjacent large pasture heavily invaded by Morrow’s honeysuckle), a 1,620 m$^2$ patch of knotweed was found in the understory where it appears to have exploited a gap in the canopy. Since no patches of knotweed were found upstream of this community and it was found upland from the brook, its origins are most likely explained by the historic agricultural land use of this location. Knotweed was most likely planted at this location or propagules were introduced in contaminated landfill. This was also supported by the landowner (verbal communication, May 2019) that stated (circa 1980) a relative who owned the farm often used heavy machinery to excavate soil from the streambanks, transporting the fill offsite and dumping it onto other areas of the property. Other than this large upland community on the northern side of the bifurcation, all other knotweed was found on the south reach downstream of the heavily invaded Garland Mill or downstream of where the two bifurcated
reaches converge. After Garland Brook flows through Garland Mill on the southern reach, it enters a large section of intact, closed canopy forest. Surprisingly, a few isolated patches of knotweed were found along the banks of these closed canopy riparian forests, downstream from disturbance. These communities were not as robust as the large monocultures growing in disturbed open areas, but some were quite dense and ranged from 0.1 m$^2$ (small individual stem) to 66 m$^2$ in size and seemed to vary depending on available light through the overstory. Other than the source population at Garland Mill, another estimated source population resides where Garland Brook meets Brook Road. Here, a 1967 m$^2$ monoculture (91.5 m x 21.5 m) occupies the entire bank and further facilitates invasion and establishment of knotweed in the lower watershed. This community was planted here (circa 1970) and has expanded a few meters annually (verbal communication with landowner, June 2018). Many large patches of knotweed can be found downstream of this extensive monoculture.

Knotweed was only found downstream of disturbed and heavily invaded areas, which was expected as it primarily reproduces by vegetative propagation (Forman and Kesseli 2003; Colleran and Goodall 2014; Cygan 2018). Knotweed also reproduces sexually, although germination of seeds and survival of seedlings have a low success rate and can vary depending on many environmental factors and may be restricted to hybrids (Forman and Kesseli 2003; Colleran and Goodall 2014). Reproducing vegetatively allows knotweed to quickly grow a new plant from a small rhizome or stem fragment (~1.25 cm) that contains at least one node (Colleran and Goodall 2014; Cygan 2018). These plant propagules (stem or rhizome fragments) are commonly washed downstream because knotweed’s brittle stems often overarch the stream and are easily broken off during high-flow events. Furthermore, it is often believed that knotweed’s shallow roots and poor ability to stabilize banks can cause them to collapse into the stream,
allowing individual plants, stems, and/or root and rhizomes fragments to enter the stream (van Oorschot et al. 2017; Arnold and Toran 2018).

Glossy buckthorn was also found along Garland Brook. The majority of glossy buckthorn was found in the lower sections of the watershed, along stream and road crossings and agricultural land use. One small individual was found in a closed canopy forest downstream of a culvert near Garland Mill, where a larger individual resides. This is common as glossy buckthorn prefers a lot of sunlight and very wet areas, even though it is often considered shade tolerant and can also colonize less disturbed habitats (Lee and Thompson 2012; IPANE 2019).

**Conclusion**

In the Garland Brook watershed, many invasive plants are dispersing downstream of disturbed areas although some appear to have the ability to disperse upstream. These invasive plants also have the ability to exist in closed canopy forests at lower densities as advanced regeneration. These semi-shade tolerant invasive species seem to be limited by light but have the ability to wait for a disturbance or canopy gap to increase their dominance. Many large invasive communities occupy the lower watershed of Garland Brook where there is a high amount of agricultural land use and reduced forest cover. If these invaded novel ecosystems expand into the upper watershed, they may pose a threat to the purity of Lancaster’s water supply, Garland Brook’s ecosystem integrity and functioning, and ability to provide critical habitat for fish and wildlife.

Negative impacts of invasive plant occurrences can be seen in all levels of forest structure, from soils to the overstory canopy. Impacts include reductions of organic inputs to streams and upland forests, loss of structural support by native trees, and reduced ability of riparian zones to
regulate stream temperature. Stream habitat is further degraded by silt and erosion entering into streams. This has many implications to water quality and with less CWD and pool formation, there is less opportunity for silt and fine sediments to settle out of the water. Finally, all of these responses can have additional cascading effects to riparian ecosystems, streams, and wildlife habitat. This study provides useful insight into mid to long-term impacts from invasive plants to riparian forests and headwater stream ecosystems. It is important to bear in mind that significant differences between plot types does not infer the cause of the response. This may be more easily deduced in some response variables over others. Ultimately, further research is warranted to fully determine causation of some of the response variables of this study.

**Management**

This research provides important baseline data on the spatial distribution and invasion patterns of terrestrial invasive plants in riparian forests along headwater streams of northern New England. The upper Connecticut Cooperative Invasive Species Management Area (UCCISMA) and Trout Unlimited (TU) have been conducting roadside and stream surveys to gather point location data for invasive plant occurrences in the upper Connecticut River watershed. In addition to supplying location data of invasive plants to the WMNF agency, these results will be added to the invasive plant location data from this study to the UCCISMA and TU database. Having a better understanding of invasion dynamics and impacts caused from invasive plants in headwater streams and their riparian forest counterpart is imperative to maintain the ecological integrity of these species-rich and ecologically important habitats. These areas are often overlooked, and their natural processes provide an immense amount of services for humans and many other organisms. This study provides useful insight into the role that native vegetation has
on ecosystem integrity and functioning, and how this may be hindered with changes to plant communities and species composition. The information from this study can also provide better understanding on which ecosystem functions are at risk from invasion, which functions may need to be restored and maintained, and how to minimize this risk to ensure the provisioning of ecosystem services.
Chapter IV Synthesis

Results and Relationships Between the Two Investigations

This study provides better understanding of the short-term and long-term impacts of terrestrial invasive plants on streams and natural and restored riparian forests in northern New England. In Chapter II, my research in central Vermont provides important short-term baseline data on the impacts of invasive plants, the success of active restoration, and the ability of biotic manipulations to increase resistance in native riparian communities. In chapter III, I provide a better understanding of the spatial distribution of terrestrial invasive plants along headwater streams in northern New England. Additionally, I provide baseline data on the mid to long-term impacts of invasive plants to riparian forests and headwater stream ecosystems in northern New Hampshire. My study shows that invasive plants can have deleterious long-term impacts to native vegetation, riparian forest structure, soil composition, soil function, stream physical habitat, and the potential for impairment to ecosystem services. I believe that the observed impacts along Garland Brook in northern New Hampshire will also be the fate of many riparian areas along the White River and its tributaries and other areas of central Vermont over longer time scales.

Intellectual Merit and Broader Impacts

It is imperative to understand the impacts of invasive plants in New England’s heterogeneous landscapes, as they can often vary depending on the species being replaced.
This research fills an important knowledge gap on invasive plants in northern New England as this is one of the first studies in the northeast to assess the effectiveness of active restoration following a major natural disturbance and also quantify the impacts of invasion to riparian forests and headwater stream ecosystems and functioning. Much of the research conducted on terrestrial invasive plants has been focused on their distribution on the landscape, how historical and present land use patterns can influence invasion, and the impacts to interior forests and commercial tree species. A better understanding of invasion dynamics is important in New England’s heterogeneous landscapes because non-native plant species are estimated to comprise 30% of New England’s flora, of which 3-5% of these are considered invasive (Mehrhoff et al. 2000; Allen et al. 2013). Furthermore, having more insight on how invasive plants can negatively impact ecosystem services such as water quality, bank stabilization, fish and wildlife habitat, and tree regeneration is imperative as storm events (i.e., natural disturbance) and invasive plant colonization is often projected to increase. This is also essential in ecologically important areas and critical fish and wildlife habitats that need to be protected such as the Garland Brook watershed, which also serve as the town drinking water supply.

The findings of this research can assist land managers and conservation efforts of headwater streams and riparian forests in New England to further understand the impacts and distribution of invasive plants. Furthermore, this study may assist conservation efforts to minimize the risk of invasion and mitigate the effects on ecosystem services and functioning. The findings of this research have been and will continue to be directly presented to policy makers and land managers (state, federal, and non-profit) at regional watershed, invasive plant, and forestry conferences and working group meetings. Point location data of invasive plants will be provided to stakeholders and interested agencies including: Trout Unlimited, Upper
Connecticut Cooperative Invasive Species Management Area (UCCISMA), and the WMNF. Additionally, plot locations and vegetation surveys may serve as valuable baseline data for long-term monitoring of these sites, especially useful to the WRP to gauge long-term success of their restoration projects and also future research projects at the University of New Hampshire.

Management Implications

Early detection and rapid response are common themes when managing invasive species, as eradication is often only feasible for invaded sites that are detected soon after invasion. Detecting and managing these invasions shortly after their arrival will increase the potential for success as populations will be smaller in size, easier to control, and costs are reasonable as resources are often limited. This is particularly true for species that store energy in their aggressive rhizome structures, such as invasive knotweeds. The more time aboveground biomass is able to persist, the greater amount of energy that will be stored. Over time, these populations may become unmanageable and the potential for eradication will greatly diminish.

One approach for managing invasive plants used by the New York Department of Environmental Conservation (NYSDEC [date unknown]) and the Vermont Fish & Wildlife Department (verbal communication and presentation, May, 2019) is to prioritize control efforts by the size and intensity of the invasion. Using this approach, control work should begin where invasive plant communities are smaller and less-dense and work toward areas more heavily invaded. Large, heavily invaded areas are left for last as they may be labor intensive and very difficult to manage. Although this approach does prevent smaller infestations from becoming unmanageable, this may not be the best approach for invasive species that are dispersing through moving water. Using this approach with species such as knotweed, large source populations may
not ever be controlled and will consistently be dispersing plant propagules downstream into new areas. In this case, the context of the landscape and areas surrounding large source populations should be considered, especially when protecting critical and natural habitats. Source populations may need to be targeted and eradicated or suppressed to limit propagule pressure and dispersal into adjacent areas. If a large invasive community is removed, planting or seeding native species may be needed to successfully restore riparian habitats and ecosystem functioning as native species propagule pressure may not be sufficient for re-colonization of native communities (Richardson et al. 2007).

After the removal of an invasive species that creates openings, available resources, or niche space, it is important to take restoration efforts to prevent new invasive species from colonizing or re-invading an area and to restore ecosystem functioning (Clements et al. 2016). Biotic manipulations (e.g., planting native vegetation) can be used to restore native plant communities when a riparian zone has been recently invaded (Richardson et al. 2007). As my results show, this can also increase the competitive ability of native communities and can be a successful tool when preventing invasion after a natural or anthropogenic disturbance, although early detection and rapid response still applies. It is important to plant fast-growing native species immediately after a disturbance to give them an adequate head start to compete with aggressive invasive species. To increase the success of suppression or eradication of invasive plants, it is my recommendation that a suite of strategies may be needed, especially when combating invasive knotweeds. These include multiple removals of aboveground biomass and/or roots and rhizomes when possible, smothering the area with thick black plastic and mulch (Cygan 2018), in conjunction with planting native vegetation, and monitoring. Invasive knotweeds impacts from allelopathic substances and secondary metabolites to soil chemistry,
microbial communities, and native plants has been of recent interest (Murrell et al. 2011; Abgrall et al. 2018). There seems to be limited knowledge of the direct and indirect effects of knotweed after removal of knotweed communities. This is because allelopathic substances emitted from knotweed can have lasting impacts to soil chemistry and soil properties that may hinder native regeneration (Murrell et al. 2011; Abgrall et al. 2018; Cygan 2018) although re-establishment of native communities is possible. For example, a study by Urgenson et al. (2014) reported that in the Pacific Northwest, native broadleaved and coniferous trees, shrubs, graminoids and forbs quickly re-colonized a riparian forest where Bohemian knotweed was removed.

Limitations of the Study

Potential limitations of this study include small sample sizes in each of the independent investigations. Data collection for both investigations was limited to one summer field season. This restricted me to 41 vegetation surveys in the 5 paired study sites (n=5) in the White River watershed (chapter II) and 36 plots between the three plot types, along with stream surveys in the Garland Brook watershed (chapter III). In addition to small sample sizes, my data was not normally distributed, and the variances were often unequal, even after data was log transformed or arcsine square root transformed. It is for these reasons that non-parametric statistical analysis was needed. Unfortunately, this may have reduced statistical power and hinder the ability to detect statistical differences. Another limitation of this study is determining causation. If statistical analysis has detected relationships between independent variables (i.e., plot types) and dependent response variables, this does not infer that the independent variable caused the response. Based on statistical analysis and additional field observations, I can say with some level of certainty that some response variables were directly or indirectly caused by terrestrial...
invasive plants, although causation on other response variables may be more challenging to
determine, even if they seem to be correlated.

**Future Research**

Data collected along the White River on the short-term impacts, the success of restoration,
and the ability to increase resistance can provide useful baseline data for future studies looking at
longer-time scales. Determining if these restoration efforts are even more or less successful and
which ecosystem functions can be preserved and maintained by restoration efforts over longer-
time scales may help to guide management and conservation of riparian areas. In the Garland
Brook watershed and all of northern New England, future research is needed on the impacts of
knotweed to wild brook trout populations and native brook trout habitat. This is warranted
because two of the most significant findings in my study in chapter III was the reduction of
CWD and organic inputs available to streams and also the increased potential for solar radiation
to enter the streams. These two variables have also been identified by Kratzer and Warren (2013)
as the two most important limiting factors for brook trout streams in the region (i.e., lack of
CWD and increased water temperatures). Additionally, further molecular research is also needed
to confidently identify the genotype of knotweed in New England and detect any presence of
possible hybrids. This is important to assess their ability to reproduce sexually and their
mechanisms for dispersal (e.g., wind dispersed seeds, vegetative propagation). Furthermore, this
is needed to effectively and efficiently reduce their establishment and spread on the landscape
and to mitigate the impacts to ecosystem functioning and ecosystem services.
LIST OF REFERENCES
References


